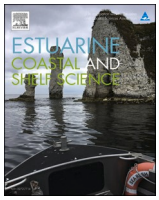
Estuarine, Coastal and Shelf Science 344 (2027) 110323



Contents lists available at ScienceDirect

Estuarine, Coastal and Shelf Science

journal homepage: www.elsevier.com/locate/ecss



Standardising tidal position of coastal wetlands across New Zealand estuaries: A national synthesis

P.J. Stewart-Sinclair [a,*], O. Lam-Gordillo [a], G. Reeve [a], R. Bulmer [b], I. Zabarte-Maeztu [a], A. Swales [a]

[a] Earth Sciences New Zealand, Hamilton, New Zealand
[b] Tidal Research Ltd, Auckland, New Zealand

ARTICLE INFO

Keywords:
Tidal frame
Coastal wetlands
Estuarine zonation
Mangroves
Saltmarsh
Seagrass

ABSTRACT

Coastal wetland distribution in estuaries is strongly controlled by position within the tidal frame, yet comparisons among systems are often confounded by large regional differences. Standardised metrics of tidal position are therefore needed to compare habitat zonation across estuaries with contrasting hydrodynamic settings. Here we apply the Standard Tidal Position Index (STPI) to quantify and compare the tidal position of coastal wetland habitats across Aotearoa New Zealand. Distributions of seagrass, mangroves, and saltmarsh were analysed for 47 estuaries spanning a range of geomorphic types. Habitat elevation was derived from 1-m resolution coastal LiDAR and referenced to estuary-specific tidal ranges from a calibrated TELEMAC-2D hydrodynamic model. Elevation boundaries between habitats were estimated using weighted ordinal regression and normalised within the local tidal frame using STPI.

Across all estuaries, STPI revealed consistent vertical structuring of habitats despite large variation in tidal range, estuary geomorphology, and absolute elevation. National median STPI values placed seagrass below mean tide level (upper boundary ≈ 0.06), mangroves between ≈ 0.06 and 0.37 where present, and saltmarsh above ≈ 0.37. Saltmarsh therefore extended lower in the tidal frame than commonly inferred from neap or spring tidal benchmarks. Standardising elevation within the tidal frame substantially reduced apparent variability in habitat position among estuaries and clarified differences in intertidal zonation between systems with and without mangroves.

These results demonstrate that STPI provides a robust and transferable framework for comparing coastal wetland zonation across estuaries with differing tidal regimes, with direct applications for restoration planning and sea-level rise assessment.

1. Introduction

Coastal wetlands are among the most dynamic and productive ecosystems on Earth, occupying the interface between terrestrial and marine environments where tidal inundation shapes strong ecological gradients and biological communities (Bulmer et al., 2024; He et al., 2025; Mcleod et al., 2011). These ecosystems, including saltmarshes, mangroves, and seagrass, support high primary productivity, provide critical habitat for fish, invertebrates, and migratory birds, and regulate a range of biogeochemical processes including nutrient cycling, carbon storage, and greenhouse gas exchange (Adame et al., 2024; Rogers et al., 2019; Wang et al., 2019). As a result, coastal wetlands are increasingly recognised for their role in climate mitigation through “blue carbon” sequestration and for the ecosystem services they provide to coastal societies (see Figs. 1 and 2).

In Aotearoa New Zealand (NZ), coastal wetlands occur within a diverse mosaic of estuarine and shoreland environments around the country's extensive coastline. These wetlands form part of a broader wetland landscape that includes bogs, fens, swamps, and marshes, with saltmarsh representing one of the principal coastal wetland types shaped by tidal hydrology (Bulmer et al., 2024; Johnson and Gerbeaux, 2004; Stewart-Sinclair et al., 2024). Despite their ecological and cultural importance, New Zealand's wetlands have experienced substantial historical loss. It is estimated that approximately 90–95% of wetlands have been lost since pre-European settlement, primarily through drainage for agriculture, urban development, and other land-use changes (Burge,

* Corresponding author.
E-mail address: phoebe.stewartsinclair@jcu.edu.au (P.J. Stewart-Sinclair).

https://doi.org/10.1016/j.ecss.2026.110323
Received 21 April 2026; Received in revised form 28 July 2026; Accepted 9 October 2026
Available online 10 October 2026
0272-7714/© 2026 The Authors. Published by Elsevier Ltd. This is an open access article under the CC BY license (http://creativecommons.org/licenses/by/4.0/).

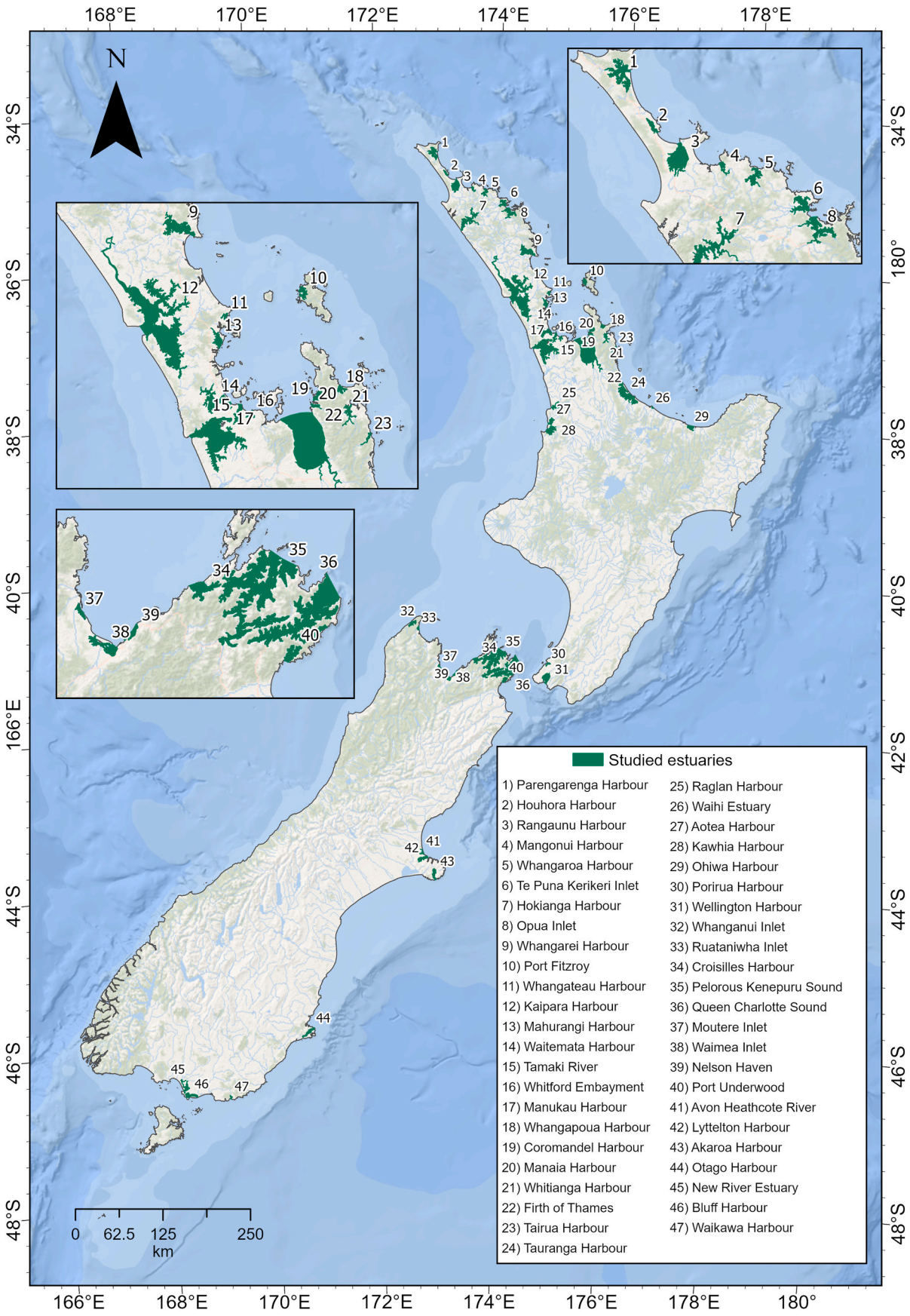


Fig. 1. Map showing the 47 studied estuaries across New Zealand.

