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Estuaries

Dynamic Ecosystems at the Land-Sea Interface

Edited by Leonel Pereira



Estuaries - Dynamic Ecosystems at the Land- Sea Interface

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Aims and Scope of the Series

The world of Earth Sciences, considering the interactions within the geosphere and between the geosphere–biosphere, is a place in which a large number of scientists find and have found over time their own relationship or sector of application precisely because it is absolutely transversal to many disciplines and subdisciplines, which do not necessarily fall within the Geosciences. The objective of this book series is to welcome original scientific contributions both in consolidated contexts and in new frontiers of research, as well as review papers included in the various disciplines of Earth Sciences, but above all, those that show a modern and transversal vision of applications and impacts on the community in a particular historical context, which, following the COVID-19 pandemic, has shifted global attention to sectors that were previously more neglected. In particular, those of mining research and fossil and renewable energy sources, environmental geology and the sustainable use of natural resources and impacts on the built environment, land consumption, geoarchaeology, forensic geology, geotourism/geoheritage, georisks and climate and environmental changes, considered at different scales, up to new applications of geostatistical and geospatial analysis, GIS and artificial intelligence for the definition of forecasting models and scenarios in various sectors of basic and applied research.

Meet the Series Editor



Dr. Maurizio Lazzari has a Ph.D. in Earth Science and is a researcher at the Italian National Research Council, Institute of Cultural Heritage Sciences. Since 2001, Dr. Lazzari has been a Professor of Pedology at the University of Basilicata (Italy) and a geoarchaeologist at the University of Salento (Italy). His research activities are focused on natural and anthropic hazards and risk factors, aimed at safeguarding and conserving settlements and the historical-monumental heritage of the Mediterranean, with particular attention to landslide processes, susceptibility maps, monitoring, and modelling. Since 2004, he has been working as a scientific coordinator for several national research projects studying landslides and triggering factors, natural and anthropogenic risks, geological and geomorphological mapping, soil erosion, preservation of historical and archaeological sites, enhancement of degraded areas, geo-touristic use, and the protection of the landscapes. He is the author of about 150 scientific publications in national and international journals, monographs, book chapters, and conference proceedings concerning applied geology, geomorphology, dynamics of artificial reservoirs, soil erosion, landslides, geoarchaeology, hydrogeological instability, natural hazards, monitoring, cultural landscape, UNESCO Heritage, geoarchaeology, and geo-tourism.

Meet the Volume Editor



Leonel Pereira holds a degree in Biology (scientific branch) and a Ph.D. in Biology, specializing in Cell Biology, from the Faculty of Science and Technology of the University of Coimbra (Portugal), where he is currently an Associate Professor with Habilitation. He is also an integrated researcher at the Centre for Functional Ecology (CFE – Science for People & Planet). His scientific interests focus on Marine Biodiversity, particularly macroalgae; Marine Biotechnology, with an emphasis on bioactive compounds from algae; and Marine Ecology, especially environmental assessment of coastal and estuarine systems. Since 2008, he has been the author and editor of MACOI – Portuguese Seaweeds Website (<http://www.flordeutopia.pt/macoi/>), a widely used digital resource on the marine flora of Portugal. He is the author/editor of more than 25 books and 91 book chapters, and has published over 180 scientific articles in international peer-reviewed journals. Throughout his career, he has delivered more than 100 lectures and oral presentations at national and international scientific events. His contributions to marine science have been recognized with several distinctions, including the Francisco de Holanda Prize (Honorable Mention, 1998), the King D. Carlos Award (18th edition), and the CHOICE Award – Outstanding Academic Title 2016 for his book *Edible Seaweeds of the World* (CRC Press). He has also been included in the World's Top 2% Scientists 2021-2025 list released by Stanford University and Elsevier, recognizing his global scientific impact and research excellence.

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Preface

Estuaries are among the most dynamic and influential ecosystems on Earth, shaped by the continuous interaction of freshwater and marine forces. Their physical complexity, ecological richness, and socio-economic importance make them essential components of coastal landscapes worldwide. This volume, “Estuaries – Dynamic Ecosystems at the Land-Sea Interface”, brings together six complementary chapters that collectively illuminate the structure, function, and vulnerability of estuarine systems across diverse geographic and scientific contexts.

The Introductory Chapter establishes the conceptual and ecological foundation for the book. It explores the defining hydrodynamic processes of estuaries, including salinity gradients, tidal mixing, sediment transport, and the formation of distinct ecological zones. It also examines the flora and fauna that inhabit these transitional environments, from macroalgae, seagrasses, and salt marsh vegetation to plankton, benthic invertebrates, and fish, highlighting their adaptations, ecological roles, and sensitivity to anthropogenic pressures such as eutrophication, habitat alteration, and climate-driven change.

Building on this framework, the chapter *Estuaries: The Lifeblood of Coastal Fisheries in West Africa* demonstrates the profound socio-ecological importance of estuarine systems for regional fisheries, food security, and coastal livelihoods. It synthesizes ecological characteristics, fisheries dependence, and the escalating threats posed by pollution, altered river flows, and climate change.

The chapter *Population Performance and Resilience in Polychaetes as Environmental Indicators of Estuarine Ecosystems* introduces a mechanistic, trait-based modelling approach to understanding how estuarine indicator species respond to environmental quality. By revealing new continuums of life-history variation, it offers refined tools for ecological assessment and resilience prediction.

In *Living Links: Coastal Vegetated Ecosystems in UK Estuaries as Blue Carbon Sinks and Food Web Foundations*, the focus shifts to seagrass meadows and salt marshes as blue-carbon sinks, biodiversity hotspots, and structural components of estuarine food webs. The chapter integrates ecological monitoring, stable isotope analysis, and policy developments to highlight the role of these habitats in climate regulation and adaptive coastal management.

The chapter *Perspective Chapter: The Significance of Characterising Geologic and Hydrogeologic Settings of Aquifers for Submarine Groundwater Discharge and Seawater Intrusion Estimations* emphasizes the importance of understanding the geologic and hydrogeologic characteristics of aquifers when estimating submarine groundwater discharge (SGD) and seawater intrusion (SWI). It argues that many current methods overlook geological heterogeneity, even though variations in lithology, structure, and

hydraulic conductivity strongly influence groundwater flow. By integrating detailed geological mapping, 3D modeling, and hydrogeologic analysis, researchers can greatly reduce uncertainties in SGD and SWI assessments and make more reliable decisions for managing coastal aquifers.

Finally, *Changes and Evolution in Positive and Negative Storm Surges Recorded in the Río de la Plata Estuary from 1905 to 2024* provides a rare century-long perspective on hydrodynamic extremes. By comparing real and hypothetical scenarios, it reveals how storm surge patterns evolve under rising water levels and shifting climate conditions, offering valuable insights for coastal risk assessment.

Together, these six chapters present an integrated and multidisciplinary view of estuarine science, linking hydrodynamics, ecology, fisheries, biogeochemistry, and climate impacts. This volume aims to support researchers, practitioners, and policy-makers in advancing the sustainable management of estuaries, ensuring that these vital ecosystems continue to sustain biodiversity, coastal resilience, and human well-being in an era of accelerating environmental change.

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Introductory Chapter: Estuaries

Leonel Pereira

1. Introduction

1.1 Anthropogenic pressures, coastal system, transition

Simply put, an estuary is a partially enclosed coastal body of water with a free connection to the ocean, located at the terminal part of a watershed, where seawater measurably mixes with freshwater from rivers [1].

More comprehensively, an estuary is an ecosystem situated in a deep coastal indentation with restricted communication with the sea—either permanent or intermittent—characterized by three distinct zones. The fluvial tidal zone lacks salinity but is affected by tidal cycles. The mixing zone is where freshwater and marine masses meet, creating strong physical, chemical, and biological gradients. The turbidity zone in the open sea, downstream of the mixing zone, is influenced by the tidal plume at low tide [2].

The physical and chemical gradients of an estuary largely depend on the freshwater discharge from the river, the morphology of the riverbed near the mouth, and surface evaporation. These factors determine hydraulic circulation and influence the dispersion of suspended particles, including salinity. Depending on the balance between river discharge and tidal flow, estuaries can be homogeneous, partially mixed, or stratified. Evaporation further classifies estuaries as positive, negative, or neutral, depending on whether freshwater input exceeds, falls short of, or equals surface evaporation [3].

The average salinity distribution along the estuary defines different zones. Upstream, the fluvial zone (limnetic) has a salinity below 0.5. The oligohaline zone, at the estuary's top, fluctuates between 0.5 and 5. The mesohaline zone, in the upper estuary section, ranges from 5 to 18. The polyhaline zone, covering the middle and lower estuary sections, varies between 18 and 25 in the middle and 25–30 in the lower section. Finally, the euhaline zone, at the estuary's mouth, records salinity levels above 30 [4].

The interaction between freshwater and seawater leads to diffusion and mechanical mixing, affecting the flocculation of transported particles. These particles settle in the estuary's inner zones, particularly in areas with reduced hydrodynamic circulation, near the margins, and in vegetation [5]. Beyond tidal oscillations and currents, wind plays a significant role in sediment dynamics within the estuary. In shallow areas, wave action can resuspend fine sediments, continuously altering the morphology of the estuary's floor [6].

Despite daily environmental fluctuations, estuarine ecosystems—such as macroalgae beds, seagrass meadows, and salt marshes—support high biological productivity, comparable to tropical forests. Ecologically, estuaries foster complex interactions among species. Although species diversity is lower than in many other ecosystems,

those adapted to estuarine conditions thrive in high densities. These organisms contribute to estuary formation, habitat diversification, primary and secondary production, ecosystem services, and ultimately, the preservation or enhancement of environmental health [7].

Historically, estuaries have served as shelters and sources of sustenance for human populations. Ports, fishing, aquaculture, trade, and recreational activities have flourished in these areas [8]. However, their natural fragility makes them highly susceptible to anthropogenic pressures. Physical alterations to shorelines, wastewater discharge, human activities within and around the estuary, disruptions to river connectivity and seasonal flow, and the introduction of nonnative species degrade the estuary's hydromorphology, increase organic and contaminated loads, and create biological imbalances that may lead to the decline or extinction of local species [9].

2. Anthropogenic pressures

Port activities, navigation, agriculture, aquaculture, and various industrial and construction activities are among the most common undertakings in estuaries and their surroundings. These activities can exert significant pressure on the ecosystem [10]. The discharge of effluents, leading to increased concentrations of nutrients and other substances in estuarine waters, may result in the rapid proliferation and colonization of opportunistic species, a process known as eutrophication. This often leads to the complete occupation of the available space [11]. Beyond these obvious pressures, even practices with seemingly lower environmental impact can still contribute to significant disturbances in estuarine ecosystems and their communities. The occupation and differential use of estuarine territories play a critical role. Fishing, salt production, tourism, and recreational activities, despite not directly releasing effluents into the environment, can also constitute substantial sources of environmental pressure on estuaries [12].

Currently, these and other sources of disturbance are frequently used to characterize and quantify the pressures present in each system. It is well understood that different biological elements are affected in distinct ways by the various acting pressures. Moreover, for comparable intensities of a given pressure, the responses of each biological element tend to be relatively consistent [13, 14].

3. Estuarine fauna and flora

Estuaries are dynamic ecosystems that support a rich diversity of plant and animal life. The unique blend of freshwater and saltwater creates a habitat where species must adapt to fluctuating conditions. The estuarine environment is shaped by factors such as salinity, turbidity, substrate availability, and seasonal variations [13, 14].

