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Dynamic economic valuation of coastal wetland restoration: A nature-based solution for climate and biodiversity

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ABSTRACT

This paper explores the dynamic linkage between coastal wetland restoration and the resulting economic benefits, with a focus on nonmarket values such as climate regulation and biodiversity conservation. Coastal wetlands are recognised as highly effective natural carbon sinks, offering significant ecosystem services that contribute to climate change mitigation and adaptation. By utilising a modelling framework that integrates ecological recovery processes and economic valuations over a 100-year period, we provide insights into optimising long-term returns from wetland restoration. This study emphasises the importance of accounting for the temporal dynamics of ecosystem recovery, highlighting the lag between restoration activities and full ecosystem functionality. Our findings highlight the importance of nature-based solutions in global climate finance strategies and emphasise the need for more accurate, targeted investment in wetland restoration. This approach ensures that resources are allocated efficiently over time, maximising the benefits of enhancing coastal resilience and achieving long-term climate goals.

1. Introduction

In recent years, there has been a growing recognition of the critical role that nature-based solutions (NBSs) can play in mitigating greenhouse gas (GHG) emissions while simultaneously addressing biodiversity loss and enhancing ecosystem resilience (Paul et al., 2024). Coastal wetlands, including mangroves, saltmarshes, and seagrass beds, are among the most effective natural carbon sinks, sequestering carbon at rates significantly higher than those of terrestrial forests (McLeod et al., 2011). Coastal wetlands also serve as natural buffers, mitigating climate change by absorbing and dissipating wave energy, which reduces the impact of coastal flooding and erosion (Barbier, 2019; Friess et al., 2020). Unlike rigid and often costly engineered solutions such as sea walls and stop banks, wetlands offer a flexible, self-sustaining approach that can adapt to changing conditions over time (Temmerman et al., 2013). Additionally, wetlands support important fisheries and are vital for wider biodiversity protection (Craft et al., 2003; Kingsford et al., 2016). They provide essential habitats for a wide array of species, supporting rich biodiversity and maintaining ecological balance.

The economic value of wetland restoration has gained increasing attention in recent years. Numerous studies have estimated the nonmarket or intangible benefits of restored wetlands. For example, the value of coastal wetlands in providing storm protection services alone has been estimated to be substantial, with wetlands in the United States providing annual flood protection benefits valued at approximately 34 billion USD (adjusted to 2024 dollars) (Costanza et al., 2008). In addition to flood protection, restored wetlands can also offer recreational, cultural, and aesthetic benefits that contribute to local economies and improve quality of life (Chen et al., 2009; Costanza et al., 1989; Ghermandi et al., 2010). Recognising these benefits, global initiatives such as the Global Mangrove Alliance and the Blue Carbon Initiative have been advocating for scaled-up funding to restore and protect these vital ecosystems as part of an integrated approach to climate mitigation and adaptation.[1] However, a significant gap persists between the financial resources allocated to engineered solutions and those directed towards NBSs (WBCSD, 2020). To address this disparity, it is crucial to validate the long-term economic benefits of investing in NBSs, particularly in the context of climate change mitigation and adaptation.

Despite numerous studies estimating the value of wetland restoration, impact assessments rarely integrate both ecological and economic aspects. A review of studies on the impact of mangrove restoration revealed that only 4.3 % of the studies (eight out of 186) reported both

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[1] More information for Global Mangrove Alliance at https://mangrovealliance.org/ and Blue Carbon Initiative at https://www.thebluecarboninitiative.org/.

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ecological and economic outcomes (Liu et al., 2024), with economic valuations largely based on market prices for provisioning services such as food production (Liu et al., 2024). However, current approaches often assume a direct, static relationship between restoration efforts and economic value, overlooking temporal dynamics and ecological complexities involved (Bulmer et al., 2024; Chaikumbung et al., 2016; Chen et al., 2009; Costanza et al., 1989; Dumax and Rozan, 2021; Ghermandi et al., 2010). This oversimplification can lead to inaccurate estimations in the impact assessment of wetland restoration (e.g., cost-benefit analysis (CBA)), failing to capture the evolving benefits of restoration, including the role in mitigating climate change impacts and enhancing ecosystem resilience.

Moreover, restored wetlands often take time to regain full functionality. For example, wetland functions associated with the detention of precipitation and floods rapidly increase under post-restoration conditions, whereas improvements in wetland habitat functions (associated with forest establishment and maturation) require additional time (Berkowitz, 2019). The trajectory of ecosystem service values during wetland restoration needs more attention. A meta-analysis of 621 wetland sites worldwide found that even a century after restoration efforts, biological structure and biogeochemical functioning remained, on average, 26 % and 23 % lower, respectively than those at reference sites (Moreno-Ger et al., 2012). Recent studies highlight the link between thresholds in restoration functions and ecosystem service values. For example, Zhou et al. (2020) reported that the value of all ecosystem services increases as wetland area increases, with an average elasticity of 0.83; Tomscha et al. (2023) reported that nitrogen retention targets were mostly met when the percentage of wetlands restored exceeded 60 %. However, the effectiveness of restoration targets across spatial scales remains unclear. For example, ecosystem services change linearly with the area restored at the basin scale but nonlinearly at sub-catchment scales (Tomscha et al., 2023). Purandare et al. (2024) proposed that wetland restoration projects should closely monitor the recovery of ecosystems, following measurable ecological indicators (Gann et al., 2019).

Adding to these challenges is the difficulty in monetising the nonmarket values of wetland ecosystem services. Most nonmarket valuation (NMV) studies rely on stated preference methods (i.e., asking people's willingness to pay/accept for wetland restoration through surveys) that are restricted by the cross-sectional nature of the survey conducted and hence provide only point estimates of the benefits at a year of the survey or a given period (e.g., five years (Ndebele and Forgie, 2017; Pattison et al., 2011)). There are only a few exemptions. Marre et al. (2015) considered the temporal effects of wetland restoration in a choice experiment, i.e., preservation for 20, 50, and 100 years for various conservation attributes, such as the quantity of fish, animals, and coastal and lagoon natural landscapes. The results show that the benefits derived from preserving various aspects of a marine ecosystem changed over time in a nonlinear manner. Hagen et al. (2017) estimated the recreation benefits of NBSs for wastewater treatment over 14 years (2002–2015) using a hedonic pricing model. They reported that the price premium associated with properties adjacent to the constructed wetland park and lagoon increased nonlinearly over time. Notably, inaccurate small-scale estimates can lead to large errors when scaling up to achieve broader climate adaptation goals, resulting in suboptimal policy-making decisions that fail to fully leverage the potential benefits of nature-based solutions (Araya-López et al., 2024; Moorhead, 2013).

Our study proposes a modelling framework designed to integrate ecosystem responses to coastal wetland restoration efforts with the economic valuation of ecosystem services. This framework serves as a tool to account for the dynamic processes of restoration and the resulting economic benefits over time. We utilise examples of climate regulation and fishery habitat protection to numerically demonstrate and validate the framework, leveraging available data. These examples highlight how the model can quantify and evaluate the substantial economic benefits provided by coastal wetlands in terms of both climate adaptation and ecological preservation. The flexibility of our framework allows for its application to various ecosystem services as new data and information become available. By incorporating ecological functions and economic valuation techniques, we aim to provide a comprehensive understanding of how restoration efforts translate into market and nonmarket economic value over time. This modelling framework allows for the consideration of the dynamics of the valuation of wetland restoration over 100 years, considering 1) ecosystem responses to restoration processes and 2) shifts in human value as ecosystems recover over time. We consider nonlinear relationships and threshold effects in the model to offer a robust framework for decision-making in wetland management and climate adaptation strategies. This integrated approach enhances the effectiveness of wetland restoration policies and contributes to more resilient and sustainable coastal ecosystems. By providing evidence of the long-term economic benefits of wetland restoration, this research contributes to the growing body of evidence supporting the need for increased funding for nature-based climate solutions. The framework ensures that investments in coastal wetland restoration are optimised to yield maximum ecological and economic returns, making it a valuable tool for guiding sustainable restoration efforts.

This paper is structured as follows. Following the introduction, Section 2 presents the conceptual modelling framework, along with the models and simulation process developed to link restoration functions to the economic valuation of ecosystem services from wetland restoration. Section 3 details the results and discusses the findings under various scenarios, followed by Section 4 to concludes the study.