2024; Dymond et al., 2021; Swales et al., 2020). Although many of these losses occurred in freshwater wetlands such as fens and swamps, coastal wetlands continue to face pressures from land development, sedimentation, and climate-driven environmental change (Bulmer et al., 2024; Stewart-Sinclair et al., 2024).

In contrast to the widespread loss of many wetland types, mangrove forests (*Avicennia marina*) have expanded rapidly in northern New Zealand estuaries over the past century. This expansion has been linked to increased sediment delivery resulting from historical catchment deforestation and land-use change, which accelerated estuary infilling and created new intertidal habitat suitable for mangrove colonisation (Morrisey et al., 2010; Swales et al., 2015). The expansion of mangroves has generated considerable debate in coastal communities and among management agencies, with some advocating for mangrove conservation while others support active management or removal in areas where mangrove encroachment is perceived to reduce amenity or alter ecosystem function (Harty, 2009; Lundquist et al., 2013). Understanding how coastal wetland habitats are distributed within the tidal frame is therefore essential for interpreting these ecological dynamics and informing management decisions.

The vertical position of wetlands within the tidal frame is a fundamental control on ecological structure and function in estuarine systems. Tidal inundation frequency and duration, collectively referred to as hydroperiod, strongly influence soil redox conditions, salinity gradients, sediment deposition, and vegetation zonation across the intertidal zone (Holmquist and Windham-Myers, 2022; Sulman et al., 2024; Swales et al., 2021). As a result, coastal wetland habitats typically occupy distinct vertical zones relative to tidal inundation, with seagrass generally occupying lower intertidal and shallow subtidal environments, mangroves often occupying mid-intertidal elevations where present, and saltmarsh occurring toward the upper intertidal margins.

Comparing wetland elevation across estuaries, however, is complicated by large regional differences in tidal range. Wetlands located at the same absolute elevation relative to mean sea level can experience very different hydroperiods depending on local tidal amplitude. For example, an elevation that lies near mean tide level in a macrotidal estuary may correspond to a much higher relative tidal position in a microtidal setting. This variability, both among and within estuaries, complicates efforts to compare wetland elevation and inundation regimes across regions using absolute elevation metrics alone. In New Zealand, tidal ranges vary considerably among estuaries, with mesotidal conditions (2-4 m) typical of many systems but with regional variation driven by coastal geometry, harbour resonance, and hydrodynamic processes (LINZ, 2025b; Stevens and Chiswell, 2009). Consequently, comparing wetland elevation across estuaries using fixed vertical datums such as mean sea level can obscure underlying ecological patterns.

The Standard Tidal Position Index (STPI) has been proposed as a relative elevation metric that standardises wetland position within the

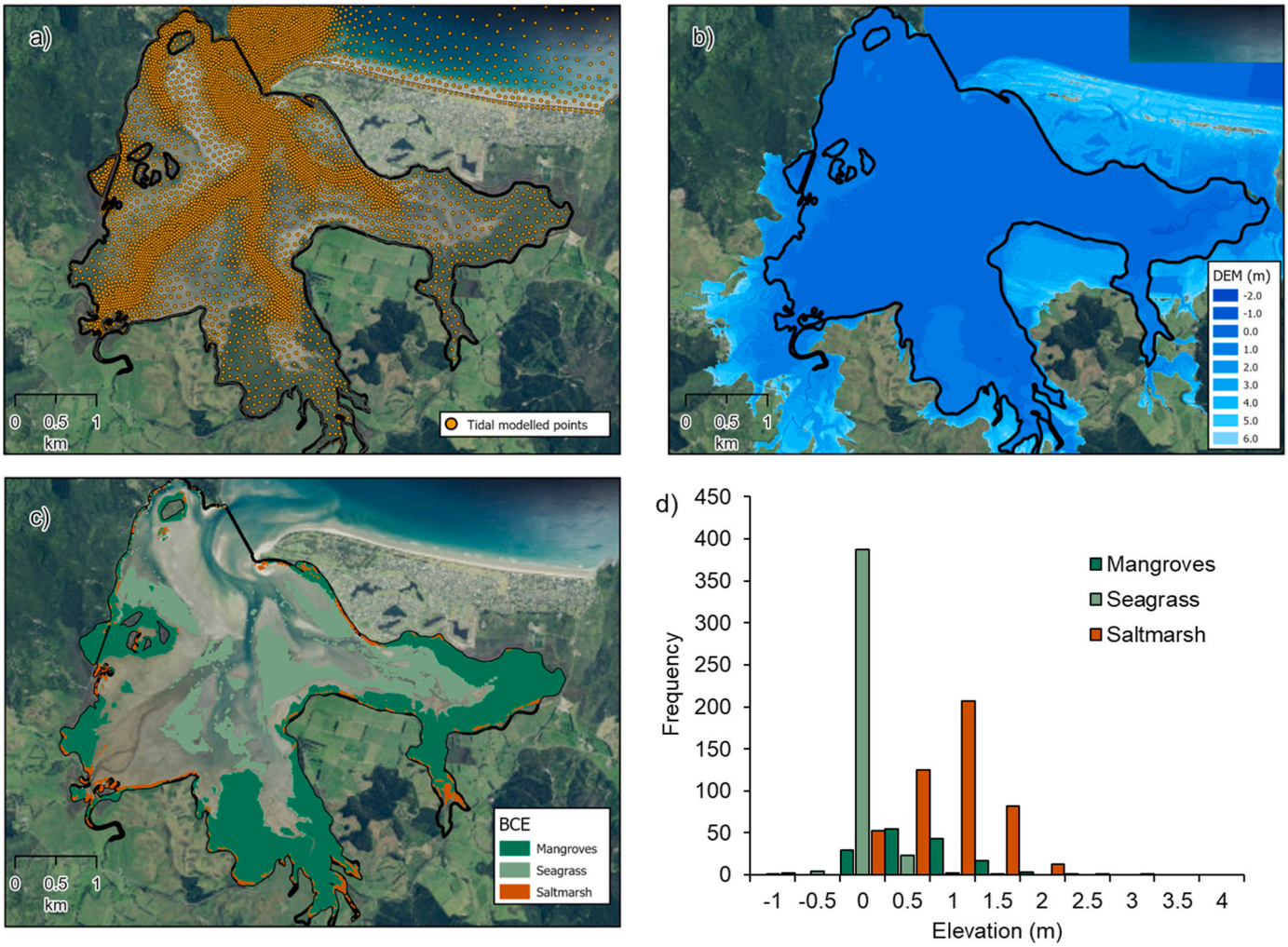


Fig. 2. Summary graphic showing the steps to produce the elevation frequency distribution for each habitat within each estuary across New Zealand. a) Tidal modelled points (HAT and LAT), b) Digital Elevation Model (DEM), c) Blue Carbon Ecosystems (BCE), and d) the elevation frequency obtained from our estuary example Whangapoua harbour.

tidal frame, allowing comparisons among systems with different tidal ranges (Holmquist and Windham-Myers, 2022; Lal et al., 2024; Lovelock et al., 2023; Neville et al., 2024). STPI scales elevation relative to local tidal datums so that habitat position is expressed as a proportion of the local tidal range rather than as an absolute elevation. Conceptually, STPI values of −1 correspond to the lower tidal limit (e.g., lowest astronomical tide), values of +1 correspond to the upper tidal limit (e.g., highest astronomical tide), and values near zero represent mid-tide conditions. Because inundation stress is experienced relative to position within the tidal frame, this normalised metric provides an ecologically meaningful basis for comparing wetland elevation and habitat zonation across estuaries with differing tidal regimes.