3.1 Macroalgae

Macroalgae play a crucial role in estuarine ecosystems, contributing to primary production and providing habitat for various organisms. Their distribution is largely influenced by salinity levels, with higher diversity near the marine-influenced areas. As one moves upstream, the number of species decreases, particularly among Rhodophyta (red algae) and Phaeophyceae (brown algae). Chlorophyta (green algae) are more commonly found in upstream areas but also exhibit reduced species richness [15].

The presence of macroalgal blooms can significantly impact estuarine habitats. Excessive growth can lead to habitat degradation, oxygen depletion, and shifts in community structure [16]. Historically, in southern Europe, macroalgae and seagrass have been harvested for agricultural use, particularly in the Ria de Aveiro estuary (Portugal), where species such as *Gracilaria/Gracilariopsis* (Rhodophyta) (**Figure 1a**), *Rhizoclonium*, and *Ulva* (Chlorophyta) (**Figure 1b**) were collected as fertilizer (Moliço) [17].

3.2 Seagrasses

Seagrasses are vital components of estuarine ecosystems, providing shelter, stabilizing sediments, and supporting diverse marine life. However, they are highly vulnerable to environmental changes. Thick macroalgal mats can overshadow seagrass beds, reducing light availability and oxygen levels, particularly during warm summer nights. Persistent macroalgal overgrowth can lead to the decline of *Zostera* meadows, altering the primary producer community and affecting associated fauna [18].

Seagrass habitats are recognized for their ecological importance and are considered in environmental assessments under European directives. For example, studies in Portugal have focused on seagrass dynamics in estuaries such as Mondego, Ria Formosa, Ria de Aveiro, Sado, and Mira, often examining their interactions with associated communities [19].

3.3 Salt marsh plants

Salt marshes develop along estuarine margins in areas with low hydrodynamic activity. These habitats are shaped by tidal influences, sedimentation processes, and vegetation dynamics. Pioneer plant species play a key role in stabilizing sediments and facilitating succession. Historically, salt marshes were drained for agriculture and urban development, but their ecological significance is now widely recognized [20].

Salt marsh plants play a crucial role in estuarine ecosystems, adapting to high salinity and fluctuating water levels while providing habitat and stabilizing sediments. Some common species include sea purslane (*Sesuvium portulacastrum*), a ground-covering plant with fleshy green leaves and pinkish flowers, and common

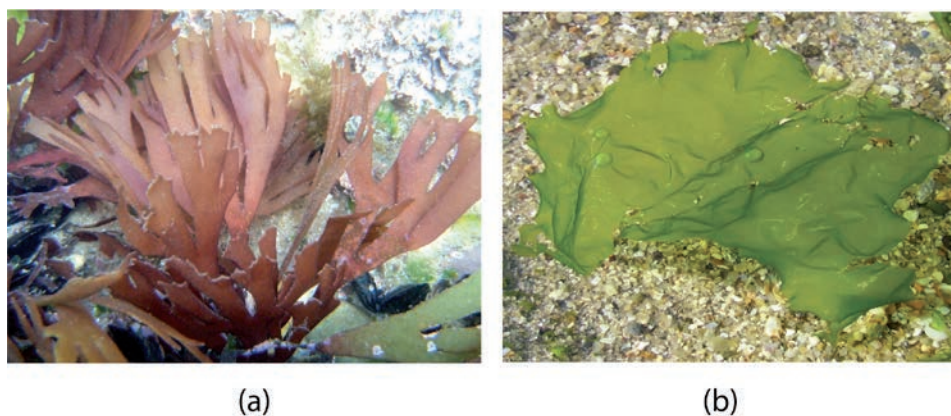


Figure 1.
a. *Gracilaria multipartita*. b. *Ulva lactuca*.

glasswort (*Salicornia bigelovii*), a distinctive species with finger-like, succulent leaves that change color from green to red. Saltwort (*Batis maritima*) is another salt-tolerant plant, known for its ability to thrive in saline environments. Smooth cordgrass (*Spartina alterniflora*) dominates low marsh areas near the water, while salt meadow cordgrass (*Spartina patens*) grows in slightly higher marsh zones with curved, vine-like leaves. Sea blite (*Suaeda linearis*) is a scrubby plant with dark green elliptical leaves and a greenish stem, contributing to sediment stabilization [21]. Among submerged estuarine plants, eelgrass (*Zostera marina*) plays a vital role in marine ecosystems by providing shelter and food for various species. Additionally, sea aster (*Tripolium pannonicum*), a flowering plant often found in salt marshes, adds to the biodiversity of these habitats, attracting pollinators and serving as an important part of the ecological balance. These species collectively contribute to the resilience and productivity of estuarine environments [22].

3.4 Plankton

3.4.1 Phytoplankton

Phytoplankton communities, consisting of aquatic microorganisms with photosynthetic capabilities that drift within the water column, are fundamental to primary production, sustaining much of the biodiversity in estuarine ecosystems. Due to their rapid response to environmental changes and contaminants, they play a crucial role in estuarine environments and are considered key components of these ecosystems. In some estuarine systems, phytoplankton biomass generally increases from April onward, influenced by temperature, light availability, and nutrient supply, while lower values are observed in autumn and winter [23].

The dominant taxonomic groups in estuarine phytoplankton include diatoms and dinoflagellates, although high densities of Chlorophyceae (green algae), Cryptophyceae (cryptophytes), Cyanobacteria (blue-green algae), Euglenophyceae (euglenophytes), and Prasinophyceae (prasinophytes) can also occur, particularly in some estuaries [24]. In general, nutrient concentration in the water column does not limit phytoplankton growth, except for silica, which is essential for diatoms. However, nutrient concentrations tend to decrease from upstream to downstream, following a similar pattern to nutrient availability. Factors such as river flow, as observed in the Tagus, and salinity, as seen in the Sado, along with air temperature and atmospheric photosynthetic radiation, can influence phytoplankton production in estuaries [25].

Elevated nutrient concentrations, primarily from pig farming in the subbasin, as well as dredging, sports activities, and dam construction, contribute to the high phytoplankton productivity observed in this estuary [26]. In some estuaries, significant seasonal reductions in river flow during spring and summer, controlled by dams, create an environment with low silica-to-nitrogen and nitrogen-to-phosphorus ratios, favoring cyanobacteria dominance over other phytoplankton groups. Occasionally, harmful or toxic species have been detected in some estuaries, including *Ceratium fusus*, *Pseudonitzschia* cf. *seriata*, *Scrippsiella trochoidea*, *Thalassiosira gravida* in some estuaries, *Dinophysis acuta*, *Dinophysis acuminata*, *Gymnodinium catenatum*, *Microcystis* sp., and *Anabaena* spp. [27].

3.4.2 Zooplankton

Zooplankton communities consist of aquatic organisms that lack photosynthetic capabilities, have limited mobility, and drift within the water column. The most frequent

groups in estuarine environments include Copepoda, Cladocera, Appendicularia, and Chaetognatha. Among these, Copepoda is typically the most abundant taxonomic group, with the genus *Acartia* often presenting the highest densities. Other occasionally observed groups include Amphipoda, Isopoda, Cumacea, Mysidacea, and Euphausiacea, as well as larval forms of Decapoda (zoea and megalopa), Mollusca (veliger and post-veliger stages of Gastropoda and Bivalvia), Polychaeta, Cirripedia (*Cypris* and *Nauplius*), Bryozoa, and Echinodermata (pluteus). Some larval stages of decapod species are found exclusively in the inner areas of estuarine systems [28].

In adjacent coastal areas, the dominant zooplankton groups are Copepoda and Tintinnida. The abundance of Copepoda is primarily due to the prevalence of nauplii and species of Cyclopoida, while *Stenosemella nivalis* is a common representative of Tintinnida. Tunicata and Cladocera tend to be more frequent in certain coastal zones. Seasonal variations in zooplankton abundance are more pronounced in some areas, with significant fluctuations in diversity influenced by water flow. High river discharge can strongly impact zooplankton communities, leading to shifts in species composition [29].

Zooplankton populations are influenced by factors such as salinity, temperature, and water turbidity. Generally, their abundance peaks during spring and summer-autumn, coinciding with increased phytoplankton availability, which serves as their primary food source [30].

3.5 Benthic invertebrates

Macroinvertebrate communities are well characterized in many estuarine systems. These organisms play a crucial role in the food chain, supporting the influx of fish, the stopover of shorebirds, and, to a lesser extent, some marine mammals searching for food. Their importance stems from the high productivity of estuarine environments, which sustain complex trophic networks that often extend to top predators [31].

These communities are typically dominated by crustaceans, mollusks, and polychaetes. Among crustaceans, amphipods (*Corophium* sp.) and isopods (*Cyathura carinata*, *Lekanesphaera levii*) are particularly notable. Mollusks include gastropods (*Peringia ulvae*) and bivalves (*Abra alba*, *Corbula gibba*, *Kurtiella bidentata*). Polychaetes are represented by families such as Nereididae (*Hediste diversicolor*), Capitellidae (*Mediomastus fragilis*), Cirratulidae (*Aphelochaeta marioni*), and Spionidae (*Aonides oxycephala*) [32]. The abundance of some of these species makes estuaries attractive for human exploitation, such as the collection of *Hediste diversicolor* for use as live bait in fishing or the harvesting of *Cerastoderma edule* (cockle) and *Solen marginatus* (razor clam) for traditional local cuisine. Additionally, the exotic species *Corbicula fluminea* is increasingly found in the subtidal zones of estuarine headwaters, where lower salinity levels favor its presence as a freshwater organism [33].

Due to their ecological significance, benthic invertebrates are essential for assessing the quality of coastal and estuarine systems, as required by environmental directives. Like other biological indicators, macroinvertebrates have been incorporated into a specialized assessment tool [34].

3.6 Fish

As with most aquatic organisms, relatively few fish species can survive in brackish environments. Estuaries, characterized by constant fluctuations in salinity due to tidal cycles, present a challenging habitat where only species with high salinity

tolerance can thrive. In closed coastal lagoons, conditions become particularly critical during periods of heavy rainfall, when salinity drops to levels unsuitable for species requiring some degree of saltwater, and in summer, when rising temperatures lead to oxygen depletion, especially in shallow areas. However, due to the limited number of fish species capable of surviving and thriving in these ecosystems, those that do tend to occur in high densities, benefiting from reduced competition and favorable trophic conditions [35].

Few fish species are permanent residents of brackish systems, meaning they can complete their life cycle within these environments. Among the most characteristic resident fish are gobies (family Gobiidae), wrasses (family Labridae), and seahorses (*Hippocampus* spp., family Syngnathidae). Gobies are more commonly found in unvegetated substrates, while wrasses and seahorses are typically associated with algae or aquatic plant communities, making them more frequent in lower estuarine and lagoon areas. In southern estuaries, the toadfish (*Halobatrachus didactylus*, family Batrachoididae) is a dominant species, with males constructing nests and providing parental care, similar to gobies. In the pelagic zone, common resident fish include anchovies (*Engraulis encrasicolus*, family Engraulidae) and silversides (*Atherina* spp., family Atherinidae) [32].

A significant portion of the fish fauna in brackish zones consists of species that use these environments as nursery areas. These species reproduce in the ocean and enter estuaries as larvae or juveniles, benefiting from protection against predators and optimal trophic and thermal conditions that support rapid growth until they reach a size suitable for returning to marine environments with reduced predation risk. Many species exhibit this behavioral pattern, including European seabass (*Dicentrarchus labrax*, family Moronidae), various species of seabream (*Diplodus* spp., family Sparidae), and flatfish such as common sole (*Solea solea*, order Pleuronectiformes) [36].