2. Methodology

2.1. The conceptual framework

There is a lag in achieving full functionality post-restoration of wetlands. Existing studies indicate that marine and coastal ecosystems require approximately 10 to 25 years to recover and become fully functional (Bulmer et al., 2024; Craft et al., 2003; Paul et al., 2024). As shown in Fig. 1, a comprehensive understanding is lacking regarding how ecosystems and their functions recover during wetland restoration, considering 1) the different ecosystem services (e.g., climate regulation, biodiversity, or recreation) and 2) the ecological indicators used to measure successful recovery. A range of indicators have been proposed for assessing the success of coastal wetlands, including plant and animal species composition and structural diversity, ecosystem function, and physicochemical conditions (Cadier et al. 2020). Importantly, vegetation establishment marks only the initial stage in recovery. The gradual accumulation and maturation of organic detritus from above and,

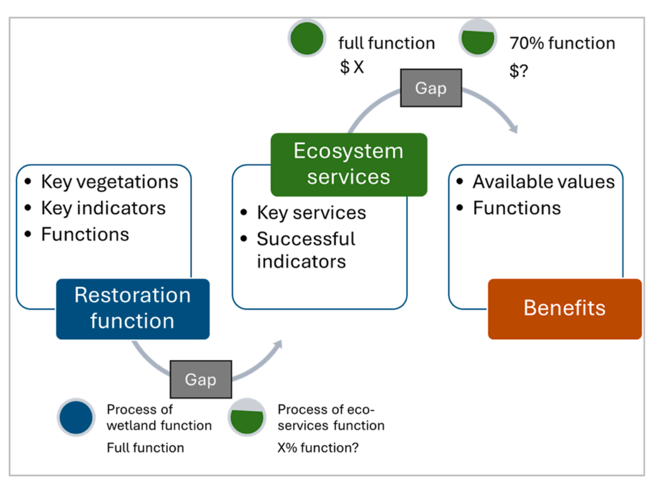


Fig. 1. Gaps in wetland restoration evaluation and the conceptual framework for valuing the associated benefits. Source: Authors.

particularly, below-ground plant biomass, as well as the trapping of sediments from the broader catchment, are crucial for restoring ecosystem functionality. For this reason, carbon storage/sequestration is a commonly referenced ecological indicator for measuring successful ecosystem recovery, which is closely tied to ecosystem integrity and functionality and is relatively easy to quantify (Liu et al., 2024; Moorhead, 2013)

However, recovery rates vary across different ecosystem services. For example, active carbon sequestration, sediment, and nutrient sequestration may recover at a faster rate than biodiversity, with instantaneous rates peaking near the start or at some midpoint of the restoration process before tapering off over time. These dynamics can be represented via different functions to better capture the varying trajectories of recovery (Liu et al., 2024; Moorhead, 2013). While qualitative assessments provide comprehensive frameworks to measure the levels of ecosystem recovery from coastal wetland restoration (Gabriel et al., 2019), they often lack the temporal mapping of recovery necessary to inform funding and policy decisions. Another gap is related to the human value system and ecosystem services. Many studies have attempted to estimate the value of wetland restoration via economic tools, presuming that values are associated with fully recovered ecosystems; at this time, ecosystems are assumed to recover and provide full services and values to society. This presumption can potentially misalign investment timing with benefit realisation, particularly in investment projects for wetland restoration aimed at climate change regulation. Investors often require a clear understanding of when benefits will be realised and how they will be tracked over time.

Considering these gaps, our framework suggests the use of varying functions to model different recovery rates of ecosystem services, providing a nuanced understanding of how restoration efforts translate into economic and ecological benefits over time. In the hypothetical example shown in the upper part of Fig. 2, wetland restoration achieves partial function from approximately year 5 (30 % full wetland function) to year 20 (80 %) and achieves full functionality only after 80 years. In this process, the ecosystem services associated with the wetland have also been built over time. Notably, while dependent on the location and biophysical characteristics of the study sites, the literature on wetland restoration shows that different ecosystem services respond differently to wetland restoration (Moreno-Ger et al., 2012; Tomscha et al., 2023). The lower part of Fig. 2 contracts a simplistic linear relationship, where both ecosystem and economic values increase proportionally (i.e., 30 %), with a more realistic nonlinear nexus. For instance, flood protection and biodiversity may recover earlier, whereas recreational/tourism service values may develop later, underscoring the need to incorporate temporal dynamics into economic valuations. Policymakers and restoration practitioners must recognise that while some functions may show rapid initial improvement, others will require significantly longer periods to fully recover.

2.2. The model

We constructed a model to validate the ideas proposed by the conceptual framework to link restoration functions to the economic values of ecosystem services provided by wetland restoration. First, we assumed that the ecosystem recovered to wetland restoration following the response function $S(t)$ at time t, i.e., the age of the planted vegetation of the restored wetland (e.g., saltmarsh and mangrove) and the value of recovering ecosystem services, such as climate regulation and biodiversity, may increase nonlinearly with $S(t)$. This relationship is modelled via a nonlinear function:

$$V(t) = V_{max}\left(1 - e^{-\lambda S(t)}\right), \tag{1}$$

where $V(t)$ represents the value of ecosystem services given the status of ecosystem recovery at time t, V_{max} is the maximum potential value of the benefits of recovering ecosystems, and λ is a parameter that defines the responsiveness of the value to changes in the response function. $S(t)$ is the ecosystem response function, such as the carbon sequestration rate (proximity for climate regulation) or species richness (proximity for biodiversity) at time t.

Recognising that certain benefits only materialise when specific restoration thresholds are met (e.g., nitrogen retention targets were mostly met when the percentage of wetlands restored exceeded 60 % (Tomscha et al., 2023)), we also modelled the relationship using a logistic growth function (Eq. (2)), following Ndebele and Forgie (2017)[2]:

$$V(t) = \frac{V_{max}}{1 + e^{-\kappa(S(t)-\theta)}}. \tag{2}$$

Here, κ determines the steepness of the curve and θ represents the threshold responsiveness of ecosystems at which significant benefits start to accrue.

To incorporate variability and uncertainty into our restoration function, we model the ecosystem response function $S(t)$ as a stochastic process. This allows for the simulation of scenarios where the restoration process is subject to random fluctuations or uncertainties over time. That is, we consider adding a noise term $\in(t)$ to the response function $S(t)$:

$$S(t) = S_{det}(t) + \in(t), \tag{3}$$

where $S_{det}(t)$ is the deterministic part of the response function, which is dependent on the ecosystem services focused on, in our case, climate regulation and biodiversity $\in(t)$ represents stochastic noise modelled as a random variable such that $\in(t) \sim N(0, \sigma^2)$ where σ is the standard deviation.

Note that $S(t)$ can represent any ecosystem response function, depending on the specific ecosystem services being considered. To numerically validate our modelling framework, we select two response functions based on the availability of data. First, when $S(t)$ is defined to measure the climate regulation function in response to restoration, we model $S_{det}(t)$ as aboveground biomass carbon for mangroves, following the approach outlined in existing studies (Lovelock et al., 2021):

$$S_{det}(t) = a * e^{\left(-\frac{k}{t}\right)}, \tag{4}$$

where a approximates the mature above-ground biomass and where k represents the rate at which the biomass increases over time. To a large extent, the aboveground biomass carbon for mangroves reflects the functionality of mangroves that provide climate regulation services, including preventing land erosion from sea level rise and providing barriers to floods. The details of the information used for modelling the aboveground biomass carbon for mangroves are shown in Appendix Table 1. Second, when $S(t)$ was defined to measure the response of biodiversity function to restoration, we followed the methods of (Craft et al., 2003) to use the logarithmic function to model key ecological indicators of nursery habitats, i.e., species richness and invertebrate density, and the age of salt marsh habitats (details of the data are shown in Appendix Figs. 1 and 2). The logarithmic function represents a common pattern in natural systems where initial growth is constrained but becomes more rapid over time.

2.3. The simulation process

Simulations of the above models are executed over a series of time steps *t*, generating values for $S(t)$ at each time step by sampling from the defined distribution. For each time step, the corresponding $V(t)$ is calculated via both the nonlinear model shown in Eq. (1) and the logistic

[2] In the study of Ndebele & Forgie (2017), the coefficient estimates associated with the utility of 20-, 50-, and 100-year conservation attributes showed a typical sigmoidal (S-shaped) curve.