Understanding wetland position within the tidal frame is also critical for predicting responses to sea-level rise. Coastal wetlands persist through a balance between vertical accretion, driven by sediment deposition and organic matter accumulation, and landward migration as sea levels rise. If sea-level rise outpaces vertical accretion, wetlands may become increasingly inundated and eventually lose vegetation, a process commonly referred to as “drowning.” Conversely, if landward migration is prevented by topography or infrastructure, wetlands may experience “coastal squeeze,” resulting in habitat loss even if vertical elevation is maintained (Lal et al., 2024). In sediment-rich estuaries, however, high sediment supply can promote vertical accretion or seaward progradation of intertidal flats, potentially enabling wetland persistence under moderate sea-level rise (Swales et al., 2015). Quantifying wetland position within the tidal frame is therefore essential for evaluating vulnerability to future environmental change.

Despite growing international application of STPI as a tool for analysing coastal wetland elevation and zonation, its use has not yet been applied systematically at a national scale in New Zealand. The relative tidal position of major coastal wetland habitats across the country's diverse estuarine systems therefore remains poorly quantified. Addressing this gap is important both for improving ecological understanding of wetland zonation and for supporting efforts to assess vulnerability to sea-level rise.

In this study, we apply the Standard Tidal Position Index to quantify the tidal position of coastal wetland habitats across New Zealand estuaries. Specifically, we (1) estimate the tidal elevation ranges of seagrass, mangrove, and saltmarsh habitats across 47 estuaries using high-resolution elevation and tidal datasets; (2) evaluate how wetland zonation varies among estuaries with different tidal and geomorphic characteristics; and (3) assess the utility of STPI as a framework for comparing coastal wetland elevation and anticipating potential habitat responses to sea-level rise.

2. Methods

2.1. Study areas

To generate robust estimates of the tidal position of coastal wetlands in New Zealand, we selected the 50 largest estuaries for which hydrodynamic tide models were available, thereby providing estuary-specific tidal water-level information required for the Standard Tidal Position Index (STPI) analysis. Three estuaries were excluded due to insufficient data availability: Patterson Inlet/Big Glory Bay (habitat elevation data were entirely invalid), Blueskin Bay (tidal statistics from the TELEMAC model were anomalous and could not be interpreted with confidence), and Jacobs River (only saltmarsh habitat was present, so no elevation transition boundary could be estimated). This resulted in a final dataset of 47 estuaries, of which 31 are located in the North Island and 16 in the South Island. This resulted in a final dataset of 47 estuaries, of which 31 are located in the North Island and 16 in the South Island.

The selected estuaries encompass a range of geomorphic types, including tidal lagoons (n = 9), shallow drowned valleys (n = 24), deep drowned valleys (n = 13), and one coastal embayment. Estuary size ranged from 2.6 km^2 (Waihi Estuary) to 743 km^2 (Kaipara Harbour), measured as water area at high tide.

The analytical workflow involved extracting elevation distributions for each habitat within each estuary, estimating habitat elevation boundaries using ordinal regression, converting elevation boundaries to relative tidal position using STPI, and summarising habitat tidal position across estuaries at national scale.

2.2. Habitat mapping and elevation data

Habitat distributions for seagrass, mangroves, and saltmarsh were obtained from nationally consistent habitat maps developed using Sentinel-2 satellite imagery collected between June 2021 and April 2023 (Bulmer et al., 2024). These maps provide polygon representations of each habitat type across all study estuaries and represent the most comprehensive national mapping of New Zealand coastal wetland habitats currently available.

Elevation data were derived from the New Zealand Coastal LiDAR 1-m Digital Elevation Model (DEM), which amalgamates the most recent coastal LiDAR surveys available nationally (LINZ, 2026). Surveys contributing to this national DEM were acquired from 2005 to present; where multiple surveys overlap at a location, the most recent survey is retained. The DEM has a stated vertical accuracy of ±0.2 m (95% confidence) and horizontal accuracy of ±1.0 m (95% confidence), referenced to NZVD2016. For each estuary, habitat polygons were overlaid with the 1-m DEM, and elevation values were extracted for each 1 × 1 m pixel using the Sample tool in ArcGIS Pro. This produced elevation distributions for each habitat type within each estuary, representing the vertical distribution of habitat surfaces relative to the national vertical datum.

2.3. Tidal data and tidal range estimation

Estuarine tidal characteristics were derived from the NZ Tide Model, a national TELEMAC-2D finite-element tide model (Reeve, 2019, 2021). This model (commonly referred to as NZTIDE) resolves tidal dynamics around the New Zealand coastline for more than 50 estuarine systems. The modelling framework builds on earlier national and regional hydrodynamic implementations (Walters et al., 2001). The model is calibrated against tide-gauge observations and provides spatially consistent estimates of tidal water levels across New Zealand's coastal ocean and estuaries.

For each estuary, TELEMAC model nodes located within the estuary were extracted, and mean values of the Highest Astronomical Tide (HAT) and Lowest Astronomical Tide (LAT) were calculated as a simple, unweighted mean across all nodes within the estuary boundary; node values were not area weighted. The number of model grid nodes varied among estuaries depending on estuary size, ranging from 19 nodes in small estuaries to more than 25,000 nodes in large estuarine systems. These tidal datums were used to define the local tidal frame for each estuary. Model calibration and validation against tide-gauge observations are reported in Reeve (2019) and Reeve and Wadhwa (2021).

2.4. Estimation of habitat elevation boundaries

Within each estuary, elevation distributions of seagrass, mangroves, and saltmarsh overlapped substantially, meaning habitat boundaries could not be defined using simple minimum or maximum elevation thresholds. To estimate consistent habitat elevation boundaries, habitat occurrence was modelled as an ordered function of elevation using weighted ordinal logistic regression.

This approach treats habitat type as an ordered categorical response along a continuous elevation gradient (seagrass - mangroves - saltmarsh). Ordinal regression was selected because habitat transitions occur gradually across elevation gradients and distributions overlap, meaning habitat boundaries are better represented as probabilistic transitions rather than discrete elevation thresholds.

Elevation observations were weighted inversely by habitat sample size within each estuary, so that each habitat class contributed equally to the regression regardless of its areal extent. Habitat elevation boundaries (En) were defined as the elevations at which the most probable habitat class changed along the elevation gradient.

For estuaries without mangroves, binomial logistic regression was used to estimate the elevation boundary between seagrass and saltmarsh. This method enabled consistent estimation of habitat elevation thresholds across estuaries while accounting for overlapping elevation distributions and unequal habitat areas.

The seagrass lower and saltmarsh upper boundaries are fixed at LAT and HAT respectively in every estuary, rather than estimated from the regression, because no habitat class exists below seagrass or above saltmarsh against which such a transition could be estimated. The seagrass-mangrove, mangrove-saltmarsh, and seagrass-saltmarsh (no-mangrove estuaries) boundaries are, by contrast, genuine regression-estimated transitions.

2.5. Calculation of the Standard Tidal Position Index (STPI)

The Standard Tidal Position Index (STPI) was used to compare the tidal position of coastal wetland habitats across regions with differing tidal ranges (Lal et al., 2024; Lovelock et al., 2023). STPI standardises elevation within the local tidal frame, scaling tidal position between −1 (Lowest Astronomical Tide) and +1 (Highest Astronomical Tide), with 0 representing Mean Tide Level (MTL).

STPI was calculated as:

$$STPI = \frac{E_n - MTL}{HAT - MTL}$$

where,

En is the elevation of the upper or lower habitat boundary (m relative to NZVD2016),

MTL is mean tide level (approximated as mean sea level, 0 m), and.

HAT is the highest astronomical tide for the estuary.