Another important component of brackish fish communities is diadromous migratory species, which transition between freshwater and marine environments. These are classified into anadromous fish, which spawn in rivers or upper estuarine areas and develop in the ocean, and catadromous fish, which spawn in the ocean and grow in estuaries or rivers. Key catadromous species include the European eel (*Anguilla anguilla*, family Anguillidae), thinlip mullet (*Liza ramada*, family Mugilidae), and European flounder (*Platichthys marinus*, family Pleuronectidae). Anadromous species include the sea lamprey (*Petromyzon marinus*, family Petromyzonidae), Atlantic salmon (*Salmo salar*), sea trout (*Salmo trutta*, family Salmonidae), and shad (*Alosa* spp., family Clupeidae). Salmonids are restricted to colder regions due to their preference for low temperatures [37, 38].

In recent decades, particularly since the 1990s, climate change has contributed to a gradual “tropicalization” of estuarine fish communities, with a decline in species typical of northern regions and an increase in populations of species characteristic of warmer southern areas [39].

3.7 The impact of opportunistic species

The excessive proliferation of opportunistic species, primarily macro- and microalgae, poses environmental and economic problems while also disrupting the ecological balance of communities within the ecosystem. Typically, macroalgal blooms lead to a decline in the quality of transitional waters’ habitats. In extreme cases, changes in dissolved oxygen concentration can result in the formation of an anoxic environment. Such conditions lead to the elimination of numerous organisms, affecting both directly

and indirectly the populations that depend on them [40]. These phenomena of organic enrichment, eutrophication, and environmental degradation have been extensively studied in the Mondego Estuary (Portugal), revealing diverse impacts on bird populations, benthic macroinvertebrates, and the coverage of macroalgae and seagrasses [41].

4. Conclusions

Estuarine ecosystems are highly dynamic and complex environments that support a diverse range of species, from phytoplankton and zooplankton to benthic invertebrates and fish. The organisms inhabiting these systems are adapted to fluctuating conditions of salinity, temperature, and nutrient availability, playing vital roles in maintaining ecological balance and supporting food webs. However, these habitats face increasing pressures from anthropogenic activities, climate change, and habitat degradation, leading to shifts in species composition and potential declines in biodiversity. Understanding ecological interactions within estuaries is essential for effective conservation and management efforts, ensuring that these productive ecosystems continue to sustain both wildlife and human communities. Recognizing the significance of estuarine environments in global biodiversity underscores the need for ongoing research and sustainable practices to mitigate environmental pressures and protect these valuable systems for future generations.

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Chapter 2

Estuaries: The Lifeblood of Coastal Fisheries in West Africa

Noble Kwame Asare

Abstract

Estuarine systems along the West African coastline serve as essential ecological and socio-economic foundations for marine and coastal fisheries. These transition zones between riverine and marine environments support high biological productivity driven by dynamic physicochemical and hydrographic conditions, and diverse habitats such as mangrove swamps, tidal flats, and seagrass beds. Thirteen coastal nations from Mauritania to Nigeria and the island state of Cabo Verde in West Africa rely extensively on these systems for fisheries recruitment, artisanal and industrial harvests, and food and nutrition security. Estuaries function as nursery grounds for numerous commercially important species that sustain regional marine capture fisheries, which remain central to national economies and coastal livelihoods. However, escalating anthropogenic pressures, including habitat degradation, pollution, altered river flows, over-exploitation, and climate change, threaten their ecological integrity and the fisheries they support. The continued productivity of coastal fisheries will therefore depend on effective conservation and management strategies that protect estuarine habitats, maintain water quality, and strengthen community resilience. This work synthesises the ecological characteristics, fisheries relevance, and major threats affecting West African estuarine systems, and presents recommendations for sustainable management to safeguard the vital contributions of these systems to regional food systems and economies.

Keywords: estuaries, coastal fisheries, West Africa, nursery habitats, fisheries productivity, coastal livelihoods, anthropogenic threats

1. Introduction

Estuaries are described as semi-enclosed coastal water bodies where freshwater from rivers mixes with tidal seawater to produce brackish conditions and to form a transitional zone between fluvial and marine environments. They are highly dynamic systems characterised by strong gradients and variability in salinity, temperature, sediment load, oxygen, and nutrients. Estuaries often change in characteristics both spatially and temporally through the influence of physical forcing associated with the tides and river discharge, resulting in mixing of nutrient-rich freshwater and seawater with large inputs of organic matter. The mixing regimes depend on the relative strengths of river discharge velocity, tidal forcing, basin morphology, and bathymetry. These characteristics place many estuaries among the most productive habitats on Earth [1].

Estuarine systems, therefore, play a crucial role in supporting coastal and marine fisheries. They support rich benthic communities, extensive marshes and seagrass beds, and provide critical habitats for various marine fish species of commercial importance (both fin- and shellfish), birds and other taxa, contributing disproportionately to regional biodiversity [2, 3]. Despite their immense productivity and critical ecological functions, estuarine systems are increasingly subjected to severe anthropogenic pressures that negatively affect their significant ecological roles in coastal fisheries [4]. Recognising these threats is, therefore, essential for promoting sustainable fisheries management and ensuring the socio-economic well-being of coastal communities in West Africa.

This chapter provides a comprehensive examination of the ecological, economic, and social relevance of estuarine systems for coastal fisheries in West Africa. It synthesises current knowledge on estuarine processes, fish assemblages, anthropogenic threats, and management strategies, emphasising the central role these systems play in sustaining regional fisheries and coastal livelihoods.

2. Description of the study region

West Africa spans a broad section of the African continent, extending approximately from 4°N to 27°N latitude and 17°W to 15°E longitude, encompassing a wide range of climatic and ecological zones. The region has thirteen coastal countries and is characterised by the arid Sahel areas in the north, humid tropical coasts in the south, and extensive riverine, estuarine, and coastal systems that support high biological

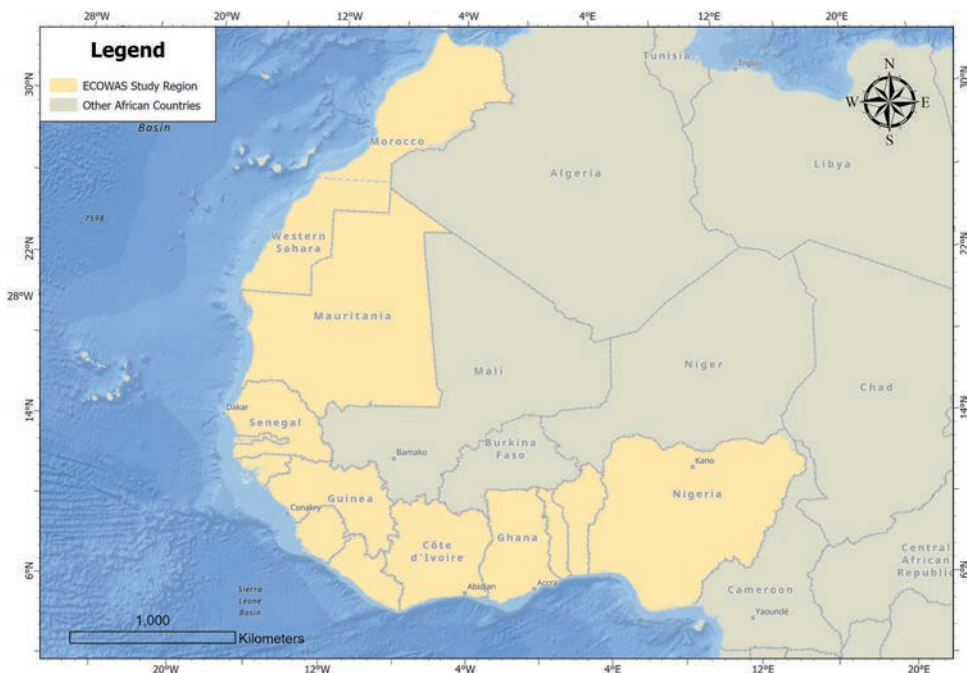


Figure 1.
A map of West Africa showing the thirteen coastal nations as the focus of this study, namely Mauritania, Senegal, The Gambia, Guinea-Bissau, Guinea, Sierra Leone, Liberia, Côte d'Ivoire, Ghana, Togo, Benin, Nigeria, and the island state of Cabo Verde.

productivity. Most of these countries border the Gulf of Guinea, while Mauritania and Senegal lie along the northwestern Atlantic coast. Cabo Verde is located offshore, influenced by open Atlantic waters.

The coastal waters of these thirteen nations (**Figure 1**) form part of two major large marine ecosystems. The Canary Current Large Marine Ecosystem includes Mauritania and Senegal and is characterised by seasonal upwelling that enhances productivity. The Guinea Current Large Marine Ecosystem stretches along the Gulf of Guinea coasts from Côte d'Ivoire to Nigeria and is influenced by equatorial and coastal currents as well as riverine inputs.

3. Ecological characteristics, fisheries contributions, and management challenges of West African estuarine systems

3.1 Characteristics and processes of tropical estuarine systems

Tropical estuarine systems are shaped by the dynamic interaction of freshwater inflows, tidal exchange, geomorphology, and climatic conditions. These processes create complex spatial and temporal gradients in salinity, temperature, sediment load, and nutrient concentrations, which together support some of the most productive ecosystems on the planet [5, 6]. In West Africa, estuaries occur across diverse climatic settings ranging from the arid coasts of Mauritania to the humid tropical regions of the Gulf of Guinea, producing a broad spectrum of estuarine types that vary in hydrology, geomorphic structure, and ecological function [7].

Freshwater discharge is a primary driver of estuarine dynamics. Seasonal rainfall patterns across West Africa produce pronounced variations in river flow, with substantial freshwater inputs during the wet season and reduced discharge during the dry season. These fluctuations influence the spatial extent of salinity intrusion and shape the distribution of estuarine habitats [8, 9]. During periods of intense river flow, salinity gradients shift seaward, increasing turbidity and nutrient availability. In contrast, reduced flows in the dry season allow for greater tidal penetration and higher salinity conditions upstream. These seasonal cycles create a temporally dynamic environment that supports a wide range of estuarine flora and fauna [5].

Tidal processes exert strong influence on estuarine hydrodynamics. Semidiurnal tides common to much of the West African coast enhance vertical and horizontal mixing, facilitating the distribution of nutrients and organic matter and aiding larval transport and recruitment into coastal waters [6, 10]. The interplay of tides and river flows shapes the structure of estuarine circulation, producing well-mixed, partially mixed, or stratified systems depending on local conditions and geomorphic setting [5, 9].

The geomorphology of West African estuaries is diverse. The region includes coastal plains, mangrove lined estuaries, lagoonal systems, river dominated deltas, and bar-built enclosed lagoons. Many estuaries are bordered by extensive mangrove forests that stabilise shorelines, regulate sediment deposition, and contribute organic matter that fuels detrital food webs [11]. Seagrass beds and tidal flats further enhance habitat heterogeneity and productivity. These habitats support high biodiversity and provide essential feeding, sheltering, and breeding areas for numerous marine organisms [10, 12].

Together, these physical and ecological processes create conditions that sustain highly productive food webs and support the life cycles of many fish species that

are economically important to coastal fisheries [2, 12]. Tropical estuarine systems in West Africa, therefore, represent critical ecological assets that support both the natural functioning of coastal ecosystems and the socio-economic stability of coastal communities [7, 13].