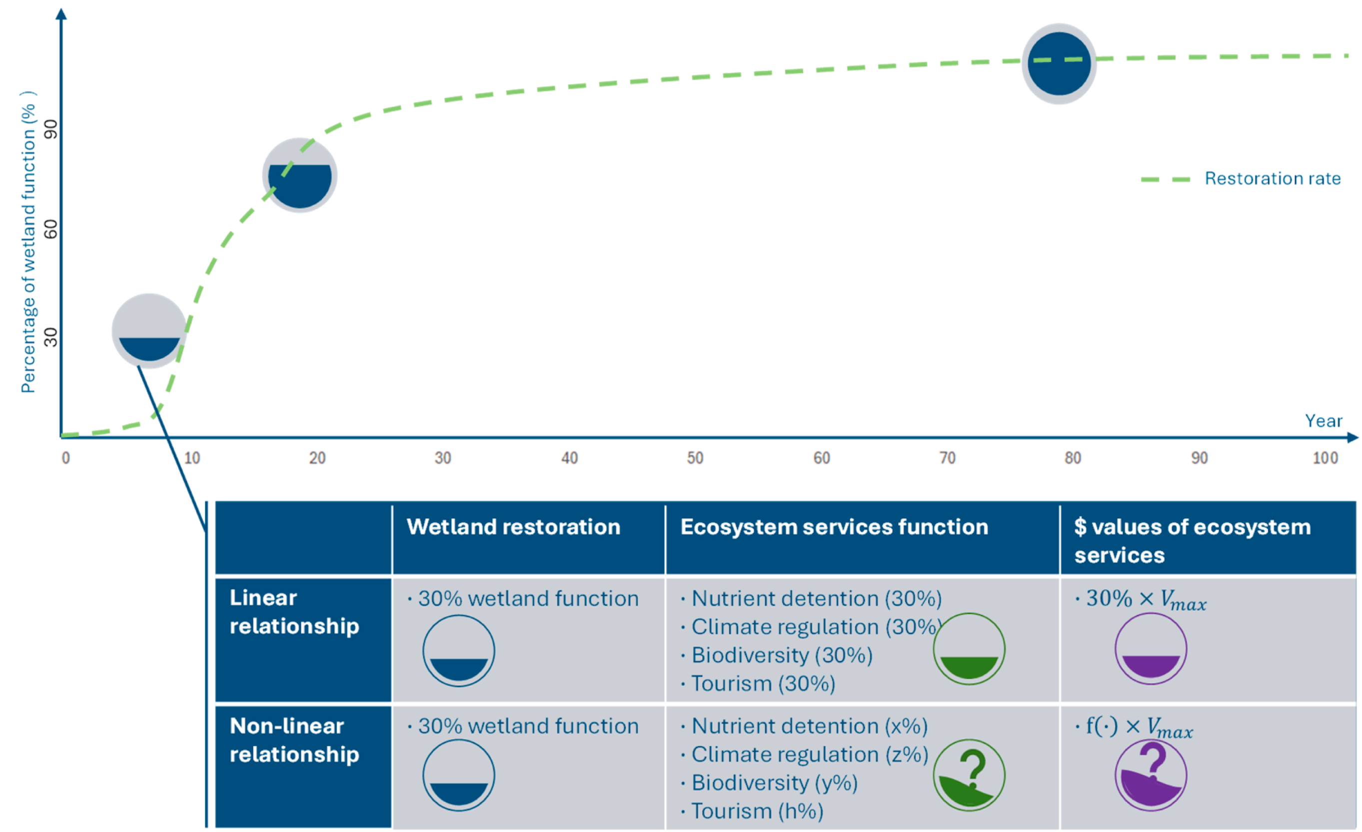


Fig. 2. A hypothetical example showing the nexus of coastal wetland restoration, ecosystem recovery, and valuation of the benefits of ecosystem services. Source: Authors.

models shown in Eq. (2). This process is repeated to observe how the economic value evolves under varying conditions of ecosystem recovery. We performed 1000 iterations per simulation using R Studio 2024.4.2.764. Key parameter settings and their justifications are as follows.

The simulation parameters were selected through a combination of literature review, expert elicitation, and sensitivity testing to ensure ecological plausibility and analytical robustness. For each equation, we specified parameter values that reflected either empirical measurements (e.g., from Lovelock et al., 2021; Craft et al., 2003), expert-informed expectations of wetland recovery behaviour, or standard practice in ecological-economic modelling. Where empirical estimates were unavailable, we performed sensitivity analyses to explore a range of plausible values, thereby assessing the stability of model outputs and the validity of the assumed parameterisation.

In the nonlinear model (Eq. (1)), λ is the key parameter that governs the responsiveness of the economic value to changes in the ecosystem recovery function S(t). A higher λ value would result in a more rapid increase in value as the ecosystem recovers, whereas a lower λ would lead to a more gradual increase. In our simulation, λ was set to a moderate value ($\lambda = 0.05$) based on empirical insights from similar studies (e.g., Liu et al., 2024) and expert elicitation, representing a scenario in which ecosystem services accumulate steadily rather than instantaneously. The simulated benefit trend is expected to be the upper part of Fig. 2 such that the benefits accumulate more rapidly in the early stages and gradually level off as S(t) approaches full ecosystem recovery. V_{max} ($V_{max} = 100$) represents the maximum potential value of ecosystem services. The value is set for scaling purposes to standardise the outputs across different scenarios. The parameter σ ($\sigma = 5$) represents the standard deviation of the Gaussian noise term added to S(t), capturing moderate random fluctuations. Although this assumes independent, normally distributed errors, we note that it may oversimplify potential autocorrelation in ecological recovery processes. It is suggested by wetland ecologist to choose a value of 5 to represent a moderate level of variability, highlighting the uncertainty in the recovery process.

The logistic growth model (Eq. (2)) introduces additional parameters: κ controls the steepness of the curve, and θ, the threshold at which significant benefits begin to accrue. In our simulation, κ ($\kappa = 0.1$) was chosen to produce a typical S-shaped curve, with benefits increasing slowly at first, then more rapidly as the recovery process crosses a critical threshold. The threshold value of θ was informed by expert input and previous studies (e.g., Ndebele and Forgie, 2017), reflecting a realistic recovery threshold for coastal wetlands, beyond which the most substantial benefits of ecosystem services are realised.

In summary, the initial settings of the simulation are based on a few assumptions. Nonlinear model: With moderate responsiveness ($\lambda = 0.05$), carbon sequestration/species richness starts increasing gradually and continues to rise toward the maximum potential value ($V_{max} = 100$), but with some random fluctuations added by the noise ($\sigma = 5$). Logistic Model: the growth in carbon sequestration/species richness begins slowly because of the small steepness ($\kappa = 0.1$) but then increases as time approaches the threshold ($\theta = 20$), eventually levelling off as it nears the maximum value ($V_{max} = 100$).

To assess robustness, we performed sensitivity analyses by varying λ, κ, and θ within plausible ranges derived from the literature and expert feedback. We expect that while the timing and magnitude of benefit accrual varied, the qualitative behaviour of the models remained consistent. Specifically, we conducted one-way sensitivity analyses across $\lambda = 0.01$ to 0.1, $\kappa = 0.01$ to 0.5, and $\theta = 1$ to 50. While the speed and timing of value accumulation shifted, the general concave (nonlinear) and S-curve (logistic) shapes were preserved, supporting the robustness of the model structure. In addition, the nonlinear model (Eq. (1)) was applied across nine countries/regions to ensure a representative analysis of diverse coastal wetland systems. Selection criteria were based on: 1) data availability: only regions with reliable and extensive data on wetland restoration and ecosystem service indicators were included (Lovelock et al., 2021); 2) geophysical diversity: regions were chosen to represent a variety of biophysical characteristics; and 3) policy relevance: selected sites align with regions that are focal points for current climate adaptation and coastal restoration initiatives. This multi-regional approach enhances the robustness and generalisability of our analysis by capturing a range of ecological and economic contexts.

3. Results and discussion

3.1. Simulated benefit trends

The simulation results illustrate the temporal evolution of benefits derived from wetland restoration, accounting for uncertainty in the ecosystem recovery process. Two models were employed to capture this relationship: a nonlinear value function and a logistic growth model. Fig. 3 represents the temporal dynamics of the benefits of climate regulation services derived from mangrove restoration across different countries. Fig. 4 represents the restoration benefits in terms of biodiversity, with a specific focus on species richness and invertebrate density based on saltmarsh restoration data from the North Carolina coast. Each graph illustrates the comparison between the nonlinear model (red solid line) and the logistic model (blue dashed line) in estimating the value of ecosystem services over time for nine different countries or regions. The shaded areas represent the uncertainty or variability in the estimates, with the purple and light blue shading indicating a confidence range.