All elevation values used in this calculation (the LiDAR DEM, HAT, LAT, and the MSL = 0 m reference) are expressed on a single national vertical datum (NZVD2016); no per-estuary datum conversion was required. For this analysis, mean tide level (MTL) was approximated as mean sea level (MSL), taken as 0 m on the New Zealand (NZVD2016). While MTL and MSL are not strictly equivalent, the difference is small in the predominantly semi-diurnal, low-skewness tidal environments of New Zealand estuaries, and this approximation is consistent with previous applications of STPI in similar settings. A single linear formula was applied across the full tidal frame; a separate formula for elevations above and below MTL was not used. Because most New Zealand estuaries in this dataset have close to symmetric tidal frames around mean sea level, LAT approximates but does not always equal −1 on the STPI scale (see Supplementary Table S5 for estuary-specific departures from symmetry). For the seagrass lower boundary and the saltmarsh upper boundary specifically, STPI was calculated directly from LAT and HAT respectively (STPI = LAT/HAT and STPI = HAT/HAT = 1), rather than from a regression-estimated transition, because no habitat class exists below seagrass or above saltmarsh against which such a transition could be estimated. The seagrass-mangrove, mangrove-saltmarsh, and seagrass-saltmarsh (no-mangrove estuaries) boundaries are, by contrast, genuine regression-estimated transitions. Mangroves were absent from southern estuaries beyond their present distribution limit, and in these estuaries STPI values were calculated only for seagrass and saltmarsh boundaries.

2.6. Estuary-scale and national STPI estimates

STPI values were calculated for the upper and lower elevation boundaries of each habitat within each estuary using estuary-specific tidal datums. National-scale habitat tidal position was then summarised by calculating median and interquartile range (IQR) STPI values across estuaries.

Median values were used rather than means because medians are robust to outliers and to variability among estuaries in tidal range, geomorphic setting, and estuary size. Estuaries were analysed separately depending on whether mangroves were present to evaluate how mangroves influence vertical partitioning of the intertidal zone.

Derived STPI values were evaluated for plausibility by comparison with well-established habitat elevation limits reported in previous New Zealand studies. For example, the upper limit of seagrass is typically close to mean tide level, providing a reference point for assessing the consistency of derived STPI estimates.

2.7. Software and data processing

Spatial analyses and elevation extraction were conducted in ArcGIS Pro using Python (arcpy). Statistical analyses and ordinal regression modelling were conducted in R (R Core Team, 2023).

3. Results

3.1. Estuary coverage and data completeness

Standard tidal position index (STPI) values were calculated for seagrass, mangroves, and saltmarsh across 47 estuaries distributed throughout the North and South Islands. Three estuaries were excluded due to insufficient DEM or tidal data. Mangroves were present in 29 estuaries, all located in the upper North Island, whereas seagrass and saltmarsh occurred across all regions. Elevation distributions were successfully extracted for all mapped habitats using the 1 m coastal LiDAR DEM, and estuary-specific tidal ranges were derived from the TELEMAC tide model for all study sites (see Fig. 3).

3.2. Habitat elevation distributions

Within individual estuaries, elevation distributions of seagrass, mangroves (where present), and saltmarsh exhibited substantial overlap, with distinct modes but broad transitional zones. Seagrass consistently occupied the lowest elevations, extending up to approximately mean tide level, saltmarsh occupied the highest elevations, and mangroves occurred at intermediate elevations in estuaries where they were present (Fig. 4).

Weighted ordinal regression successfully identified non-overlapping, estuary-specific elevation boundaries between habitat types in all estuaries. Upper and lower habitat boundaries (E_n) were defined as the elevations at which the most probable habitat class changed along the elevation gradient. Absolute elevation ranges varied widely among estuaries, reflecting differences in tidal range and geomorphic setting. Representative model fits illustrating habitat probability distributions and elevation boundaries are shown in Fig. 4, while full estuary-specific model outputs and boundary estimates are provided in Supplementary Material (Table S1–S5).

3.3. Estuary-level STPI values

When expressed as STPI, habitat elevation boundaries became directly comparable among estuaries due to standardisation of tidal range inherent in the index.

Across all estuaries, STPI revealed consistent but distinct tidal position ranges for seagrass, mangroves (where present), and saltmarsh. Seagrass occupied the lowest portion of the tidal frame, saltmarsh the highest, and mangroves occurred at intermediate tidal positions.

In estuaries containing mangroves (Table 1), seagrass typically extended from near lowest astronomical tide (though subtidal distributions likely extend below this limit) to STPI values close to mean tide

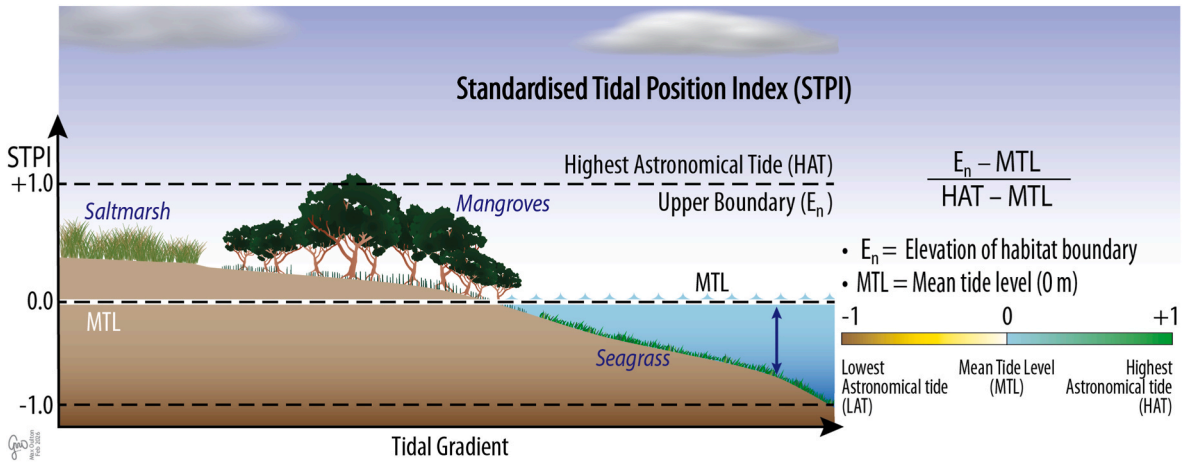


Fig. 3. Conceptual illustration of the Standardised Tidal Position Index (STPI), which normalises habitat elevation using highest astronomical tide (HAT = +1) and mean tide level (MTL = 0) as fixed reference points; lowest astronomical tide (LAT) approximates but does not always equal −1, depending on tidal-frame symmetry (see Supplementary Table S5). Typical tidal zonation of seagrass, mangroves, and saltmarsh is shown.

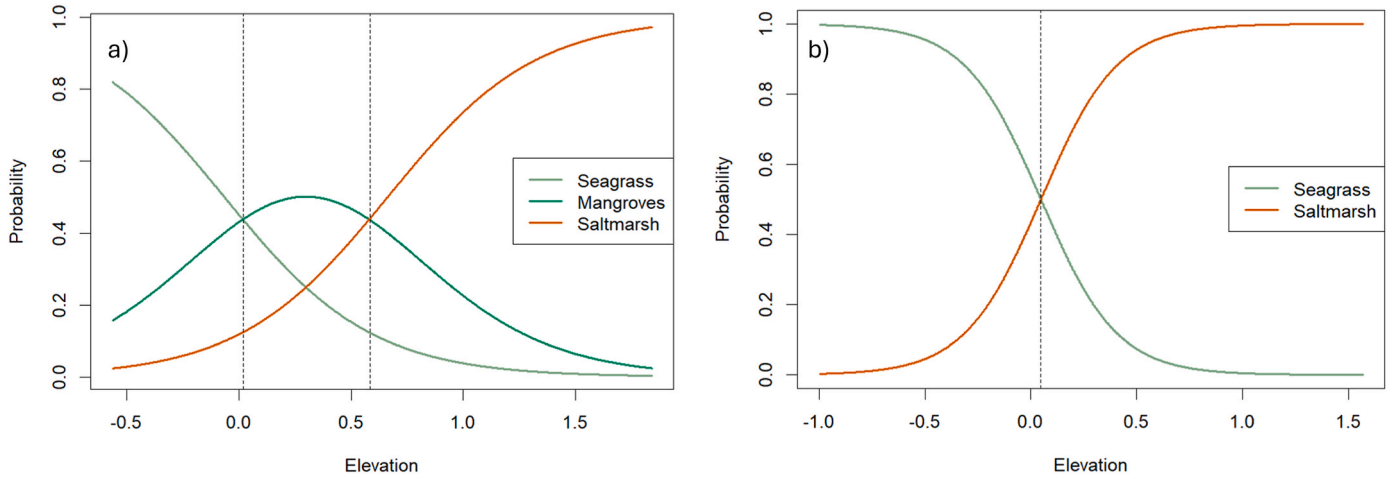


Fig. 4. Representative regression model outputs showing habitat probability distributions across the elevation gradient. (a) Ordinal logistic model for an estuary with mangroves (Whangapoua harbour), illustrating transitions between seagrass, mangroves, and saltmarsh. (b) Binomial logistic model for an estuary without mangroves (Bluff Harbour), showing the transition between seagrass and saltmarsh. Vertical dashed lines indicate estimated habitat elevation boundaries (En).

level, mangroves occupied a mid-intertidal range centred between STPI ≈ 0 and 0.5, and saltmarsh was confined to the upper tidal frame approaching STPI = 1.