3.2 Ecological roles of estuarine systems in coastal and marine fisheries

Estuarine ecosystems provide essential ecological functions that support coastal and marine fisheries across West Africa. Their unique characteristics, attributed to freshwater inputs, tidal influence, and high habitat complexity, produces conditions that promote productivity and resource availability for a wide range of fish and invertebrate species [6, 10]. The structural diversity of mangrove channels, tidal creeks, seagrass meadows, mudflats, and shallow subtidal zones enhances ecological connectivity and supplies critical habitats for estuarine-dependent fauna [2, 11].

A central role of estuaries is the provision of nursery habitat. Numerous marine species that support fisheries rely on estuarine habitats during vulnerable juvenile stages when access to abundant food and refuge from predators increases survival [2, 12]. In West Africa, species such as croakers, snappers, groupers, mullets, penaeid shrimps, and several small pelagic fishes demonstrate strong ontogenetic links to estuarine habitats, where early life stages benefit from elevated productivity and protection offered by mangrove and lagoon systems [14]. These habitats frequently support high densities of predominantly juvenile fish and contribute to their survival and growth [15]. This nursery function therefore enhances recruitment into coastal fisheries and maintains stock resilience.

Estuaries also sustain fisheries through detritus and nutrient pathways that enrich adjacent coastal waters. Mangrove leaf litter, phytoplankton production, and sediment nutrient fluxes fuel detrital food webs and support secondary production across trophic levels [11, 13]. Tidal exchange facilitates the export of organic matter and juvenile fish from estuarine systems into coastal fishing grounds, strengthening broader ecosystem productivity and enhancing catch potential [9, 10].

Seasonal hydrological regimes further support the fisheries by regulating the timing and magnitude of recruitment pulses. Wet season freshwater discharge drives nutrient enrichment and promotes plankton blooms that sustain larval and juvenile fish, while dry season saline intrusion expands habitable areas for estuarine tolerant species [6, 8]. This hydrological variability supports a diverse assemblage of species with different life history strategies and enables estuaries to function as biological corridors between freshwater and marine environments.

These ecological roles make estuaries indispensable to coastal fisheries in West Africa. Their contribution to recruitment, productivity, and trophic connectivity is therefore essential to the sustainability of artisanal and industrial fisheries and directly influences regional food security and economic stability [7, 12].

3.3 Diversity of estuarine fish assemblages in West Africa

Estuarine fish assemblages in West Africa exhibit remarkable diversity shaped by strong environmental gradients, hydrological variability, and habitat heterogeneity along the regional coastline. The transitional nature of estuaries allows the coexistence of freshwater, brackish water, and marine species, resulting in species rich communities that contain both resident and transient taxa [12, 14]. These assemblages reflect the interplay between salinity regimes, geomorphology, vegetation structure,

and tidal connectivity, which together determine the composition and spatial distribution of fish within estuarine systems [16, 17].

Across the region, fish assemblages are dominated by several functional groups with distinct ecological roles. Estuarine resident species such as species of the genera *Tilapia*, *Sarotherodon*, *Arius*, and *Ethmalosa* often tolerate wide salinity ranges and complete most of their life cycles within estuarine environments [14]. These species play foundational roles in local food webs and represent an important component of small-scale fisheries. Marine migrant species, including croakers (*Pseudotolithus* spp.), snappers (*Lutjanus* spp.), grunts (*Pomadasys* spp.), and penaeid shrimps (e.g. *Penaeus* spp.), enter estuaries seasonally to feed or to use sheltered habitats as nursery grounds during juvenile stages [2, 12].

West African estuaries also support a diverse suite of freshwater species that descend into brackish water zones during periods of high river discharge. Species in families such as Cichlidae, Clupeidae, Cyprinidae, and Mormyridae exhibit varying degrees of salinity tolerance that allow temporary or seasonal occupation of estuarine habitats [16]. This influx contributes to the dynamic nature of estuarine fish communities, where species composition often changes markedly between wet and dry seasons [14].

Habitat structure also plays a major role in shaping assemblages. Mangrove swamps, tidal creeks, mudflats, and shallow lagoon systems each support distinct fish communities based on their levels of shelter, food availability, and hydrodynamic conditions [10, 11]. Mangrove-fringed estuaries typically support high abundances of juvenile fishes [15] due to elevated food resources and physical protection, while open lagoon systems may exhibit greater dominance of estuarine-tolerant or marine-derived species [12, 18].

The biogeography of West African estuaries reflects both regional and local environmental processes. Northern estuaries in Mauritania and Senegal tend to show lower species richness due to strong salinity variability and arid climatic conditions, whereas estuaries in Guinea, Sierra Leone, and the Niger Delta exhibit much higher species diversity due to extensive mangrove cover, stable year-round freshwater inflow, and diverse habitat mosaics [18]. These patterns align with broader trends in tropical estuarine ecology where habitat complexity and freshwater availability consistently promote higher fish diversity [10, 17]. Specific examples from the sub-region include reports of 18 to 35 species in the Kakum River Estuary in the Central Region of Ghana [15, 19, 20], depending on the season; 67 species in the Gambia River estuary [21]; and 57 species from the lower Bonny River of the Niger Delta in Nigeria [22].

Fish assemblages of West African estuaries, therefore, demonstrate high ecological variability and functional importance. The combination of resident estuarine species, freshwater inputs, and marine-derived juveniles creates diverse communities that are essential for the sustainability of coastal fisheries. These assemblages also reflect the sensitivity of estuarine ecosystems to hydrological change, habitat degradation, and anthropogenic pressures that affect ecological balance and long-term fishery productivity across the region.

3.4 Coastal and marine fisheries production in West Africa

Estuarine systems are among the most productive aquatic environments globally. This high productivity sustains diverse food webs that support juvenile and adult fish, crustaceans, molluscs, and plankton communities. Detrital pathways are particularly important in tropical estuaries, where organic matter exported from wetlands fuels nearshore food webs that support coastal marine fisheries [23].

The combination of allochthonous and autochthonous nutrient sources results in elevated secondary production that enhances fisheries productivity in adjacent coastal zones. Studies have shown that fish biomass and species richness often decline in regions where estuarine productivity is reduced through degradation or hydrological alteration [24].

Many estuarine fish species are central to artisanal fisheries and local food security. The West African sub-region produces an estimated 4–5 million metric tonnes (MT) of fish from coastal and marine fisheries representing about 70% of the annual total fish production in the sub-region for which estuarine systems play a critical role.

Reported marine capture fisheries production across West African coastal nations (**Table 1**) demonstrates both the scale of dependence on marine resources and the uneven distribution of fisheries output along the regional coastline.

Mauritania reports the highest national marine capture volume at 967,706 MT, followed by Nigeria with 783,102 MT, and Senegal with 513,957 MT. These three nations collectively account for a substantial proportion of the sub-region's total marine landings. Their reported catches represent major contributions to national food and nutrition security because marine fish constitute essential and often irreplaceable sources of affordable animal protein for coastal and inland populations.

Country	Production (MT)	Year	Source
Mauritania	967,706	2018	<i>FAO fisheries and aquaculture country profiles: Mauritania</i> [25]
Senegal	513,957	2021	<i>FAO WAPI factsheet on aquaculture growth potential: Senegal</i> [26]
The Gambia	55,686	2016	<i>FAO fisheries and aquaculture country profiles: The Gambia</i> [27]
Guinea Bissau	~6550	2016	<i>FAO fisheries and aquaculture country profiles: Guinea Bissau</i> [28]
Guinea	~326,270	2021	<i>FAO WAPI factsheet on aquaculture growth potential: Guinea</i> [29]
Sierra Leone	215,140	2022	<i>FAO fisheries and aquaculture country profiles: Sierra Leone</i> [30]
Liberia	~12,600	2017	<i>FAO fisheries and aquaculture country profiles: Liberia</i> [31]
Côte d'Ivoire	~92,000	2023	<i>FAO Meeting demand for tasty tilapia in Côte d'Ivoire</i> [32]
Ghana	~170,149	2019	<i>FAO Fishery and aquaculture country profile and national estimates: Ghana</i> [33]
Togo	~18,600	2017	<i>FAO fisheries and aquaculture country profiles: Togo</i> [34]
Benin	14,800	2017	<i>FAO fisheries and aquaculture country profiles: Benin</i> [35]
Nigeria	783,102	2020	<i>FAO WAPI factsheet on aquaculture growth potential: Nigeria</i> [36]
Cabo Verde	~26,580	2018	<i>FAO fisheries and aquaculture country profiles: Benin</i> [37]

Table 1.
Reported annual marine capture fisheries production (metric tonnes) for the thirteen West African coastal states from the most recent year available.

In Senegal and Nigeria, particularly, per capita fish consumption remains among the highest in sub-Saharan Africa, and a large share of this intake originates directly from these marine capture fisheries. The economic importance is also pronounced, as fisheries contribute significantly to national revenue, employment and export earnings, with artisanal sectors supporting at least hundreds of thousands of livelihoods in each nation.

Countries with intermediate production levels also play vital roles in regional food systems. Ghana's reported 168,437 MT and Guinea's 127,358 MT are critical to domestic markets where fish remains a major component of diets and a stabilising factor in household level nutrition. Sierra Leone's 78,000 MT and Côte d'Ivoire's 58,000 MT likewise support both urban and coastal food supply chains. Even in nations where capture volumes are lower in absolute terms, such as The Gambia (55,686 MT), Guinea-Bissau (37,700 MT), Benin (22,500 MT), Liberia (22,300 MT), and Togo (17,000 MT), fisheries production remains central to livelihoods and local economies.

These countries rely heavily on artisanal fishing, which provides widespread employment, supports local processing and trade networks, and sustains the nutritional requirements of communities where fish is often the primary or sole affordable source of high-quality protein. Cabo Verde's reported 26,580 MT reflects a smaller but economically significant fishery that supports national food supply and contributes to export-oriented tuna value chains.

Across all thirteen countries, the significance of these reported production values extends beyond their numerical magnitude. Marine capture fisheries act as safety nets for vulnerable coastal populations, buffer households against economic shocks, and contribute to national foreign exchange earnings where exports are substantial. In much of West Africa, small pelagic species that dominate many of these landings are essential not only for dietary sufficiency but also for micronutrient intake, especially in lower income households. As a result, even modest changes in capture production can have significant consequences for nutrition outcomes and household food access.

Although the production values represent broad scale marine harvests, estuarine systems are vital for a considerable portion of this regional productivity. Countries with extensive estuarine networks, such as Senegal, Guinea, Guinea Bissau, Sierra Leone and Nigeria, benefit from these systems through enhanced fish recruitment that supports both artisanal and industrial sectors. Even in lower volume producers, estuarine-dependent species dominate local catches, linking estuarine health directly to the sustainability of the fisheries that support national food security and coastal livelihoods. Thus, the reported capture values are not only measures of marine exploitation but also indirect indicators of the ecological functions provided by estuarine systems across the West African coastline.

The scatter plot (**Figure 2**) compares the total marine capture (in metric tonnes) against the percentage of national GDP derived from fisheries for 13 West African nations. The X-axis uses a logarithmic scale to accommodate the wide disparity in capture production volumes. The plot illustrates a complex relationship, highlighting Mauritania and Nigeria as having the highest total capture, while several smaller economies (Cabo Verde, Gambia, Liberia, Mauritania) show a proportionally higher dependence on the sector, with a fisheries share of >5.0% of their national GDP.