As shown in the set of graphs in Fig. 3, in most cases, the nonlinear model predicts a rapid increase in climate regulation benefits, which then stabilises over time for all spatial sites (except India). The logistic model, on the other hand, shows a slower initial increase but eventually converges to the same or slightly lower value than the nonlinear model. In addition, the benefit trends for carbon sequestration across various countries and spatial sites reveal notable differences in both the magnitude and growth patterns of values over time.

For most regions (**Vietnam, Indonesia (Bali), French Guiana, Indonesia (Papua), the Philippines and Malaysia)**, both models produce similar trajectories. Early in the restoration process, both models rapidly approach the maximum value with minimal uncertainty, suggesting that restoration benefits stabilize quickly under these conditions (Mitsch and Gosselink, 2015). The shaded areas (uncertainty) are minimal, indicating that both models provide consistent estimates with low variability.

In contrast, significant divergence is observed in **India and Australia.** In India, the nonlinear model reaches the maximum value considerably faster than the logistic model, with a wider uncertainty range. The uncertainty range (shaded area) is wider in this region, indicating a greater variability in the predicted benefits. This divergence may reflect the complex coastal geomorphology, variable restoration practices, and high anthropogenic pressures observed in Indian coastal systems (Nayak, 2017; Ragavan et al., 2019).

In **Australia (Australia (NSW) and Australia (SA)**), similar to India, there is a divergence between the models, but the gap persists throughout the time series. The shaded area indicating variability is larger, particularly in the early stages, indicating greater uncertainty in the estimates. This might reflect that local climatic variability and differing regulatory frameworks may slow the accumulation of restoration benefits (Saintilan et al., 2019; Petter et al., 2013). These factors, in combination with socio-economic conditions, likely contribute to the observed differences in model behaviour.

The observed differences in benefit trends between the nonlinear and logistic models are primarily driven by the mathematical structure of the ecosystem response functions S(t). The nonlinear model, which applies an exponential-like growth function, is more sensitive to small initial gains in ecosystem recovery, resulting in a rapid early accumulation of value. In contrast, the logistic model features a sigmoidal (S-shaped) response, where benefit growth is initially slow, accelerates past a threshold, and eventually levels off. As such, the nonlinear model often predicts earlier benefit realization, especially in the initial years of restoration, while the logistic model captures delayed but more gradual growth processes. These inherent differences in curve shape directly influence the timing, pace, and magnitude of projected benefit accumulation across time.

Beyond the model structure itself, variations in benefit trajectories across countries are governed by both ecological characteristics and institutional factors. Overall, the trends reveal key regional differences in the effectiveness and speed of restoration efforts, as well as in the reliability of predictions. In most regions (e.g., Vietnam and Indonesia), restoration benefits stabilise quickly and show minimal uncertainty, suggesting that both models offer consistent and reliable estimates. However, the divergence in India and Australia underscores potential ecological, socio-economic, and policy-related influences. In India, the rapid early gains captured by the nonlinear model could be driven by localized interventions and variable coastal processes, while the logistic model's more gradual increase may reflect delays due to complex management challenges (Nayak, 2017). In Australia, climatic variability such as fluctuations in rainfall and temperature and stringent environmental regulations may result in a more conservative benefit accumulation, as indicated by the persistent gap between the two models (Saintilan et al., 2019). These interpretations are consistent with studies that emphasize the importance of local context in restoration outcomes (Tilman et al., 2001; Connell and Slatyer, 1977).

As shown in Fig. 4, the simulation results of the two ecological indicators for biodiversity, i.e., species richness and invertebrate density, show similar benefit trends. For species richness, both models produce nearly identical results, with the logistic model being slightly more responsive at the beginning, but both reach the maximum value very quickly. The close alignment of the two models and the narrow-shaded area indicate that the models agree well in estimating species richness benefits, with low variability. Similar to species richness, the models for invertebrate density show a high degree of similarity, with minor differences in the early stages. Both models indicate a rapid increase in biodiversity benefit, quickly reaching the maximum value, with minimal uncertainty.

3.2. Scenario analysis results

3.2.1. The maximum value of ecosystem services

Besides using hypothetical values for the valuation of ecosystem services described above ($V_{max} = 100$), we also referred to the Ecosystem Services Valuation Database (ESVD at https://www.esvd.net/) to draw values for both mangroves and saltmarshes on the ecosystem services of climate regulation and biodiversity (Brander et al., 2024). The benefit value of climate regulation services (mangrove), including erosion prevention and moderation of extreme climate events, is estimated to be USD 5192 per hectare per year (adjusted to 2024 dollars), and the value of biodiversity services (saltmarsh) is estimated to be USD $3448 per hectare per year (adjusted to 2024 dollars).[3] Detailed information about the data collection and valuation is provided in the Appendix. The sets of graphs shown in Figs. 5 and 6 present similar benefit trends as those shown in Figs. 3 and 4; the timings of convergence and stabilisation are also similar. By varying V_{max} to test the sensitivity, the results indicate that our models are robust regardless of the maximum

[3] The original values drawn from the ESVD were 2020 USD and we adjusted to the 2024 USD, considering inflation etc.

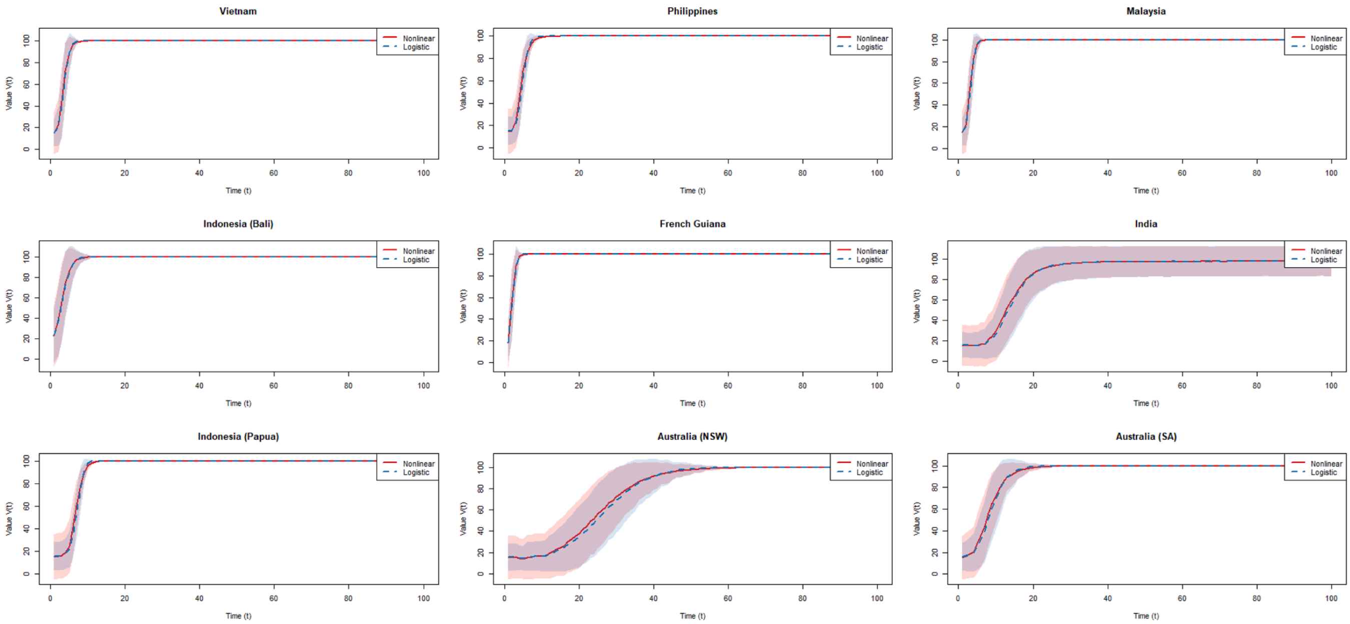


Fig. 3. Simulated benefit trends of restored wetlands as NBSs to provide climate regulation services over time.