In estuaries without mangroves (Table 2), seagrass and saltmarsh directly spanned the intertidal zone, with the seagrass–saltmarsh transition occurring across a broad STPI range (−0.41 to 0.36; median −0.09). This indicates greater variability in the tidal position of habitat boundaries where an intermediate habitat is absent.

3.4. National STPI values

Across the 47 estuaries, national median STPI values were calculated for the upper and lower boundaries of each habitat type, yielding non-overlapping regression-estimated transition thresholds between habitat types (raw elevation distributions themselves overlap substantially within each estuary, as described above) (Fig. 5). These values represent the typical tidal position of seagrass, mangroves, and saltmarsh relative to the local tidal frame. Median values were used to summarise national patterns, as they are robust to outliers arising from regional variability in tidal range, estuary size, and geomorphic setting.

At the national scale, median STPI values indicate that in estuaries with mangroves, seagrass occupies the lower tidal frame with a median lower boundary near lowest astronomical tide (STPI ≈ −0.98) and an upper boundary near mean tide level (median STPI = 0.06). Mangroves occupy an intermediate tidal band between the seagrass–mangrove transition (median STPI = 0.06) and the mangrove–saltmarsh transition (median STPI = 0.37), while saltmarsh occupies the upper tidal frame with an upper boundary consistently at highest astronomical tide (STPI = 1).

In estuaries without mangroves, seagrass similarly extends from near lowest astronomical tide (median STPI ≈ −0.97) but transitions directly to saltmarsh at a lower tidal position (median STPI = −0.09), with saltmarsh again extending to highest astronomical tide (STPI = 1) (Table 3).

These patterns are summarised conceptually in Fig. 5, which illustrates the contrasting vertical organisation of coastal wetland habitats in estuaries with and without mangroves. Estuaries containing mangroves exhibit a three-tiered structure within the tidal frame, with seagrass, mangroves, and saltmarsh occupying distinct elevation bands. In contrast, estuaries without mangroves show a direct transition between seagrass and saltmarsh across a broader portion of the tidal frame.

Together, these results demonstrate consistent vertical structuring of coastal wetland habitats within a standardised tidal framework, while highlighting systematic differences in intertidal organisation associated

Table 1
Standard tidal position index (STPI) values for estuaries with mangroves (n = 29) summarising the estuary-average threshold values at habitat boundaries: between seagrass and mangroves, and mangroves and saltmarsh, within the intertidal zone.† Fixed at LAT (seagrass lower) or HAT (saltmarsh upper) in every estuary; not estimated from the regression. See Methods: Estimation of habitat elevation boundaries.

Estuary	Seagrass lower†	Seagrass | Mangrove	Mangrove| Saltmarsh	Saltmarsh upper†
Aotea Harbour System (AHS)	−0.91	0.22	0.51	1.00
Coromandel Harbour	−0.96	−0.26	0.02	1.00
Firth of Thames	−0.96	−0.05	0.42	1.00
Hokianga Harbour System	−1.01	0.19	0.52	1.00
Houhora Harbour	−1.02	0.40	0.51	1.00
Kaipara Harbour System	−1.00	0.18	0.66	1.00
Kawhia Harbour System (KHS)	−1.01	0.18	0.38	1.00
Mahurangi Harbour System	−0.97	0.07	0.31	1.00
Manaia Harbour	−0.96	−0.23	0.07	1.00
Mangonui Harbour	−1.01	0.09	0.45	1.00
Manukau Harbour System (MHS)	−1.02	0.06	0.39	1.00
Ohiwa Harbour	−0.94	−0.04	0.34	1.00
Opua Inlet System	−1.00	0.10	0.38	1.00
Parengarenga Harbour System	−0.95	−0.11	0.55	1.00
Port Fitzroy/Port Albercrombie	−1.01	−0.43	−0.30	1.00
Raglan Harbour System (RHS)	−0.99	0.14	0.38	1.00
Rangaunu Harbour	−0.99	0.09	0.58	1.00
Tairua Harbour	−0.66	−0.29	−0.03	1.00
Tamaki River	−0.97	−0.02	0.36	1.00
Tauranga Harbour System	−1.00	−0.25	0.23	1.00
Te Puna/Kerikeri Inlet System	−1.01	−0.17	0.18	1.00
Waihi Estuary	−0.70	0.19	0.34	1.00
Waitemata Harbour System	−0.95	0.12	0.37	1.00
Whangapoua Harbour	−0.92	0.02	0.50	1.00
Whangarei Harbour System	−0.98	−0.18	0.23	1.00
Whangaroa Harbour	−1.01	0.11	0.37	1.00
Whangateau Harbour	−0.99	−0.08	0.15	1.00
Whitford Embayment System (WES)	−0.97	0.18	0.49	1.00
Whitianga Harbour (MBS)	−0.97	−0.12	0.25	1.00

with the presence or absence of mangroves.

4. Discussion

4.1. Standardising tidal position across estuaries

This study demonstrates that the Standard Tidal Position Index (STPI) provides a robust and ecologically meaningful framework for comparing the vertical position of coastal wetland habitats across New Zealand estuaries with highly variable tidal ranges. By expressing habitat elevation relative to the local tidal frame, STPI substantially reduces the apparent variability observed in absolute elevation metrics and reveals consistent national-scale patterns in habitat tidal positioning. This finding is consistent with long-standing recognition in coastal ecology that habitat zonation is governed primarily by tidal inundation regime rather than elevation relative to a fixed datum such as mean sea level (Lovelock et al., 2023).

Table 2
Standard tidal position index (STPI) values for estuaries without mangroves (n = 18) showing the habitat breaks between seagrass and saltmarsh within the intertidal zone.† Fixed at LAT (seagrass lower) or HAT (saltmarsh upper) in every estuary; not estimated from the regression. See Methods: Estimation of habitat elevation boundaries.

Estuary	Seagrass lower†	Seagrass | Saltmarsh	Saltmarsh upper†
Akaroa Harbour	−1.02	−0.09	1.00
Avon-Heathcote River	−0.22	0.03	1.00
Bluff Harbour	−1.03	0.03	1.00
Croisilles Harbour	−0.99	−0.08	1.00
Lyttelton Harbour	−0.98	0.04	1.00
Moutere Inlet	−0.97	0.33	1.00
Nelson Haven	−0.98	0.27	1.00
New River (Oreti) Estuary	−0.85	−0.23	1.00
Otago Harbour	−1.04	−0.41	1.00
Pelorous/Kenepuru Sound	−0.97	−0.28	1.00
Porirua Harbour	−0.93	−0.31	1.00
Port Underwood	−1.01	−0.10	1.00
Queen Charlotte Sound (Totaranui)	−1.05	−0.16	1.00
Ruataniwha Inlet	−0.99	0.19	1.00
Waikawa Harbour	−0.96	−0.19	1.00
Waimea Inlet	−0.96	0.34	1.00
Wellington Harbour	−0.95	−0.17	1.00
Whanganui Inlet	−0.38	0.36	1.00

Much of the existing New Zealand literature describes habitat position using local tidal datums such as mean sea level, mean high water neap, or mean high water spring tides. While informative at site scale, these approaches limit direct comparison across regions because tidal range varies markedly among estuaries. The STPI approach reconciles these site-specific descriptions into a common, standardised tidal frame, enabling national synthesis of habitat position while remaining consistent with established ecological understanding. Because tidal inundation in New Zealand estuaries is predominantly semi-diurnal, a given STPI value corresponds to an approximately uniform hydroperiod across estuaries regardless of absolute tidal range, and it is hydroperiod that is the primary driver of intertidal habitat zonation. STPI therefore provides not merely a geometric standardisation of elevation, but a functionally meaningful basis for comparing habitat distributions across systems with contrasting tidal regimes.