The analysis of marine capture production volume against the fisheries sector's contribution to gross domestic product (GDP) of respective nations reveals a complex, non-linear relationship across the West African coastal states. The scatter plot data also indicates that total annual output alone is not the principal factor dictating the economic dependence of a country on its coastal and marine fisheries resources.

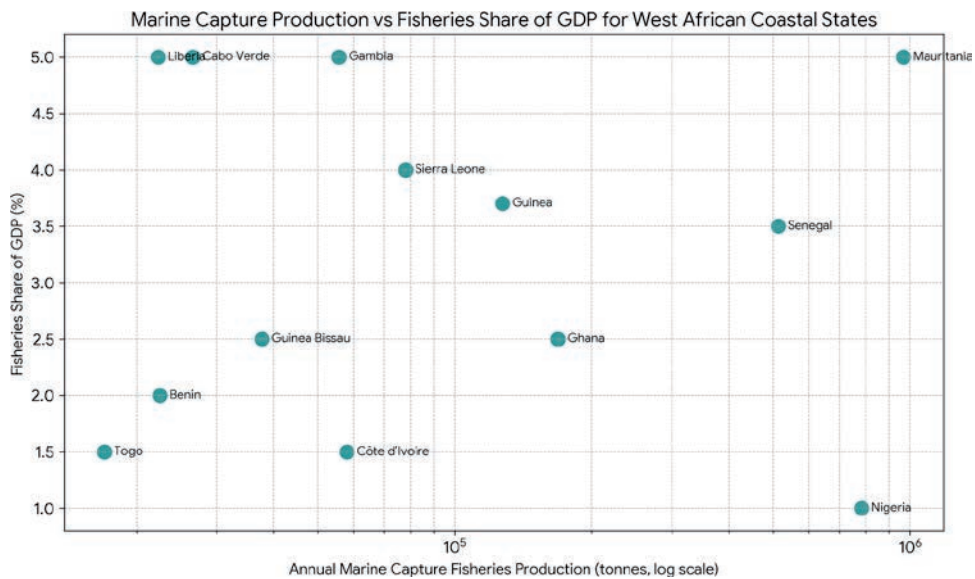


Figure 2.

Relationship between annual marine capture production and fisheries contribution to gross domestic product (GDP) in West African Coastal States. Data sources Refs. [38–40].

The regional producers fall into distinct categories. Mauritania and Nigeria dominate in terms of sheer capture volume, both reporting figures exceeding 700,000 MT. Mauritania, benefitting from the highly productive Canary Current upwelling system, records the highest capture (over 900,000 MT) in the sub-region, which corresponds to a significant 5.0% of its GDP. This establishes the sector as critical to the national economy, albeit not overwhelmingly dominant. In contrast, Nigeria's comparable capture volume accounts for only a minimal 1.0% of its GDP. This stark difference illustrates the “economic dilution” effect, where the output of a major resource sector is diminished in proportional significance by a large, diversified national economy.

Conversely, a key cluster of nations exhibits a disproportionately high economic reliance on capture fisheries, ranging from 3.7 to 5.0% of their GDP. Countries like Cabo Verde, The Gambia, and Liberia all show a 5.0% dependence despite having relatively low capture volumes (less than 60,000 MT). This high proportional share signals a heightened level of economic vulnerability. For these smaller, less diversified economies, negative exogenous shocks such as climate change impacts, resource depletion from overfishing could severely destabilise national economic performance and threaten coastal livelihoods. Sierra Leone and Guinea also fall into this category of high reliance, affirming the vital role of the sector in food security and employment in these states.

Mid-range producers like Senegal (with ~ 513,000 MT, contributing 3.5% of GDP) and Ghana (with ~ 168,000 MT contributing about 2.5% to its GDP) demonstrate more moderate reliance. Their positioning reflects economies that, while still heavily influenced by marine fisheries resources, have achieved greater diversification compared to their high-dependence neighbours. At the lower end of the spectrum, countries like Togo and Benin exhibit both low capture volumes and a low share of GDP (about 1.5 and 2.0%, respectively), which may indicate resource constraints, lower fishing capacity, or a significantly advanced non-fisheries economic base.

The data demonstrates that the economic importance of fisheries in West Africa, as measured by GDP share, often inversely correlates with the overall size and diversification of national economies. These findings underscore a critical management duality whereby high capture states face large-scale resource sustainability challenges, while countries with high GDP dependence (often moderate-to-low capture) face greater risks to socio-economic resilience should resource stability falter.

3.5 Socio-economic relevance of estuarine systems in West Africa

Estuarine systems play essential socio-economic roles in sustaining livelihoods, food security, and local and national economies across West Africa. Their function in coastal and marine fisheries, as earlier described, provide employment and income to millions of people living along the coastlines of West African sub-region [41, 42]. For many coastal communities, estuarine and lagoon fisheries represent the primary source of animal protein and serve as a foundation for nutrition, cultural identity, and household resilience [43].

The artisanal fisheries sector benefits significantly from the accessibility and productivity of estuarine waters. Small-scale fishers depend heavily on estuaries for daily catches of finfish and crustaceans. Shellfisheries such as oyster and cockle harvesting support tens of thousands of women in Senegal, the Gambia, Sierra Leone, and Ghana; providing reliable income streams that are often more stable than offshore fisheries [44]. These fisheries together supply informal markets and local trade networks that support fish processors, traders, transporters, and a wide range of secondary occupations, thereby creating extensive socio-economic linkages [45]. Women are particularly active within estuarine value chains through fish processing, smoking, and marketing, making these systems important for gender-inclusive economic opportunities [46].

Estuarine production also contributes significantly to national economies. Countries such as Nigeria, Senegal, Ghana, and Guinea rely on estuarine-dependent species for a major share of their marine landings, which in turn support domestic consumption and regional trade [41, 42]. The economic value of estuarine fisheries extends beyond direct harvests to include cultural ecosystem services such as spiritual attachment to fishing grounds, customary resource practices, and traditional ecological knowledge that guide sustainable use within many communities [47].

Food and nutrition security in West Africa also depends strongly on estuarine production. Fish derived from estuaries often represent affordable and locally available sources of essential nutrients including proteins, polyunsaturated fatty acids, vitamins, and minerals that are critical for human health [48]. In several countries, per capita fish consumption would decline substantially without the contributions of estuarine and lagoon fisheries that supply both fresh and processed fish to inland populations [41, 43].

Furthermore, estuarine systems contribute to livelihood resilience by offering seasonal safety nets during agricultural lean periods or economic shocks. When crop yields decline or employment opportunities fluctuate, many coastal households increase dependence on estuarine fishing and gathering to meet subsistence needs [45]. This adaptive capacity is particularly important in regions experiencing climate-related stressors, shoreline change, and declining offshore fish stocks.

3.6 Anthropogenic threats and their implications to coastal fisheries

Multiple human activities across catchments and along the coast of West Africa have altered estuarine hydrology, sediment dynamics, habitat extent, and water

quality. Rapid population growth, urban expansion, industrial activity, and intensified resource extraction along the coastline have accelerated habitat degradation, pollution, and biodiversity loss within estuarine catchments [7]. These pressures disrupt ecological processes that support productive fish assemblages and compromise the socio-economic resilience of communities that depend on estuarine resources for food and income.

One of the most significant threats is the widespread loss and degradation of mangrove forests. Unsustainable fuelwood harvesting, conversion of wetlands for agriculture and aquaculture, and urban development have resulted in extensive mangrove decline across the region [49–51]. Mangrove removal reduces nursery habitat, alters sediment dynamics, diminishes organic matter production, and decreases recruitment of estuarine-dependent species into coastal fisheries [10, 11]. Countries with extensive mangrove systems, such as Nigeria, Guinea Bissau, and Sierra Leone, have experienced substantial losses that directly affect fisheries productivity and household income.

Pollution poses another major challenge. Untreated domestic wastewater, agricultural runoffs, industrial effluents, and petroleum contamination degrade water quality and reduce habitat suitability for fish and invertebrates. Heavy metal contamination from illegal small-scale mining activities in Ghana, Guinea, and Sierra Leone, along with petroleum spillage in the Niger Delta, have led to long-term ecological damage in several estuaries [52, 53]. Elevated nutrient loads cause eutrophication, harmful algal blooms, and hypoxic conditions that reduce species richness and impair fish health and survival [9]. These impacts diminish the availability and quality of fish resources that coastal communities depend upon.

Overfishing and destructive fishing practices exacerbate ecological decline. Intense artisanal fishing pressure through open access, combined with the incursion of industrial trawlers into inshore and inshore zones, has reduced the abundance of key fish species and altered community structure [41, 54]. Illegal, unreported and unregulated (IUU) fishing contributes significantly to stock depletion [55, 56], particularly for estuarine-dependent species. Habitat damaging practices, including the use of small mesh nets and fine monofilament gear, further deplete juveniles and undermine recruitment into coastal fisheries [12, 57].

Hydrological alteration also presents a growing threat. Large-scale river damming, water diversion, and upstream extraction reduce freshwater inflow and modify salinity regimes in estuaries such as that of the Volta, Senegal, and Gambia rivers [8, 58]. Reduced freshwater discharge diminishes nutrient delivery, increases saline intrusion, and constrains suitable habitat for estuarine and freshwater species. This hydrological imbalance disrupts seasonal fish migration patterns and limits the productivity of estuarine nurseries.

Climate-related processes add further strain. Sea level rise, increased storm surges, and changing rainfall patterns intensify erosion, flooding, and habitat loss along the West African coast [59, 60]. Rising temperatures and shifting salinity gradients alter species distributions and reduce the resilience of estuarine ecosystems already weakened by anthropogenic stress. These climate-driven changes amplify the vulnerability of coastal fisheries and the communities that depend on them.

3.7 Sustainable management strategies and recommendations

Effective management of estuarine systems in West Africa requires an integrated approach that addresses ecological degradation, resource over-exploitation, and

socio-economic vulnerability. Estuaries in the region are embedded within densely populated coastal zones where competing demands for land, water, and fisheries heighten pressure on already fragile ecosystems [7]. Because estuarine productivity sustains artisanal fisheries, management frameworks must balance conservation objectives with livelihood need to ensure long-term sustainability and equitable resource use.

Strengthening governance is central to the effective management of estuarine ecosystems. Many West African countries face challenges related to weak enforcement, limited institutional capacity, and fragmented regulatory frameworks that reduce the effectiveness of existing environmental and fisheries laws [41, 60]. Integrated coastal zone management strategies that bring together government agencies, traditional authorities, scientists, and community organisations can improve coordination, support evidence-based decision making, and promote adaptive responses to ecological change [61].

Mangrove rehabilitation programmes in Nigeria, Senegal, and Guinea Bissau demonstrate that community-led restoration of degraded habitats can improve shore-line stability, enhance nursery habitats, and strengthen fish recruitment [62, 63]. Restoration efforts should be expanded to other degraded estuarine zones where deforestation, erosion, and pollution have disrupted ecological processes. Protecting mangroves, seagrass beds, and tidal flats through marine protected areas or community resource reserves can safeguard essential habitats and promote recovery of estuarine-dependent species [10].

Effort to reduce over-exploitation and encourage sustainable harvests is necessary for improved fisheries management. Community-based fisheries governance, seasonal closures, gear restrictions, and monitoring of juvenile bycatch can prevent recruitment overfishing and maintain stock productivity [45, 54]. Efforts to combat illegal and unreported fishing, particularly by industrial fleets operating near inshore zones, are vital to protect local access rights and ensure equitable resource distribution [41].