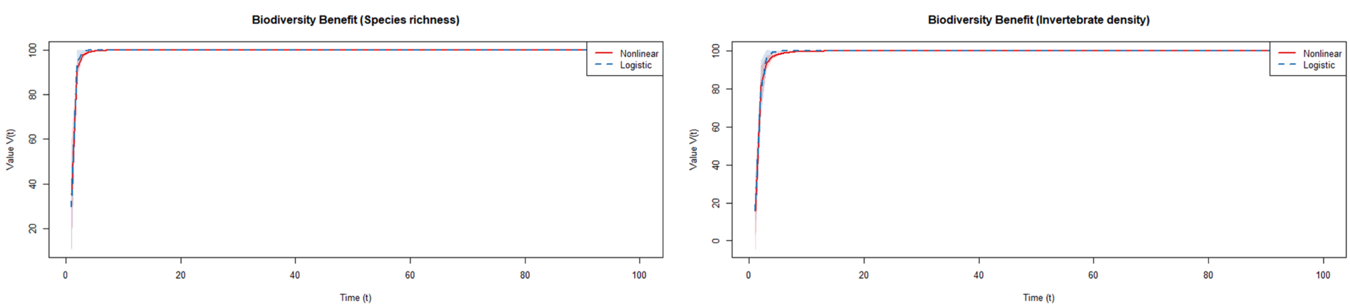


Fig. 4. Simulated benefit trends of restored wetlands as NBSs to provide biodiversity protection (fish habitat nurseries) over time.

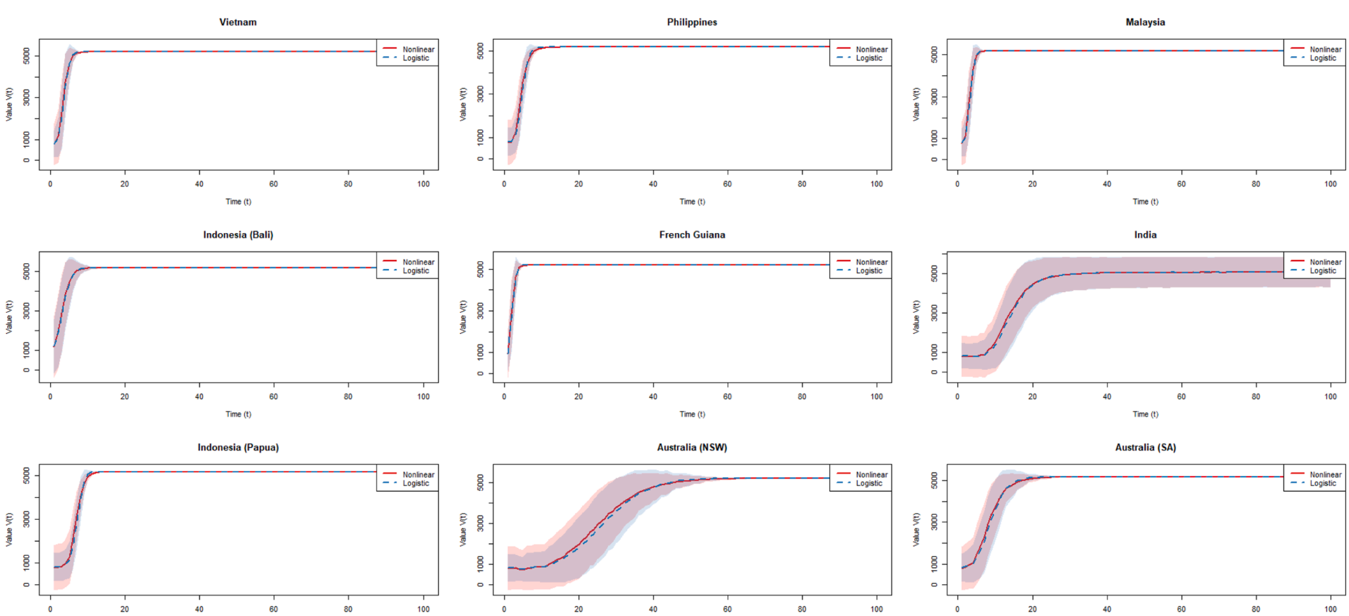


Fig. 5. Simulated benefit trends of restored wetlands as NBSs to provide climate regulation services over time, $V_{max} = \$5,192$ USD per hectare per year.

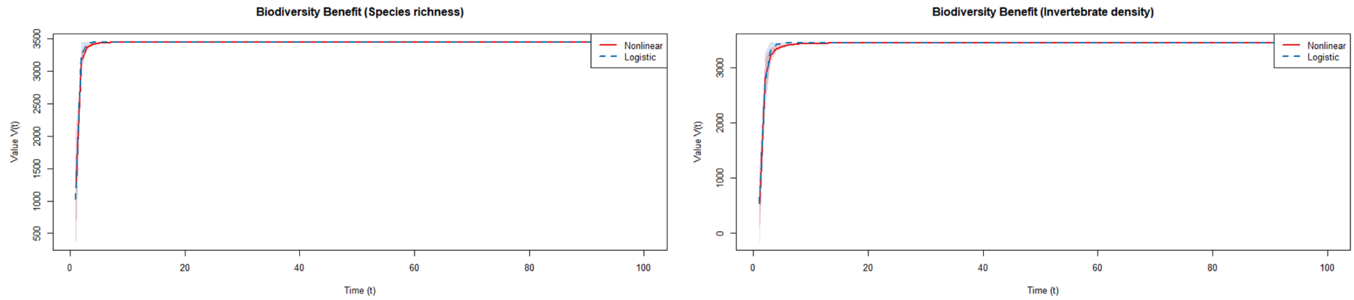


Fig. 6. Simulated benefit trends of restored wetlands as NBSs to provide biodiversity protection (fish habitat nurseries) over time, $V_{max} = \$3,448$ USD per hectare per year.

values of ecosystem services.

Although the scaling of ecosystem service values through V_{max} does not alter the shape of benefit curves, it highlights how regional differences in valuation are governed by both ecological productivity and socio-economic valuation frameworks. For instance, countries with strong climate finance mechanisms or carbon markets (e.g., Indonesia or Vietnam) may assign higher monetary values to services like carbon sequestration, which aligns with the faster benefit accumulation in those models. Conversely, countries with emerging valuation systems or limited data (e.g., Papua, India) may reflect more conservative estimations.

3.2.2. Varying responsiveness (λ), steepness (κ), and threshold (θ)

The scenario analysis results, presented in Figs. 7 and 8, illustrate the sensitivity of the models to key parameters—namely, the responsiveness (λ), steepness (κ), and threshold (θ). We changed the values of the key parameters that are relevant to the responsiveness of benefit values to ecosystem response functions: examining how the simulation results respond to high and low responsiveness and steepness in the nonlinear model and logistic model, respectively. As shown in Fig. 7a, both models align closely, indicating that under these conditions (i.e., high responsiveness and steepness), the process of climate regulation services and its economic valuation are well captured by either model with minimal uncertainty. In a low-responsiveness and steepness scenario (Fig. 7b), the models diverge slightly, with the nonlinear model showing a more cautious growth trajectory, leading to discrepancies in the timing and magnitude of the value stabilisation. The increased uncertainty reflects a more complex and less predictable recovery process. These results suggest that when the valuing system is highly responsive to changes, simpler models such as the logistic model can perform just as well as more complex nonlinear models. However, under less responsive conditions, the nonlinear model might offer a more realistic representation of gradual improvements, although at the cost of increased uncertainty and complexity. We constructed the same scenario analysis for the ecosystem response function of biodiversity protection, and the results (in Fig. 8) show trends like those in Fig. 7, indicating that, regardless of different types of ecosystem responses, our modelling framework provides robust and consistent simulation results for the evaluation of ecosystem services.