4.2. Habitat-specific tidal position ranges

At a national scale, STPI revealed distinct and robust tidal position ranges for each habitat. In contrast, at an estuarine scale, transitions between habitats exhibited high variability, reflecting the large overlap in elevation distribution between habitats and the influence of local processes superimposed on the broader tidal framework. Within individual estuaries, spatial variability in tidal range, driven by estuary geometry, friction, and resonance effects, adds further complexity; however, this was partially addressed here by deriving estuary-specific tidal datums from spatially distributed TELEMAC-2D model nodes rather than a single tidal gauge, providing a more representative characterisation of the local tidal frame. By expressing these patterns within a standardised tidal frame, STPI provides a quantitative synthesis that unifies disparate regional observations into a coherent national picture.

Seagrass consistently occupied the lowest portion of the tidal frame, with upper limits occurring close to mean tide level (median STPI of 0.06 where mean tide level is 0). This aligns with numerous New Zealand studies describing seagrass as primarily occupying low intertidal and shallow subtidal environments, with upper limits constrained by substrate muddification, exposure, and reduced inundation driven desiccation and photorespiration (Turner and Schwarz, 2006;

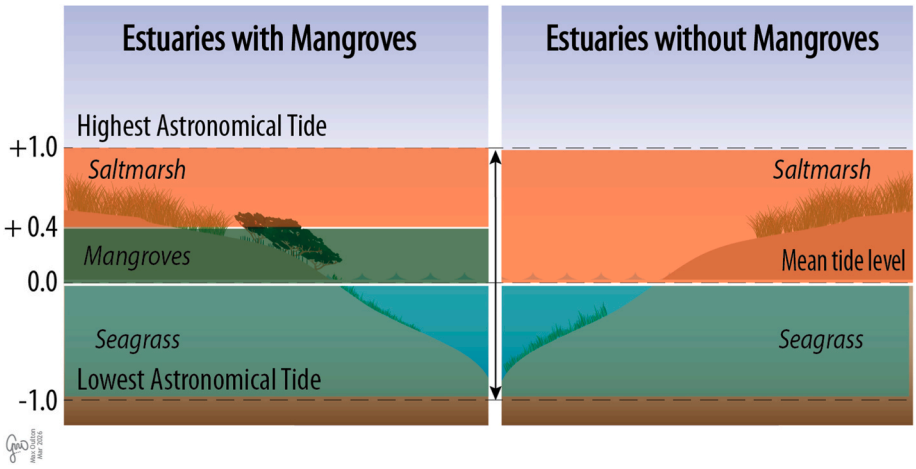


Fig. 5. Conceptual schematic of coastal wetland zonation within the tidal frame for estuaries with and without mangroves. In estuaries with mangroves (left), habitats are vertically structured into distinct zones, with seagrass occupying the lower tidal frame, mangroves the mid-intertidal, and saltmarsh the upper intertidal. In estuaries without mangroves (right), seagrass transitions directly to saltmarsh across a broader elevation range. The central marker indicates mean tide level (STPI = 0). Habitat boundaries reflect national median STPI values derived from weighted ordinal regression, constrained to non-overlapping bands within the tidal frame.

Table 3
National median and interquartile range (IQR) of Standard Tidal Position Index (STPI) values for habitat elevation boundaries across estuaries with and without mangroves.† Fixed at LAT (seagrass lower) or HAT (saltmarsh upper) in every estuary; not estimated from the regression. See Methods: Estimation of habitat elevation boundaries.

Habitat boundary	Estuaries with mangroves Median STPI	IQR	Estuaries without mangroves Median STPI	IQR
Seagrass lower†	−0.98	0.05	−0.97	0.06
Seagrass upper/ Mangrove lower	0.06	0.26	–	–
Mangrove upper/ Saltmarsh lower	0.37	0.25	–	–
Seagrass–Saltmarsh boundary	–	–	−0.09	0.34
Saltmarsh upper†	1.00	0.00	1.00	0.00

Zabarte-Maeztu et al., 2021). The seagrass lower boundary is fixed at LAT in every estuary rather than estimated from elevation data (see Methods); comparatively low seagrass-lower STPI values at Tairua Harbour and Waihi Estuary (Table 1) reflect the asymmetric tidal frames at these two sites rather than an ecological difference in seagrass distribution.

Saltmarsh was confined to the upper intertidal zone, with an upper limit of HAT. However, this methodology constrains estimations of tidal position to the intertidal zone, and it is likely that saltmarshes occur above HAT in the supratidal zone. National guidance and regional mapping studies describe saltmarsh as occurring above mean high water neap (Stevens, 2018, 2020) or spring tides (Stevens and Southwick, 2021). While our tidal model did not use mean high water neaps or springs, looking at regional standard port tidal levels, nationally neap tides have a median STPI of 0.47, and spring tides have a median STPI of 0.81 (LINZ, 2025a). At a national scale, we found that saltmarsh habitat occurs somewhat lower than national neap tides in the intertidal zone, with a median STPI value of 0.37 (i.e., 0.37 of the elevation between mean tide level and HAT), and much lower than spring tide levels. This suggests saltmarsh may tolerate higher levels of inundation than commonly assumed, a hypothesis that warrants direct testing with accretion and migration models rather than being inferred as a demonstrated resilience outcome from tidal position alone.

Where present, mangroves occupied an intermediate elevation within the tidal frame between seagrass and saltmarsh. Our methodology constrained habitats to non-overlapping tidal positions within each estuary, such that the upper tidal limit of mangroves corresponded to the lower boundary of saltmarsh (STPI = 0.37). In natural systems, however, elevation distributions of coastal wetland habitats frequently overlap. Field observations indicate that mangrove forests can extend to elevations near the upper tidal limit in some estuaries, while lower colonisation limits may occur near mean tide level where inundation duration is reduced (Morrisey et al., 2007; Swales et al., 2016).

Our national median values of STPI for seagrass, saltmarshes, and mangroves were comparable with temperate environments in Australia, without mangroves, where seagrass habitat occurred at STPI < 0, and saltmarsh occupied most of the available intertidal zone (STPI = 0 – 1). In our study estuaries with mangroves, estimates of saltmarsh position were somewhat lower in the tidal frame (STPI = 0.37 - 1) than in Australia, where saltmarsh occupied the upper intertidal (STPI 0.45 – 1). In Australian temperate systems, mangroves occupied the mid-intertidal zone between STPI 0-0.45 (Lovelock et al., 2023).