Water quality management is equally important. Strengthening pollution control requires investments in wastewater treatment, regulation of mining effluents, and improved management of agricultural runoff. Reducing contaminant loads will help restore ecological function and improve the safety and marketability of estuarine-derived fish [52]. Managing hydrological alteration through coordinated upstream water releases can also sustain natural salinity gradients and support migration patterns of estuarine and freshwater species [8, 58].

Resilience of coastal communities depends on both ecological restoration and socio-economic interventions. Strengthening fish value chains through improved processing facilities, enhanced cold storage, and market access can increase incomes and reduce post-harvest losses that disproportionately affect small-scale fishers, especially women processors [64]. Diversification of livelihoods through aquaculture, ecotourism, mangrove-based enterprises, and climate-smart agriculture can reduce dependence on declining fisheries and buffer households against climate-related or economic shocks [47].

Knowledge transfer and participatory monitoring offer additional benefits. Incorporating traditional ecological knowledge into management decisions fosters community ownership and improves compliance, while participatory monitoring provides real time information on habitat health, species abundance, and fishing effort [45]. Education and awareness programmes can further strengthen stewardship by improving understanding of the ecological and economic importance of estuaries.

Building long-term resilience requires coordinated regional action. Because many estuarine-dependent fish stocks migrate across national boundaries, collaborative management within the Canary Current Large Marine Ecosystem and Guinea Current Large Marine Ecosystem frameworks can support shared data systems, harmonise fisheries policies, and address transboundary threats such as pollution and industrial fishing pressure [54].

4. Conclusions

West African estuarine systems are the critical ecological infrastructure for the coastal fisheries of the region. These productive, dynamic zones provide essential nursery habitats for fish recruitment, which are directly responsible for supporting regional marine capture and maintaining food security and millions of livelihoods across the sub-region. This work demonstrates that while total fish capture varies widely across the sub-region, the economic importance (share of GDP) is often inversely related to the size of the national economy, creating high socio-economic vulnerability in smaller states. However, the integrity of these systems is severely threatened by several human-induced activities. Sustaining coastal fisheries therefore requires urgent, integrated governance approach focused on addressing these challenges and strengthening community-based management to ensure long-term resilience against increasing anthropogenic and climate pressures.

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Population Performance and Resilience in Polychaetes as Environmental Indicators of Estuarine Ecosystems

Isabel M. Smallegange, Lukas H.A. Edwards and Amy Attle

Abstract

Understanding how estuarine populations respond to environmental changes is crucial for estuarine ecosystem management. Life history theory, particularly the fast-slow and parity continuums, offers a framework to predict population performance and resilience. However, applying these continuums to estuarine, environmental indicator species like spionida polychaetes remain underexplored. We investigated the applicability of the fast-slow and parity continuums as trait-based indicators of performance and resilience in seven estuarine polychaete species (*Axiiothella serrata*, *Boccardia proboscidea*, *Dimorphilus gyrotilatus*—formerly *Dinophilus gyrotilatus*, *Ophryotrocha labronica*, *Polydora cornuta*, *Pygospio elegans*, and *Streblospio benedicti*). Using a Dynamic Energy Budget Integral Projection Model (DEB-IPM), we quantified eight life history traits across different feeding levels, reflecting environmental quality. Principal component analysis revealed that life history variation in polychaetes can be decomposed into two primary continuums: adult performance and growth. Species with higher adult performance exhibited increased population growth rates, while resilience was influenced by the interaction between adult performance and growth. Interestingly, these two continuums identified in estuarine polychaetes diverge from the traditionally recognised fast-slow and parity continuums. This divergence is likely because, in our framework, life history variation was driven by differences in environmental quality rather than resource allocation as in the fast-slow parity framework. Since polychaete presence and abundance are often used as indicators of environmental quality, our approach can therefore enhance the accuracy and effectiveness of environmental assessments for estuarine ecosystems. More generally, these findings highlight the importance of considering context-specific continuums and mechanistic models in life history analyses to better predict population responses to environmental changes.

Keywords: dynamic energy budget integral projection model, elasticity analysis, evolutionary demography, matrix modelling, morphometric similarity

1. Introduction

Estuaries are dynamic and productive ecosystems that provide critical services such as nutrient cycling, habitat for diverse species, and protection from storm surges [1, 2]. However, estuarine ecosystems are also some of the most heavily used and threatened natural systems globally [3–5], and their deterioration due to human activities is intense and increasing [6]. Understanding how populations within these ecosystems respond to environmental changes is thus crucial for effective management and conservation [7].

Life history theory provides tools for understanding how populations of organisms respond to environmental changes. One particularly effective application of this theory is structuring the wide diversity in life history traits across plants and animals along two main axes: the “fast-slow” continuum [8, 9] and the “parity” continuum [10, 11]. The fast-slow continuum ranks species from fast-growing, short-lived organisms with high fecundity at one extreme to slow-growing, long-lived species with low fecundity at the other [10, 12, 13]. The parity continuum ranks reproductive strategies, traditionally categorised as semelparity (reproducing once) and iteroparity (reproducing multiple times) on a spectrum rather than as distinct alternatives. For a wide range of species, their position along these two continuums predicts their population performance [12, 14], conservation status [15], and resilience to disturbances [12, 14]. As a result, especially the fast-slow continuum is increasingly used as a trait-based predictor for how species [16] and entire communities [17] respond to anthropogenic impacts.

While initially developed for terrestrial species, the fast-slow continuum is increasingly utilised in marine systems [18–20]. However, the extent to which the fast-slow continuum can be used as an environmental indicator for estuarine communities remains underexplored. Polychaetes, for example, are widely recognised as bioindicators in marine and estuarine ecosystems [21, 22], yet the explicit application of the fast-slow continuum to polychaetes within indicator frameworks is rare, even though their life history traits often align well with this continuum [23, 24]. What is more, the parity continuum has seldomly been applied to trait-based indicator frameworks across any taxa [25], including polychaetes. These gaps highlight the need to integrate trait-based approaches into polychaete-based environmental assessments to enhance our understanding and management of estuarine ecosystems.

In this chapter, we investigated to what extent the fast-slow and parity continuums can serve as trait-based indicators of the performance and resilience across varying levels of environmental quality for seven estuarine polychaetes: *Axiiothella serrata*, *Boccardia proboscidea*, *Dimorphilus gyrotilatus*—formerly *Dinophilus gyrotilatus*, *Ophryotrocha labronica*, *Polydora cornuta*, *Pygospio elegans*, and *Streblospio benedicti* (Table 1). To achieve this, we first captured each species’ full life history by calculating eight life history traits that represent schedules of survival, growth, and reproduction (Table 2) using a Dynamic Energy Budget Integral Projection Model (DEB-IPM) [11, 33] (Figure 1: objective 1a). A DEB-IPM is a demographic model that describes the mortality of juveniles and adults as well as how a species’ growth and reproduction depend on its energy uptake during feeding. The feeding level in a DEB-IPM is a proxy for environmental quality [19] and varies between zero (empty gut) and unity (full gut). We calculated the eight life history traits across a wide range of feeding levels to capture polychaete performance across environmental qualities. A fraction of the energy acquired during feeding is allocated to maintenance and

Species	Lifespan	Reproduction	Larval type	Indicator role	Ref.
<i>Axiiothella serrata</i>	Years (likely)	Parental care, brooding or gelatinous	Lecithotrophic	Stable, less disturbed sediments	[26]
<i>Boccardia proboscidea</i>	Months	Multiple, poecilogonous	Adelphophagic/ Planktotrophic	Opportunist, disturbed habitats	[27]
<i>Dimorphilus gyrotilatus</i>	Weeks	Multiple	Direct	Microhabitat specialist, rapid turnover	[28]
<i>Ophryotrocha labronica</i>	Weeks to months	Multiple	Direct	Rapid coloniser, enriched sites	[29]
<i>Polydora cornuta</i>	Months	Multiple, poecilogonous	Adelphophagic/ planktotrophic	Opportunist, disturbed habitats	[30]
<i>Pygospio elegans</i>	Months to year	Multiple, plastic, poecilogonous	Adelphophagic/ planktotrophic	Early coloniser, disturbed sites	[31]
<i>Streblospio benedicti</i>	Months	Multiple, poecilogonous	Planktotrophic / lecithotrophic	Opportunist, disturbed habitats	[32]

Reproduction modes: Annual, iteroparous (*reproduces once per year, with individuals capable of multiple reproductive cycles over their lifespan*); multiple (*capable of several reproductive events per year or over the lifespan*); plastic (*flexible reproductive strategy, can switch between sexual and asexual reproduction or vary reproductive output depending on environmental conditions*); Parental care, brooding (*adults invest in offspring by protecting or brooding eggs/larvae, often within tubes or burrows*) or poecilogonous (*single species produces multiple larval types, e.g., both planktotrophic and lecithotrophic larvae*). Larval types: adelphophagic (*larvae ingest unfertilised nurse eggs with which they share a cocoon; usually have a shorter pelagic phase and more limited dispersal*); lecithotrophic (*larvae develop directly from egg masses and do not rely on external food sources during their early stages*); planktotrophic (*larvae feed in the plankton, typically have a longer pelagic phase, and can disperse over greater distances*); direct (*no free-swimming larval stage; juveniles develop directly into benthic forms, usually with limited dispersal*).

Table 1.
 Life history characteristics of the seven polychaete species in this study and their indicator role.

Life history trait	Symbol	Trait informs on	Definition	Equation
Generation time	T	Turnover	Number of days required for the individuals of a population to be fully replaced by new ones.	$T = \frac{\log(R_0)}{\log(\lambda)}$
Age at maturity	L_a	Longevity	Number of days that it takes an average individual in the population to become reproductive.	$L_a = \eta_{L_b}$
Progressive growth	γ	Growth	Mean probability of growing to a larger length across the length domain W .	$\gamma = \sum_i^m \bar{G}_{i,j} \Big _{i < j}$
Retrogressive growth	ρ	Growth	Mean probability of growing to a smaller length across the length domain W .	$\rho = \sum_i^m \bar{G}_{i,j} \Big _{i > j}$
Mean recruitment success	φ	Reproduction	Mean per-capita number of recruits across the length domain W .	$\varphi = \sum_i^m \bar{V}_{i,j}$
Degree of iteroparity	S	Reproduction	Coefficient of variation in age at reproduction.	$S = \sqrt{V_f} / \bar{a}_f$
Net reproductive rate	R_0	Reproduction	Mean number of recruits produced during the mean life expectancy of an individual in the population.	$R_0 = \sum_{x=0}^{x=\eta_e} l_x m_x$

Life history trait	Symbol	Trait informs on	Definition	Equation
Mature life expectancy	L_m	Reproduction	Number of days from the mean age at maturity (L_a) until the mean life expectancy (η_e) of an individual in the population.	$L_m = \eta_{L_p}$