Notably, the nonlinear model shows greater responsiveness at low values of the responsiveness parameter (e.g., low λ), suggesting that even small changes in the ecosystem's recovery rate or initial conditions may lead to relatively large changes in the estimated value of ecosystem services early on (shown in Fig. 7b and 8b). This implies that in the early stages of restoration, the model is more sensitive to small improvements in wetland functions, making it highly reactive to early changes. The nonlinear model better reflects ecosystem recovery processes that respond rapidly at the beginning but may not sustain high growth over time. Early-stage sensitivity in restoration processes has been noted by studies emphasizing rapid recovery in certain ecological functions. For instance, Zedler and Callaway (1999) discuss how wetlands and other ecosystems often exhibit a phase of rapid functional recovery, particularly in processes like sediment trapping and nutrient cycling, during early restoration. In these early stages, even small ecological improvements can yield substantial benefits, as seen in the rapid increase predicted by the nonlinear model. This model effectively captures the dynamic, nonlinear nature of such processes, where early gains can be disproportionately large compared to later stages, a feature noted by Mitsch and Gosselink (2015) in their comprehensive work on wetland ecosystems. Moreover, the notion that early interventions yield disproportionately higher economic returns has been discussed in the context of cost-benefit analysis of restoration projects. For example, Hanley et al. (2012) emphasize that prioritizing early investments in ecosystem services can result in greater initial returns, particularly for services like carbon sequestration and water quality improvement. This supports the idea that restoration projects targeting wetland services may be economically more viable in their early phases, as indicated by the rapid gains seen in the nonlinear model.

In contrast, the logistic model's slower initial growth due to its sigmoid shape may indicate that improvements in ecosystem services are more gradual and less responsive to early changes. The logistic model typically has a lag phase before reaching a point of rapid growth, which suggests that it is less sensitive to early restoration efforts but more stable over time. Hence, the logistic model is more appropriate for processes with gradual accumulation, such as biodiversity restoration or organic detritus buildup, where slow initial growth is followed by a phase of rapid improvement before levelling off. Dobson et al. (1997) and Tilman et al. (2001) highlight that biodiversity recovery is often a slow process, characterized by initial lags before accelerating as species richness and ecosystem complexity increase. This is particularly relevant for services tied to long-term ecological processes, such as soil organic matter accumulation and the rebuilding of ecosystem structure, which accumulate benefits gradually but ultimately stabilise at higher levels, as observed in the logistic model. This reflects the gradual development of ecosystems, as described by (Connell and Slatyer, 1977) in their work on succession models. For long-term projects focused on services that require prolonged ecological maturation (e.g., biodiversity recovery), this model demonstrates the importance of sustained investment, as it accounts for delayed but significant improvements. This finding is consistent with research by Bullock et al. (2011) who suggests that long-term funding is crucial for ecosystem services like biodiversity recovery that may show limited progress in the early years but provide

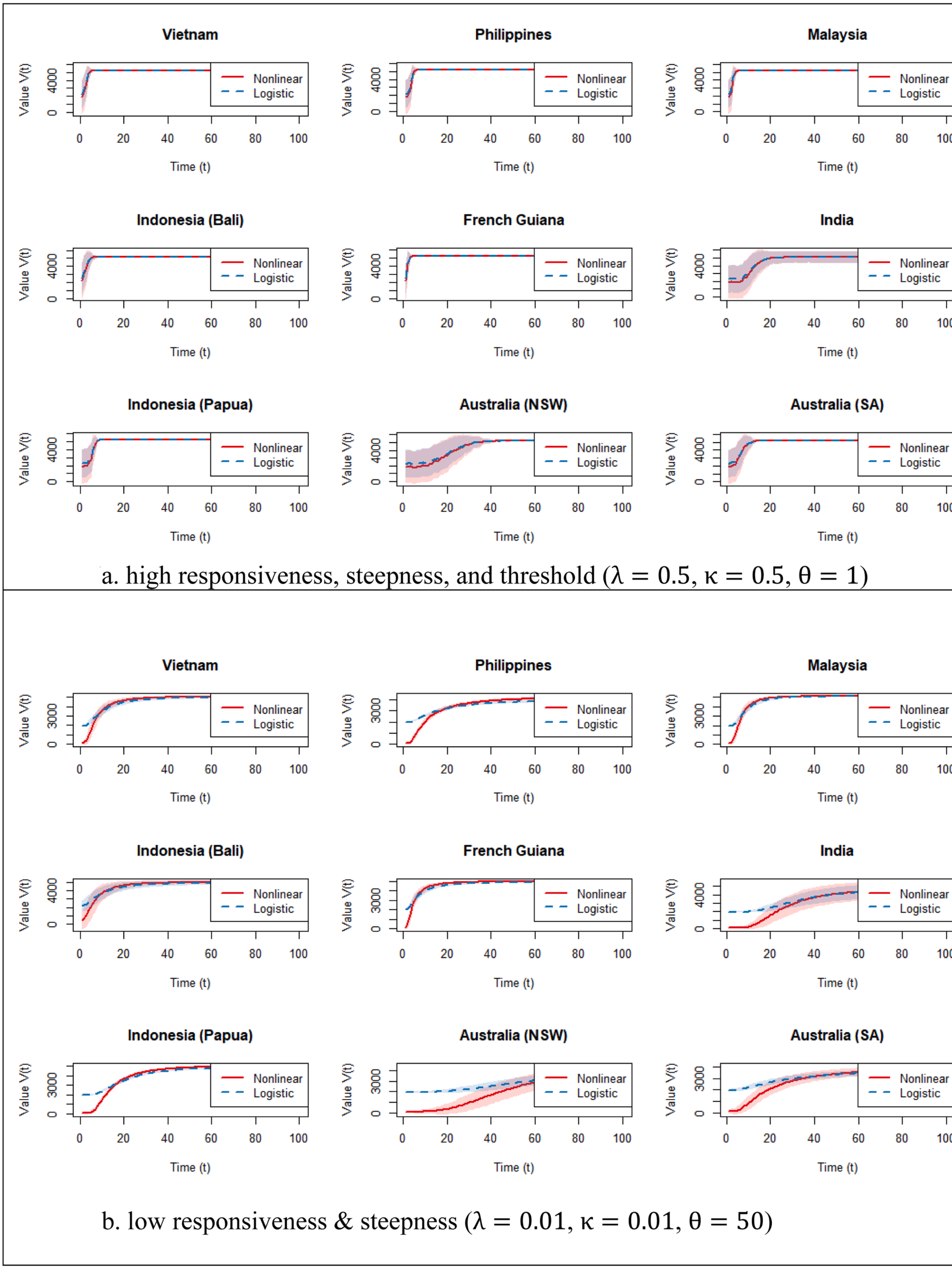
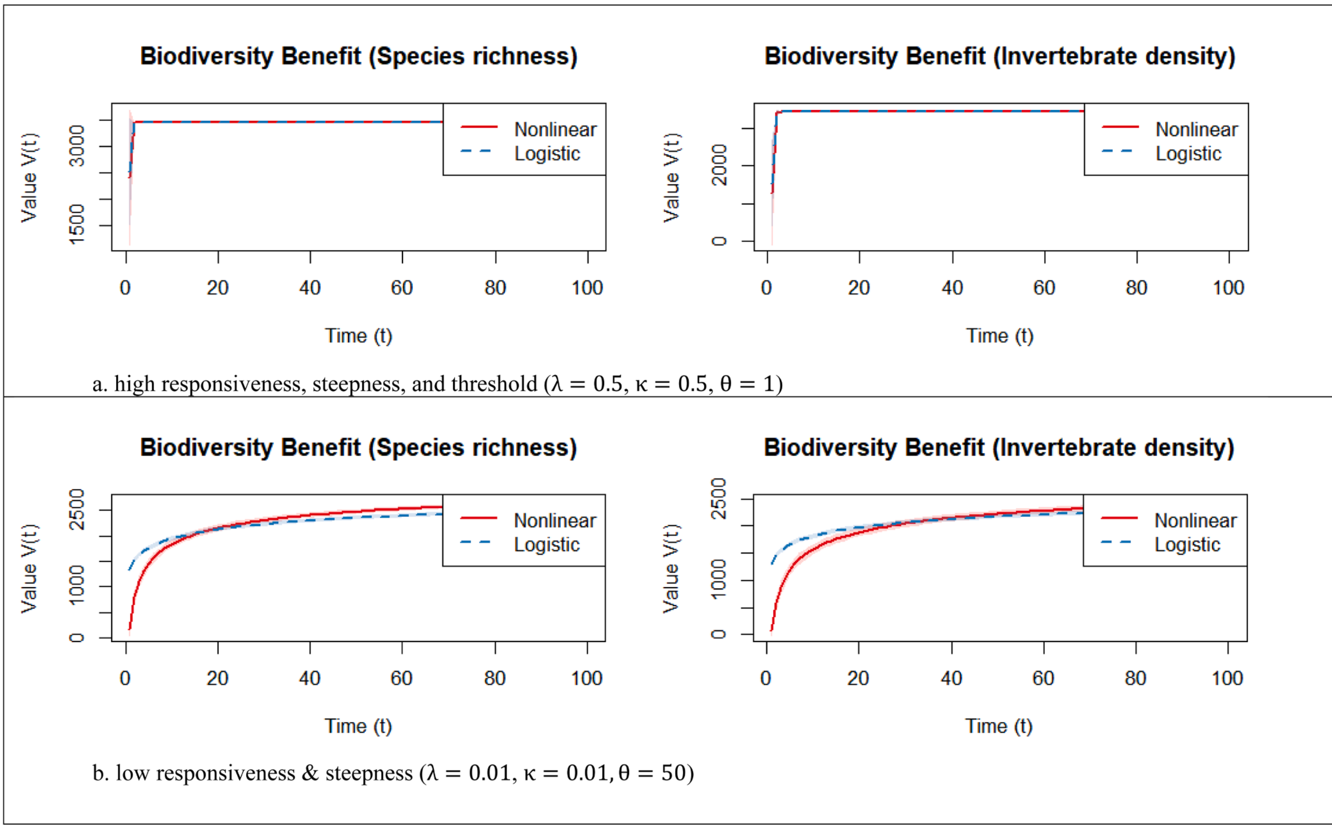