4.3. *Differences between estuaries with and without mangroves*

The presence or absence of mangroves strongly influenced how coastal wetland habitats were partitioned in the intertidal zone. All 29 mangrove-containing estuaries in this dataset are located in the upper North Island, so mangrove presence is confounded with latitude, climate, and regional sediment supply; our results cannot fully separate the effect of mangrove presence itself from these co-varying regional factors. In estuaries without mangroves there is a direct transition between seagrass and saltmarsh, with the seagrass–saltmarsh boundary spanning a much broader range of STPI values between estuaries. This could be due to a lack of competition from mangroves, which have been observed to outcompete saltmarsh and seagrass in ecotones (Swales and Lundquist, 2024) or could be due to physical drivers such as wave exposure, sedimentation rates, freshwater input, and geomorphic setting. Lower marsh communities are more likely to be dominated by sea rush (*Juncus kraussii* subsp. *australiensis* that can extend lower into the tidal frame in sheltered systems or where sediment supply is high, while in more exposed or constrained estuaries the lower limit of saltmarsh occurs higher (Cochran et al., 2014). If mangroves expand poleward under climate change, as has been projected for temperate coastlines globally, these findings suggest that saltmarsh communities currently occupying STPI values between approximately 0.06 and 0.37

— the tidal range presently occupied by mangroves in northern New Zealand estuaries — are likely to face displacement, with implications for saltmarsh extent, blue carbon storage, and intertidal biodiversity in currently mangrove-free systems.

4.4. Responses to sea-level rise

Coastal wetland response to sea-level rise is fundamentally governed by position within the tidal frame and resulting hydroperiod regime, making STPI particularly well suited for assessing vulnerability and adaptive capacity. Habitats occupying lower STPI positions, such as seagrass, are likely to experience increased hydroperiod under rising sea levels, with greater water depth potentially reducing light availability in turbid estuaries, while habitats occupying higher STPI positions, such as saltmarsh, may become increasingly constrained by limited accommodation space and subject to coastal squeeze (Lovelock et al., 2015; Schuerch et al., 2018). The presence of a distinct mid-intertidal habitat in mangrove-containing estuaries suggests a greater capacity for gradual, stepwise reorganisation of habitat zones under sea-level rise and this is reflected in their contemporary migration seaward onto unvegetated, vertically accreting mudflats and landward/upslope expansion into saltmarsh habitats. These findings are consistent with New Zealand assessments highlighting variable resilience of coastal wetlands to sea-level rise depending on geomorphic setting, sediment supply, and landward accommodation space (Lawrence and Bell, 2024).

4.5. Restoration and coastal management

The national STPI ranges derived in this study provide a quantitative basis for identifying tidal elevation envelopes suitable for the persistence or restoration of coastal wetland habitats in New Zealand. Because STPI is directly linked to hydroperiod regime, it offers a practical tool for evaluating whether current or proposed restoration sites fall within the tidal position ranges required to meet habitat prescriptions for target habitats, both under present conditions and under projected sea-level rise, which will be important for predicting permanence of potential sites.

The consistency of national median STPI values suggests that these ranges may be transferable across regions, supporting national-scale restoration planning while remaining grounded in local tidal context. The lower-than-expected tidal position of saltmarsh, specifically the searush (*Juncus kraussii* subsp. *australiensis*) also has implications for saltmarsh restoration opportunities, with prior work assuming a lower bound for saltmarsh of MHWN (Bulmer et al., 2024). This includes saltmarsh restoration opportunities on lowland agricultural land that is gradually being inundated by SLR. This study therefore provides new knowledge to complement existing New Zealand restoration frameworks that emphasise tidal reconnection and elevation suitability but lack a standardised method for comparing sites across regions with differing tidal ranges (Bulmer et al., 2024).

4.6. Limitations and future directions

While STPI provides a powerful standardisation of tidal position, several sources of uncertainty may influence local habitat boundaries. Five estuaries (Avon-Heathcote River, Whanganui Inlet, Tairua Harbour, Waihi Estuary, and New River (Oreti) Estuary) show larger departures from a symmetric tidal frame than the rest of the dataset (Supplementary Table S5); STPI values for these estuaries, particularly the seagrass lower boundary, should be interpreted with caution. Tidal datums were calculated as a simple average of TELEMAC model nodes within each estuary; the range of node-level HAT and LAT values within each estuary (Supplementary Table S2) provides a measure of this spatial variability, though it does not capture spatial variability at the level of individual habitat locations. Formal confidence intervals on the regression-estimated habitat elevation boundaries themselves (as distinct from tidal datum variability) were not computed in this revision and remain a direction for future work, for example via a spatial or bootstrap correction accounting for pixel-level autocorrelation. Habitat elevation boundaries were estimated from ordinal regression on 1 m pixel-level elevation data; spatial autocorrelation among neighbouring pixels was not formally corrected for, though this is expected to primarily affect the precision rather than the direction of estuary-level boundary estimates. LiDAR survey dates (2005-present) do not perfectly coincide with the 2021-2023 habitat mapping period, and LiDAR-derived elevations reflect the vegetated canopy surface rather than bare-earth ground level in mangrove and saltmarsh areas, which may introduce a systematic upward elevation bias relative to seagrass. In addition, STPI represents a static characterisation of tidal position and does not explicitly incorporate dynamic feedbacks between vegetation, sediment accretion, and hydrodynamics. Future work could extend this framework by incorporating time-varying STPI under sea-level rise scenarios, coupling STPI with sediment accretion and hydrodynamic models, and validating predicted habitat transitions using long-term observational datasets from New Zealand estuaries.

5. Conclusions

This study provides the first national-scale assessment of coastal wetland tidal position in Aotearoa New Zealand using a standardised tidal framework. By applying the Standard Tidal Position Index across 47 estuaries, we demonstrate that seagrass, mangroves (in northern estuaries), and saltmarsh occupy consistent and ecologically meaningful portions of the tidal frame, despite large variation in estuary size, tidal range, and absolute elevation.

STPI provides a transferable framework for anticipating how coastal wetlands may respond under sea-level rise. By linking habitat distribution directly to tidal position, STPI offers a valuable tool for integrating coastal wetland ecology, restoration planning, and climate adaptation within a consistent national context.

CRediT authorship contribution statement

P.J. Stewart-Sinclair: Writing – review & editing, Writing – original draft, Validation, Resources, Project administration, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **O. Lam-Gordillo:** Writing – review & editing, Visualization, Methodology, Formal analysis, Data curation. **G. Reeve:** Writing – review & editing, Data curation. **R. Bulmer:** Writing – review & editing, Supervision, Methodology, Data curation, Conceptualization. **I. Zabarte-Maeztu:** Writing – review & editing, Methodology, Data curation, Conceptualization. **A. Swales:** Writing – review & editing, Supervision, Resources, Project administration, Methodology, Investigation, Funding acquisition, Conceptualization.

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.

Acknowledgements

This research was funded by the New Zealand Ministry of Business, Innovation and Employment (Contract CO1X2107) through the Future Coasts Aotearoa Programme.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecss.2026.110323.

Data availability

Code and data supporting this study are archived at https://github.com/phoebestewartsinclair/nz-estuary-tidal-position (v1.0.0) and permanently available via Zenodo: https://doi.org/10.5281/zenodo.21635140.

References

Adame, M.F., Kelleway, J., Krauss, K.W., Lovelock, C.E., Adams, J.B., Trevathan-Tackett, S.M., Noe, G., Jeffrey, L., Ronan, M., Zann, M., 2024. All tidal wetlands are blue carbon ecosystems. Bioscience 74, 253–268.

Bulmer, R.H., Stewart-Sinclair, P.J., Lam-Gordillo, O., Mangan, S., Schwendenmann, L., Lundquist, C.J., 2024. Blue carbon habitats in aotearoa New Zealand—Opportunities for conservation, restoration, and carbon sequestration. Restor. Ecol. 32.

Burge, O.R., 2024. Wetland extent. In: Lohrer, D., et al. (Eds.), Information Stocktakes of Fifty-Five Environmental Attributes Across Air, Soil, Terrestrial, Freshwater, Estuaries and Coastal Waters Domains. Prepared by NIWA, Manaaki Whenua Landare Research, Cawthron Institute, and Environet Limited for the Ministry for the Environment.

Cochran, U.A., Reid, C.M., Clark, K.J., Litchfield, N.J., Marsden, I., Ries, W., 2014. The Avon-Heathcote estuary as a recorder of coseismic vertical deformation. GNS Science.