To calculate life history traits [11], we discretised the IPM by dividing the length domain Ω into very small-width discrete bins (we chose 200 bins), resulting in a matrix $\mathbf{A}$ of size 200×200 and which dominant eigenvalue equals λ_0 , the population growth rate. Mean lifetime reproductive success R_0 is the dominant eigenvalue of the matrix $\mathbf{F} = \mathbf{V}(\mathbf{I} - \mathbf{GS})^{-1}$, where $\mathbf{I}$ is the identity matrix and $\mathbf{V} = \mathbf{DR}$, with $\mathbf{D}$ as the parent-offspring association, $\mathbf{R}$ the reproduction, $\mathbf{G}$ the growth, and $\mathbf{S}$ the survival matrix; this gives generation time $T = \log(R_0)/\log(\lambda_0)$. The mean life expectancy, η_e , is calculated as $\eta_e = \mathbf{1}^T \mathbf{N}$, where $\mathbf{1}$ is a vector of ones of length m and $\mathbf{N}$ is the fundamental matrix $\mathbf{N} = (\mathbf{I} - \mathbf{S})^{-1}$. The longevity of an individual of length L is η_L , which means we can calculate age at sexual maturity $L_a = \eta_{L_b}$ and mature life expectancy $L_m = \eta_{L_p}$ so that $\eta_e = L_a + L_m$. The parameter l_x is the probability of surviving to age at least x , and m_x is the average fertility of age class x . $\bar{\mathbf{G}}$ is the mean of $\mathbf{G}$, $\bar{\mathbf{V}}$ is the mean of $\mathbf{V}$, and i and j are the row and column entries of the matrix, respectively. The vital rates included in progressive growth γ , retrogressive growth ρ , and sexual reproduction ϕ , were averaged across the columns j (the length bins), weighted by the relative contributions of each stage at stationary equilibrium. For example, to calculate mean sexual reproduction ϕ , we summed the values in the columns j of the $\mathbf{V}$ matrix and multiplied each ϕ_{ij} by the corresponding j th element w_j of the stable stage distribution $\mathbf{w}$, calculated as the right eigenvector of $\mathbf{A}$. Finally, the degree of iteroparity = $\sqrt{V_f}/\bar{a}_f$, with $V_f = \bar{a}_f^2 - (\bar{a}_f)^2$, where $\bar{a}_f^2 = \left[(\mathbf{c} \cdot \mathbf{c})^T \otimes \sum_{x=0}^{x=\eta_e} m_x \right] / \left[\mathbf{1}^T \otimes \sum_{x=0}^{x=\eta_e} m_x \right]$ and $\bar{a}_f = \left[\mathbf{c}^T \otimes \sum_{x=0}^{x=\eta_e} m_x \right] / \left[\mathbf{1}^T \otimes \sum_{x=0}^{x=\eta_e} m_x \right]$, with $\mathbf{c}^T = (1, 2, \bullet, \bullet, \bullet, p, \bullet)$.

Table 2.
Eight life history traits that inform on a species' turnover rate, longevity, growth and reproduction.

growth, while the remainder goes to reproduction. This energy allocation is implicitly represented in the model through a species' maximum length (L_m) and maximum reproduction rate (R_m) (**Figure 1**) [34] and is constant for each species, independent of feeding level. Although we cannot determine actual energy allocation values, we assume that the diverse life history traits of the polychaetes in our study (**Table 1**) capture a wide range of energy allocations. To complete objective 1, using the eight life history traits for all species across all feeding levels, we identified life history continuums by decomposing the total life history variance into significant principal component (PC) axes (life history continuum axes [12]) using principal component analysis (PCA) and discriminant analysis to identify any species clustering (**Figure 1**: objective 1b). For objective 2 (**Figure 1**), we tested whether the position of polychaete species along the life history continuums identified in objective 1 predicts population performance, resilience to disturbances, and sensitivity to perturbations. The insights gained from both objectives are crucial for predicting how polychaete populations will respond to environmental stressors such as estuarine acidification, pollution, and climate change. Because we examined individual development, trade-offs, and life history traits, we can underpin these insights with the demographic mechanisms that drive polychaete performance and resilience.

2. Methods

2.1 Dynamic energy budget integral projection model (DEB-IPM)

The Dynamic Energy Budget Integral Projection Model (DEB-IPM) is a cohort-based population model that simulates the life cycle of female individuals, incorporating the demographic processes of survival, growth, fecundity, and the size

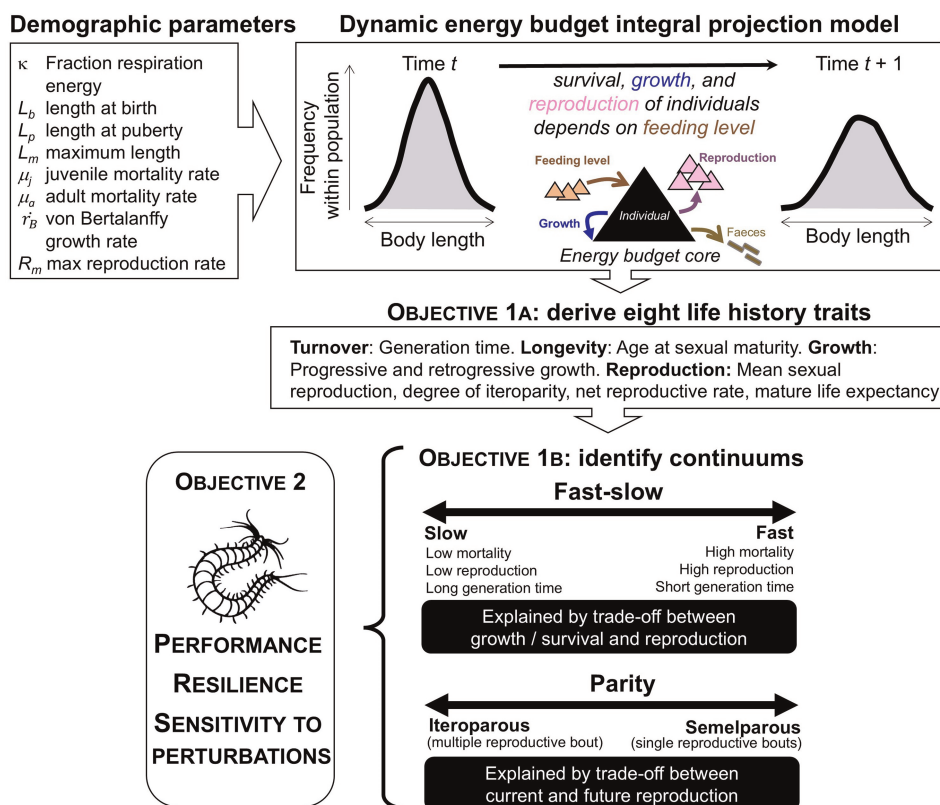


Figure 1. Workflow. Eight demographic parameter values (top-left box) are required to parameterise a DEB-IPM (top-right box). Once parameterised, we used each DEB-IPM to calculate eight derived life history traits (Table 2) across a wide range of feeding levels (objective 1a). We then decomposed each species' total life history variance into significant principal component (PC) axes, or life history continuums (objective 1b), to assess if either inform on polychaete population performance, resilience to disturbances, or sensitivity to perturbations (objective 2).

relationship between mothers and their offspring. It operates using eight key life history parameters (Figure 1; [11, 33]) and can incorporate environmentally induced variability in individual growth and reproductive output [33]. A summary of the model is included in the Appendix (Figshare repository [35]), while detailed guidance on parameterisation and implementation is available in Ref. [11].

In essence, the DEB-IPM's growth and reproduction functions are derived from the Kooijman-Metz model [34] (see Appendix in Figshare repository [35]), a simplified form of Kooijman's standard DEB theory [36]. This model captures the fundamental trade-off between growth and reproduction and meets the criteria for general explanatory models of individual energetics [36].

The Kooijman-Metz model assumes isomorphic growth, with individuals born at length L_b , maturing at length L_p and consuming food at a rate that is proportional to their feeding level $E(Y)$. This feeding level, which reflects environmental quality *via* gut fullness observations [37], ranges from nearly empty ($0 \leq E(Y) < 0.1$) to completely full ($0.9 \leq E(Y) \leq 1$) [37]. At the maximum feeding level ($E(Y) = 1$), individuals attain a maximum length L_m and reproduce at a peak rate R_m .

As individuals feed and uptake energy, a constant fraction κ of assimilated energy is allocated to respiration to cover maintenance costs and somatic growth.

The remaining fraction $1-\kappa$ of assimilated energy is allocated to reproduction in the case of adults and to the development of reproductive organs in the case of juveniles. Note that this role of κ in mediating the trade-off between growth versus reproduction is implicit: R_m is a composite parameter that includes $(1-\kappa)$, whereas L_m is a composite parameter that includes κ [34]. Importantly, both R_m and L_m are empirically derived from data rather than computed directly from their constituent components. The parameter κ only enters a DEB-IPM explicitly to define the starvation condition, where individuals of length L die when $L > L_m \cdot E(Y)/\kappa$ (see Appendix in Figshare repository [35]). Only in the starvation condition does κ function as a direct input parameter (**Figure 1**), influencing survival thresholds (rather than being inferred from composite traits). If an individual survives from year t to year $t + 1$ (juveniles have a different mortality rate μ_j , than adults, μ_a) she grows in length following a von Bertalanffy growth curve characterised by $\dot{r}_B$; the von Bertalanffy growth rate. If a surviving female is an adult, she produces offspring at a maximum rate of R_m if she experiences the maximum feeding level of $E(Y) = 1$.

2.2 Parameterisation of the DEB-IPMs

Parameter estimates were taken directly from the literature (**Table 3** and references therein). We could not find estimates for the energy allocated to respiration, κ , and instead followed common practice and assumed that $\kappa = 0.8$ as in the generalised animal (**Table 3**) [36]. Mortality rates are all estimates of natural mortality rates.

Species	κ (–)	L_b (cm)	L_p (cm)	L_m (cm)	μ_j (d ⁻¹)	μ_a (d ⁻¹)	$\dot{r}_B$ (d ⁻¹)	R_m (# d ⁻¹)
<i>Axiobella serrata</i>	0.8	0.050 (6)	0.680 (6)	1.000 (7)	0.0040 (7)	0.0040 (7)	0.015 (7)	22 (7)
<i>Boccardia proboscidea</i>	0.8	0.04 (8)	0.130 (8)	4.2710 (8)	0.1054 (9)	0.0035 (8)	0.8405 (8)	101 (10)
<i>Dimorphilus gyrociliatus</i>	0.8	0.0010 (11)	0.0080	0.0120 (11)	0.0042 (11)	0.0585 (11)	0.1311 (11)	2 (11)
<i>Ophryotrocha labronica</i>	0.8	0.0500 (7)	0.180 (7)	0.4500 (12)	0.0187 (13,14)	0.0026 (12)	0.0290 (12)	33 (12)
<i>Polydora cornuta</i>	0.8	0.05 (15)	1.000 (16,17)	3.200 (18)	0.2171 (19)	0.0480 (19)	0.0159 (15)	42 (19)
<i>Pygospio elegans</i>	0.8	0.0399 (20)	0.359 (20)	1.500 (21)	0.0300 (22)	0.0625 (23)	0.0016 (24)	1 (25)
<i>Streblospio benedicti</i>	0.8	0.0150 (26)	0.450 (26)	0.9400 (26)	0.0090 (26)	0.0055 (26)	0.1840 (27)	8 (28)

References: (6) [39]; (7) [26]; (8) [40]; (9) [41]; (10) [42]; (11) [43]; (12) [44]; (13) [45]; (14) [46]; (15) [47]; (16) [48]; (17) [49]; (18) [50]; (19) [51]; (20) [52]; (21) [53]; (22) [38]; (23) [54]; (24) [55]; (25) [56]; (26) [57]; (27) [58]; (28) [59].

Table 3.