Fig. 7. Simulated benefit trends of restored wetlands as NBSs to provide climate regulation services over time, with high and low responsiveness and steepness, V_{max} = \$5,192 per hectare per year.



a. high responsiveness, steepness, and threshold (λ = 0.5, κ = 0.5, θ = 1)

b. low responsiveness & steepness (λ = 0.01, κ = 0.01, θ = 50)

Fig. 8. Simulated benefit trends of restored wetlands as NBSs to provide biodiversity protection (fish habitat nurseries) over time, with high responsiveness and steepness, $V_{max} = \$3,448$ per hectare per year.

substantial long-term benefits once key thresholds are reached.

The differences in benefit growth trajectories under varying parameter values are shaped not only by mathematical structure but also by ecological realities. For instance, high responsiveness (λ) or steepness (κ) may realistically represent regions with strong institutional support, rapid ecological recovery (e.g., sediment accretion, species recolonization), and active stakeholder engagement. In contrast, low λ or high thresholds (θ) may be more representative of areas where delayed benefits result from degraded baseline conditions, invasive species, or insufficient long-term funding. This sensitivity analysis demonstrates that model outputs are contingent upon a combination of intrinsic ecological response patterns and extrinsic management, economic, and governance conditions. Therefore, model selection and interpretation should always consider site-specific enabling or constraining factors.

The results and findings of the scenario analysis have significant implications for real-world investment strategies. For example, regions or projects demonstrating rapid early gains may attract short-term investments, whereas areas with gradual benefit accumulation will likely require sustained funding. Policymakers and investors, particularly those involved in global climate finance mechanisms such as the Green Climate Fund or blue carbon markets, can use these insights to better align funding schedules with the dynamic nature of ecosystem recovery, ensuring that investments are both timely and effective. Recent policy reports (e.g., United Nations Environment Programme, 2021; World Bank, 2022) emphasise the need for adaptive investment frameworks that accommodate the inherent uncertainty and variability in ecosystem service recovery. Moreover, research by Hanley et al. (2012) supports the view that prioritising early interventions can yield substantial short-term returns, while maintaining long-term funding commitments is crucial for achieving enduring ecological and economic benefits. These adaptive strategies are essential for maximising both ecosystem resilience and the economic returns on restoration investments.

4. Conclusion

This study highlights the significant economic benefits of wetland restoration as a nature-based solution for addressing critical environmental challenges such as climate regulation and biodiversity protection. Using both nonlinear and logistic models, our analysis captures the temporal dynamics of ecosystem service valuation across diverse geographical regions, providing valuable insights for policy formulation and investment strategies. The nonlinear model's prediction of rapid early-stage benefits suggests that restoration efforts can yield immediate returns, particularly in regions where short-term economic gains are a priority. This responsiveness to initial restoration efforts is critical for decision-makers looking to justify upfront investments in ecosystem recovery. Conversely, the logistic model's slower, more gradual growth trajectory reflects the need for sustained, long-term strategies, especially in regions where ecosystem recovery and associated services take time to fully materialise. Our scenario analysis demonstrates that both models can provide robust and reliable estimates under varying conditions of ecosystem responsiveness and recovery steepness, with minimal uncertainty in high-responsiveness scenarios. However, the divergence between the two models in low-responsiveness scenarios underscores the complexity of accurately valuing ecosystem services in regions with unpredictable recovery patterns, such as India and Australia. This finding reinforces the importance of tailoring restoration policies and investments to local ecological and socio-economic conditions. The integration of maximum ecosystem service values from the ESVD further supports the validity of the models across different ecosystems.

As policymakers and environmental managers seek to mobilise climate finance and implement effective strategies, several key implications emerge from our analysis results. First, policymakers need to integrate dynamic valuations into policy frameworks, including existing cost-benefit analysis (CBA) protocols. Traditional cost-benefit analyses and valuation methods often overlook the temporal dynamics of ecosystem recovery and the delayed realisation of economic benefits, leading to inaccurate estimations of the benefits of wetland restoration. The integration of dynamic valuation requires enhanced data collection, rigorous monitoring strategies to adjust policies as ecosystems evolve over time. By doing so, funding and resource allocation can be better aligned with the actual timelines of ecosystem recovery, ensuring that investments in wetland restoration yield maximum long-term returns. In addition, given the variable rates at which different ecosystem services recover, adaptive management is essential. Policymakers should promote flexible, iterative management frameworks, such as dynamic adaptive pathway planning (DAPP), which allow for adjustments as ecosystems evolve. Monitoring and evaluation protocols should be established to track the recovery of key indicators, such as carbon sequestration, biodiversity, and nutrient retention, enabling timely interventions to optimise restoration outcomes along adaptive pathways. Second, while rapid early-stage benefits may attract short-term investments, our findings suggest that sustained, long-term funding is crucial for areas with gradual benefit accumulation. The long-term economic benefits of wetland restoration demonstrate the need for more targeted and informed investments in NBSs, ensuring that resources are allocated efficiently. Policymakers should advocate for the inclusion of precise, data-driven approaches in national and international climate finance mechanisms, such as the Green Climate Fund and carbon offset markets. Emphasising the co-benefits of NBSs, including disaster risk reduction, biodiversity conservation, and local livelihoods, can attract diverse funding streams and promote broader adoption. Additionally, the economic value of nonmarket benefits, such as flood protection, carbon storage, and habitat provision, needs to be recognised and integrated into policy decisions. While challenging, efforts to monetise these benefits can support more informed decision-making and justify public and private investments in restoration projects. The development of standardised methodologies for nonmarket valuation that consider the dynamics of ecosystem recovery will increase the robustness of economic assessments. Finally, the operationalisation of our models within policy frameworks will depend on the development of standardised methodologies for nonmarket valuation, coupled with robust monitoring and evaluation systems. Policymakers should support research initiatives that explore the thresholds and tipping points in wetland restoration, as well as the socioeconomic impacts of these projects. Building a robust evidence base will improve the effectiveness of policies and management practices, leading to more resilient and sustainable coastal ecosystems. Policymakers should also consider adaptive management frameworks that recognise these temporal variations and include mechanisms to mitigate potential unintended consequences, such as ecological oversimplification or trade-offs between different ecosystem services.

Despite these insights, the study has certain limitations. First, the models are based on simplified assumptions about ecosystem recovery processes and may not fully capture the complex, context-dependent interactions between ecological factors and socioeconomic conditions. Additionally, the temporal resolution of the simulation may overlook short-term fluctuations or delays in ecosystem responses that could affect the accuracy of service valuations. Data limitations also constrain the generalisability of our results, particularly in regions with insufficient or inconsistent data on wetland ecosystems. Although our simulations incorporate real-world ecological data from studies such as Lovelock et al. (2021) and Craft et al. (2003), the valuation inputs used were based on average figures from the ESVD. While these data provide a strong empirical foundation, future studies could test the modelling framework using first-hand valuation data from a broader range of regions. Such empirical validation will further calibrate our predictions and ensure even greater robustness and policy relevance of our findings. Also, future research could focus on refining these models by incorporating additional ecosystem services, such as nutrient cycling and flood protection, as well as ecosystem response functions and data.