Dymond, J.R., Sabetizade, M., Newsome, P.F., Harmsworth, G.R., Ausseil, A.-G., 2021. Revised extent of wetlands in New Zealand. N. Z. J. Ecol. 45, 1–8.

Harty, C., 2009. Mangrove planning and management in New Zealand and South East Australia–A reflection on approaches. Ocean Coast Manag. 52, 278–286.

He, Q., Li, Z., Elphick, C.S., Sun, Y., Sutherland, W.J., Taylor, N.G., Wasson, K., 2025. Making restoration effective for dynamic coastal wetlands. Glob. Change Biol. 31, e70455.

Holmquist, J.R., Windham-Myers, L., 2022. A conterminous USA-scale map of relative tidal marsh elevation. Estuaries Coasts 45, 1596–1614.

Johnson, P., Gerbeaux, P., 2004. Wetland Types in New Zealand. Department of Conservation, Wellington, New Zealand.

Lal, K.K., Woodroffe, C.D., Zawadzki, A., Rogers, K., 2024. Coastal wetland elevation dynamics, sedimentation, and accommodation space across timescales. Estuaries Coasts 47, 1828–1843.

Lawrence, J., Bell, R., 2024. Coastal Hazards and Climate Change Guidance. Ministry for the Environment, New Zealand.

LINZ, 2025a. Standard Port Tidal Levels. Land Information New Zealand.

LINZ, 2025b. Tides Around New Zealand. Land Information New Zealand.

LINZ, 2026. New Zealand Coastal Lidar 1m DEM. Land Information New Zealand.

Lovelock, C.E., Adame, M.F., Bradley, J., Dittmann, S., Hagger, V., Hickey, S.M., Hutley, L.B., Jones, A., Kelleway, J.J., Lavery, P.S., 2023. An Australian blue carbon method to estimate climate change mitigation benefits of coastal wetland restoration. Restor. Ecol. 31, e13739.

Lovelock, C.E., Cahoon, D.R., Friess, D.A., Guntenspergen, G.R., Krauss, K.W., Reef, R., Rogers, K., Saunders, M.L., Sidik, F., Swales, A., 2015. The vulnerability of Indo-Pacific mangrove forests to sea-level rise. Nature 526, 559–563.

Lundquist, C.J., Morrisey, D.J., Gladstone-Gallagher, R.V., Swales, A., 2013. Managing mangrove habitat expansion in New Zealand. Mangrove Ecosystems of Asia: Status, Challenges and Management Strategies. Springer, pp. 415–438.

Mcleod, E., Chmura, G.L., Bouillon, S., Salm, R., Björk, M., Duarte, C.M., Lovelock, C.E., Schlesinger, W.H., Silliman, B.R., 2011. A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO2. Front. Ecol. Environ. 9, 552–560.

Morrisey, D., Beard, C., Morrison, M., Craggs, R., Lowe, M., 2007. The New Zealand Mangrove: Review of the Current State of Knowledge, 325. Auckland Regional Council Technical Publication, p. 162.

Morrisey, D.J., Swales, A., Dittmann, S., 2010. The Ecology and Management of Temperate Mangroves. CRC Press, Boca Raton(USA), pp. 43–160.

Neville, J.A., Guntenspergen, G.R., Grace, J.B., Osland, M.J., Chivoiu, B., 2024. A conterminous united states–wide validation of relative tidal elevation products. Estuaries Coasts 47, 2227–2237.

Reeve, G., 2019. Offshore Boundary Conditions for a NZ Tidal Model: What's the Best Global Tide Model for our Region? National Institute of Water & Atmospheric Research Ltd, Prepared for Land Information, New Zealand.

Reeve, G., Wadhwa, S., 2021. New Zealand Tidal Model Calibration. National Institute of Water & Atmospheric Research Ltd, Prepared for Land Information, New Zealand.

Rogers, K., Kelleway, J.J., Saintilan, N., Megonigal, J.P., Adams, J.B., Holmquist, J.R., Lu, M., Schile-Beers, L., Zawadzki, A., Mazumder, D., 2019. Wetland carbon storage controlled by millennial-scale variation in relative sea-level rise. Nature 567, 91–95.

Schuerch, M., Spencer, T., Temmerman, S., Kirwan, M.L., Wolff, C., Lincke, D., McOwen, C.J., Pickering, M.D., Reef, R., Vafeidis, A.T., 2018. Future response of global coastal wetlands to sea-level rise. Nature 561, 231–234.

Stevens, C., Chiswell, S., 2009. Ocean currents and tides - tides, Te Ara – the Encyclopedia of New Zealand. https://teara.govt.nz/en/ocean–currents–and–tides/page–2.

Stevens, L.M., 2018. Tokomairiro Estuary: Broad Scale Habitat Mapping 2018. Prepared for Otago Regional Council, Wriggle Coastal Management, p. 40.

Stevens, L.M., Forrest, B.M., 2020. Broad Scale intertidal habitat mapping of Te Awarua-o-Porirua Harbour. prepared for Greater Wellington regional council. Salt Ecology 46.

Stevens, L.M., Southwick, M., 2021. Evaluation of a simple GIS-based methodology to determine estuary extent and assess potential changes in response to sea level rise. prepared for Greater Wellington Regional Council. Salt Ecology 11.

Stewart-Sinclair, P.J., Bulmer, R.H., Macpherson, E., Lundquist, C.J., 2024. Enabling coastal blue carbon in Aotearoa New Zealand: opportunities and challenges. Front. Mar. Sci. 11.

Sulman, B.N., Wang, J., LaFond-Hudson, S., O'Meara, T.A., Yuan, F., Molins, S., Hammond, G., Forbrich, I., Cardon, Z.G., Giblin, A., 2024. Integrating tide-driven wetland soil redox and biogeochemical interactions into a land surface model. J. Adv. Model. Earth Syst. 16, e2023MS004002.

Swales, A., Bell, R.G., Lohrer, A., 2020. Estuaries and lowland brackish habitats. In: Hendtlass, C., Morgan, S., Neale, D. (Eds.), New Zealand Coastal Society, pp. 55–64.

Swales, A., Bentley, S.J., Lovelock, C.E., 2015. Mangrove-forest evolution in a sediment-rich estuarine system: opportunists or agents of geomorphic change? Earth Surf. Process. Landf. 40, 1672–1687.

Swales, A., Denys, P., Pickett, V.I., Lovelock, C.E., 2016. Evaluating deep subsidence in a rapidly-accreting mangrove forest using GPS monitoring of surface-elevation benchmarks and sedimentary records. Mar. Geol. 380, 205–218.

Swales, A., Lundquist, C., 2024. Mangrove forest extent and quality. In: Lohrer, D., et al. (Eds.), Information Stocktakes of Fifty-Five Environmental Attributes Across Air, Soil, Terrestrial, Freshwater, Estuaries and Coastal Waters Domains. Prepared by NIWA, Manaaki Whenua Landare Research, Cawthron Institute, and Environet Limited for the Ministry for the Environment.

Swales, A., Pritchard, M., McBride, G.B., 2021. Biogeomorphic evolution and expansion of mangrove forests in New Zealand's sediment-rich estuarine systems. Dynamic Sedimentary Environments of Mangrove Coasts. Elsevier, pp. 3–45.

Turner, S., Schwarz, A.-M., 2006. Management and conservation of seagrass in New Zealand: an introduction. Science for Conservation 264. Department of Conservation, Wellington, New Zealand.

Walters, R.A., Goring, D.G., Bell, R.G., 2001. Ocean tides around New Zealand. N. Z. J. Mar. Freshwater Res. 35, 567–579.

Wang, F., Lu, X., Sanders, C.J., Tang, J., 2019. Tidal wetland resilience to sea level rise increases their carbon sequestration capacity in United States. Nat. Commun. 10, 5434.

Zabarte-Maeztu, I., Matheson, F.E., Manley-Harris, M., Davies-Colley, R.J., Hawes, I., 2021. Fine sediment effects on seagrasses: a global review, quantitative synthesis and multi-stressor model. Mar. Environ. Res. 171, 105480.