CRediT authorship contribution statement

Wei Yang: Writing – review & editing, Writing – original draft, Visualization, Validation, Resources, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Chris C. Tanner:** Writing – review & editing, Methodology, Investigation, Conceptualization. **Paula Holland:** Writing – review & editing, Project administration, Investigation, Funding acquisition. **Zoe Qu:** Investigation, Data curation.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.envc.2025.101182.

Appendix A. Tables and Figures

Appendix Table 1

. Ecosystem response functions (aboveground biomass carbon) of mangroves across different countries.

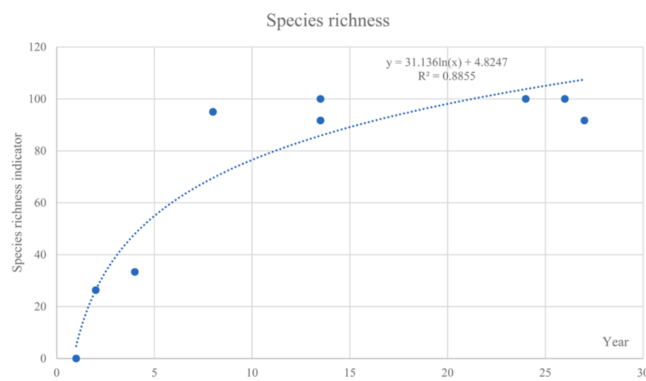
Location[a]	Species	Activity	a[b]	k	R2	Reference
Australia (NSW)	*Avicennia marina*	Natural regeneration	405.7 ± 27.7	97.8 ± 25.6	0.696	Unpublished
Australia (SA)	*Avicennia marina*	Natural regeneration	171.5 ± 24.9	23.9 ± 6.7	0.383	Unpublished
French Guiana	*Avicennia germinans*	Natural regeneration	377.7 ± 27.7	7.61 ± 1.98	0.484	Walcker et al. 2018
India	*Avicennia marina*	Planting mud flats	621.5 ± 308a	58.1 ± 11.9	0.721	Kandasamy et al. 2021
Indonesia (Bali)	*Rhizophora apiculata*	Planted shrimp ponds	465.7 ± 111.6	11.99 ± 6.40	0.642	Sidik et al. 2019
Indonesia (Papua)	*Rhizophora apiculata*	Natural regeneration	464.6 ± 26.2	26.4 ± 1.74	0.99	Sillanpaa et al. 2016
Malaysia	*Rhizophora apiculata*	Plantation	604.4 ± 53.5	13.12 ± 2.36	0.817	Adame et al. 2018
Philippines	*Rhizophora apiculata*	Plantation	193.3 ± 11.9	13.49 ± 1.33	0.895	Salmo et al. 2013
Vietnam	*Rhizophora apiculata*	Plantation	467.6 ± 61.5	13.69 ± 3.09	0.632	Phan et al. 2019

Note:

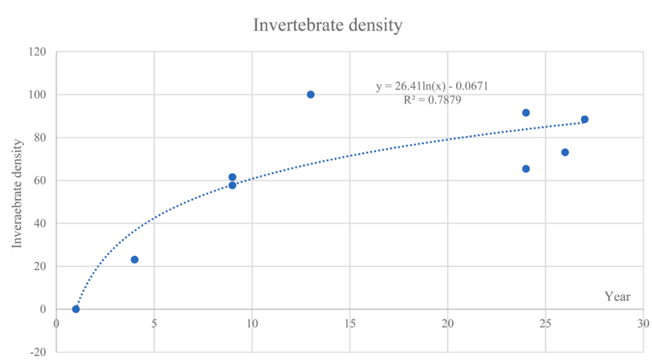
[a] The countries were ordered alphabetically by country name

[b] the parameters of a and k reflect site-specific ecological and environmental conditions, including differences in species growth rates, plantation history, soil properties, and climatic factors. As noted in Lovelock et al. (2021), the variation across regions and planting contexts is expected and was statistically analysed in their original study.

Source: Lovelock et al. (2021).



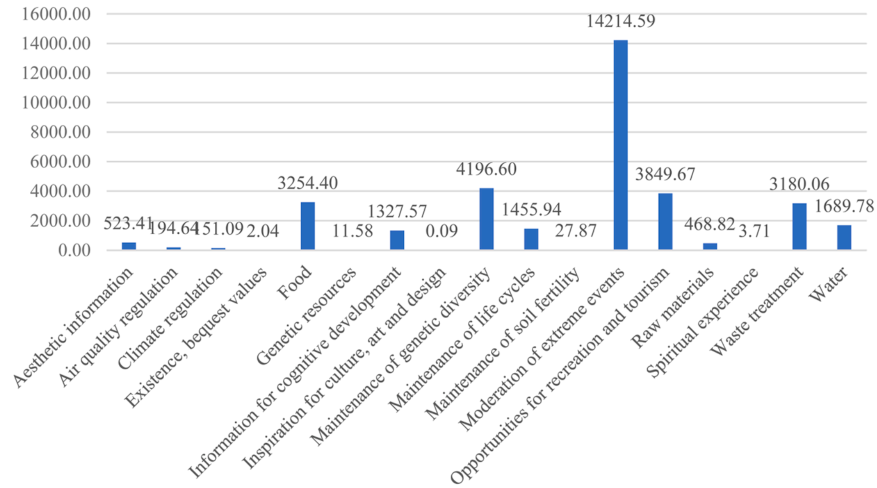
Appendix Fig. 1. The fitting of species richness over time (the age of the restored wetland - saltmarshes).



Appendix Fig. 2. The fitting of invertebrate density over time (the age of the restored wetland - saltmarshes).

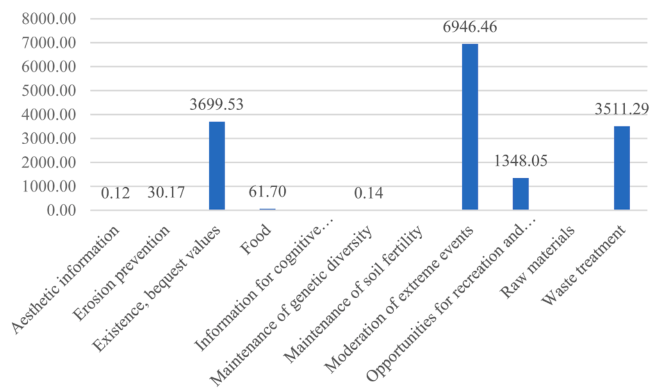
Appendix B. nonmarket values of ecosystem services for mangroves and saltmarshes

This analysis aims to collect economic evaluation data on the value of saltmarshes from various perspectives, such as coastal restoration and evaluation of coastal or marine ecosystems. The data used in the analysis were sourced from the Ecosystem Services Valuation Database (ESVD), which was developed with the long-term goal of providing robust and easily accessible information on the economic benefits of ecosystems and biodiversity and the costs of their loss to support decision making regarding nature conservation, ecosystem restoration and sustainable land management. It is the largest database (now 10,889 value records) with monetary values mapped across different ecosystem services and geographic regions. The search strategy is as follows. First, we used the search filter to include only biomes of marine and coastal systems. Second, we use the keyword “salt marsh” to narrow down the focus to studies on/relevant to salt marshes. This produced 438 value records. We further filtered the ecosystems to include only the categories of “coastal salt marshes and reedbeds”, producing a final list of 164 observations. A further breakdown of the values by ecosystem services is shown in the following figure. We used the average values for maintenance of generic diversity and maintenance of the life cycle (including providing refugia for migratory and resident species, such as fish). bird. Mammal and nursery services) to represent the value of biodiversity protection and adjust it to 2024 USD.



Appendix Fig. 3. Benefit values across different types of ecosystem services of saltmarsh wetlands.

We conducted the same search and analysis process for nonmarket values of ecosystem services for mangroves, which provided a list of 58 value estimates, and the values were further categorised into different ecosystem services (shown below). We used the values for erosion prevention and moderation of extreme events and adjusted them to 2024 USD.



Appendix Fig. 4. Benefit values across different types of ecosystem services of mangrove wetlands.

Appendix Fig. 1, Appendix Fig. 2, Appendix Fig. 3, Appendix Fig. 4

Data availability

data and codes are shared in the supplementary

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