Demographic model parameters for each of the eight polychaete species. Parameters are: the fraction of assimilated energy allocated to respiration, κ , length at birth, L_b , length at puberty, L_p , maximum length, L_m , juvenile mortality rate, μ_j , adult mortality rate, μ_a , von Bertalanffy growth rate, $\dot{r}_B$, and maximum reproduction rate, R_m ; all with units in brackets below. Numbers in brackets are references (see footnotes). To avoid eviction [38], we set $\dot{r}_B = 0.015$ instead of 0.059 d^{-1} for *A. serrata*, $L_b = 0.04 \text{ cm}$ (instead of 0.135 cm) for *B. proboscidea*, $L_b = 0.05 \text{ cm}$ (instead of 0.0249 cm) for *P. cornuta*, $\dot{r}_B = 0.005 \text{ d}^{-1}$ (instead of 0.0016 d^{-1}) for *P. elegans*.

2.3 Identifying life history continuums (objective 1)

We extracted eight key life history traits based on survival, growth, and reproductive schedules (**Table 2**), following the established methodologies [12, 13, 15]. These calculations were performed in MATLAB Version 9.12 (The MathWorks Inc., 2022) using the implementation provided by Smallegange and Lucas [11]. To address objective 2 (**Figure 1**), we also calculated the population growth rate, denoted as λ_0 . This was achieved by discretising the DEB-IPM (see Appendix in Figshare repository [35]: Eq. (1)), partitioning the continuous length domain Ω into 200 narrow intervals. This discretisation yielded a square matrix **A** of dimensions 200×200 , which dominant eigenvalue corresponds to λ_0 . Additionally for objective 2, we quantified resilience to disturbances using the damping ratio ξ , defined as $\xi = \lambda_0/|\lambda_2|$, where λ_2 is the second largest eigenvalue of matrix **A** [60]. While other resilience indicators exist—such as simulated recovery time or transient dynamics [61]—we opted for the damping ratio to facilitate comparison with prior studies.

For each species, we calculated the eight derived life history traits, population growth rate, and damping ratio across a range of feeding levels ranging from $E(Y) = 0.6$ to $E(Y) = 1$ in increments of 0.01. These values represent a gradient from low to high, respectively [19]. Feeding levels below $E(Y) = 0.6$ were not considered, as they were insufficient to support individual growth and reproduction in many species. Specifically, at low-feeding levels, individuals may shrink once their length exceeds $L > L_m E(Y)$, potentially preventing them from reaching their maturation size (L_p). Moreover, individuals are assumed to die of starvation when their length exceeds $L > L_m E(Y)/\kappa$ [33]. Notably, when $E(Y) = \kappa$, individuals die of starvation at their maximum length. For most species, we set $\kappa = 0.8$ [11]), meaning that at feeding levels significantly below 0.8, individuals die at lengths well below their maximum size. These constraints collectively result in a declining population growth rate and increased extinction risk as feeding levels decrease.

To summarise the life history variation across feeding levels for each species, we performed a principal component analysis (PCA) on the calculated life history traits using the *pca* function in R [62]. Prior to analysis, all traits were standardised (mean = 0, standard deviation = 1) to meet PCA assumptions. We evaluated the significance of the principal components using Kaiser's criterion, retaining only those axes with eigenvalues greater than one [63].

To investigate the separation of species based on their life history variation, we performed a linear discriminant analysis (LDA) on the scaled life history traits of all species combined. The LDA model was fitted using the *lda* function from the MASS package in R [62]. We calculated Wilks' lambda for each discriminant function to assess their significance: Wilks' lambda values close to zero indicate strong separation, while values close to unity indicate weak separation. The significance of each discriminant function was tested by converting Wilks' lambda to an F-statistic and calculating the corresponding p-values. Because LDA showed no clustering of species (see "Results" section), we proceeded to identify life history continuums for all populations of all species using a PCA with and without a body length correction on the combined dataset of life history traits. We corrected for body length by calculating the residuals from linear models fitted between the scaled life history traits and the ultimate length (L_∞) at each feeding level ($L_\infty = L_m \times E(Y)$) of the corresponding species [10, 64, 65]. The length-corrected PCA was performed on the length-corrected and standardised residuals as before. We could not perform a phylogenetically-corrected PCA as no phylogenetic tree of these polychaetes is

available, but the length correction will capture at least some of the phylogenetic signal [66].

2.4 Linking life history continuums to performance, resilience, and sensitivity to change (objective 2)

We used general linear mixed models (GLMMs) to test if a species' position along a continuum predicts its population performance (λ_0), resilience to disturbances (ξ), and what part of its life history is most sensitive to change. We tested if the PC scores of significant principal components (life history continuums—continuous variables) and their two-way interactions impact population performance (λ_0) and resilience to disturbances (ξ), with species identity included as a random factor and feeding level as a continuous covariate. Although PCA assumes that principal components are uncorrelated, this assumption applies to the variance structure of the original data, and the components themselves are not necessarily statistically independent. To assess whether combinations of principal components offer additional explanatory power beyond their individual effects, we tested for interactions between them in relation to our response variables.

To investigate whether a species' position along a life history continuum predicts its sensitivity to specific DEB-IPM demographic parameters, we first assessed the elasticity of the population growth rate λ_0 to perturbation in the demographic parameters (**Figure 1**). Using MATLAB (Version 9.12; The MathWorks Inc. 2022), we perturbed each parameter by 1% and calculated the resulting elasticity of λ_0 . The parameters tested included length at birth L_b , length at puberty L_p , maximum length L_m , juvenile mortality rate μ_j , adult mortality rate μ_a , von Bertalanffy growth rate $\dot{r}_B$, and maximum reproduction rate R_m (**Figure 1**). We excluded κ , as it cannot be perturbed directly (aside from its explicit role in the starvation condition, κ is implicitly embedded in the model through its influence on L_m (proportional to κ) and R_m (proportional to $[1-\kappa]$) [34]). To test whether species' positions along the principal component axes predicted which demographic parameter their population growth rate was most sensitive to, we used ordinal regression. Principal component scores were treated as continuous predictors and species identity was included as a random effect.

All GLMMs and ordinal regression analyses were conducted in R version 4.3.2 [62] using RStudio [67]. For each GLMM, we verified model assumptions—including normally distributed errors, homoscedasticity, and absence of multicollinearity—by inspecting diagnostic plots and error structures in R [62, 68, 69]. All datasets generated and analysed, along with the MATLAB and R code used, are available in our Figshare repository [35].

3. Results

3.1 Identifying life history continuums (objective 1)

In our first exploration of the structuring of life history variation across species, we found that for most polychaetes, two separate PC axes significantly structured their life history variation across feeding levels (**Table 4**). However, only one PC axis significantly structured the life history variation of *B. proboscidea* and *O. labronica* (**Table 4**).

Life history trait	<i>Axiiothella serrata</i>		<i>Boccardia proboscidea</i>	<i>Dimorphilus gyrocaliatus</i>		<i>Polydora cornuta</i>	
	PC1	PC2	PC1	PC1	PC2	PC1	PC2
Generation time	-0.38	-0.21	0.36	-0.38	0.12	-0.39	0.02
Age at maturity	0.34	-0.25	0.35	-0.38	0.11	0.35	0.39
Progressive growth	0.37	0.22	0.37	0.38	0.01	-0.36	0.19
Retrogressive growth	-0.36	-0.40	-0.33	-0.37	0.15	-0.35	0.42
Mean recruitment success	0.37	-0.30	0.37	0.37	0.20	0.37	0.26
Degree of iteroparity	0.36	-0.35	0.35	0.36	0.30	0.39	0.07
Net reproductive rate	0.36	-0.38	0.35	0.37	0.26	0.36	0.34
Mature life expectancy	0.28	0.57	0.35	-0.16	0.87	0.24	-0.67
Eigenvalues	$\lambda_K = 2.51$	$\lambda_K = 1.12$	$\lambda_K = 2.67$	$\lambda_K = 2.62$	$\lambda_K = 1.03$	$\lambda_K = 2.56$	$\lambda_K = 1.07$
% variance	78%	16%	89%	86%	13%	82%	14%
Correlation coefficient between PC1 and feeding level	0.88		0.98	0.92		0.98	

Life history trait	<i>Pygospio elegans</i>		<i>Ophryotrocha labronica</i>	<i>Streblospio benedicti</i>	
	PC1	PC2	PC1	PC1	PC2
Generation time	-0.34	0.41	-0.36	-0.04	0.81
Age at maturity	-0.41	-0.17	0.36	-0.38	0.24
Progressive growth	0.42	0.17	-0.36	-0.38	-0.24
Retrogressive growth	-0.16	0.84	0.29	0.35	0.33
Mean recruitment success	0.42	0.17	0.37	-0.39	-0.07
Degree of iteroparity	0.42	-0.04	0.37	-0.39	-0.13
Net reproductive rate	0.41	0.19	0.35	-0.37	0.19
Mature life expectancy	NA	NA	0.35	-0.38	0.23
Eigenvalues	$\lambda_K = 2.35$	$\lambda_K = 1.05$	$\lambda_K = 2.71$	$\lambda_K = 2.53$	$\lambda_K = 1.19$
% variance	79%	16%	92%	80%	18%
Correlation coefficient between PC1 and feeding level	0.96		0.99	-0.99	

Table 4.
 Life history traits and their loadings onto the principal components (PC) axes. Only those PC axes were retained which eigenvalues $\lambda_K > 1$ (Kaiser criterion [63]). In bold are loadings higher than 0.35 or lower than -0.35, and assumed to contribute to their PC axis.

For all species, PC1 explained most life history variation (ranging from 79% for *P. elegans* to 89% for *B. proboscidea*), and nearly all eight life history traits (Table 2) were loaded onto PC1 (Table 4). We first considered life history traits with loadings higher than 0.35 or lower than -0.35 to contribute to the variation captured by the corresponding PC, bearing in mind that PC1 is significantly correlated with feeding level (correlation coefficients are higher than 0.88: Table 4). For all species, three life

history traits associated with reproduction—degree of iteroparity, net reproductive rate, and mean recruitment success—all increased with increasing PC1 scores (feeding level) (**Table 4**), indicating an increase in performance as environmental quality increases. Likewise, in all species, retrogressive growth (indicating low environmental quality) decreased with increasing feeding level (**Table 4**). Progressive growth (indicating high environmental quality), in turn, increased with increasing feeding level for all species bar *P. cornuta* and *O. labronica* (**Table 4**). For those species where generation time loaded onto PC1 (**Table 4**), it decreased with increasing feeding level for all bar *B. proboscidea*. For those species where mature life expectancy loaded onto PC1, it increased with increasing feeding level. All these relationships support the notion that polychaete life history performance increases with increasing feeding level. Somewhat counterintuitively, though, as feeding level increased, age at maturity increased for all species, bar *D. gyrotiliatus*, and *P. elegans*.

PC2 explained limited life history variation for some species (ranging from 13% for *D. gyrotiliatus* to 18% for *S. benedicti*). In *A. serrata*, *D. gyrotiliatus*, and *P. cornuta*, mature life expectancy is loaded onto PC2 (**Table 4**). In *P. cornuta* and *P. elegans*, retrogressive growth was loaded onto PC2 (**Table 4**). Finally, generation time was loaded onto PC2 for *S. benedicti* and *P. elegans* and progressive growth was loaded onto PC2 for *P. cornuta* (**Table 4**).

To assess if species can be separated according to their life history variation, we applied linear discriminant analysis. The first two discriminant functions (LD1 and LD2) explained most of the life history variation across species (63 and 25%, respectively). However, they had high Wilks' lambda values ($\lambda_W = 0.99$ in both cases) with non-significant associated p-values (LD1: $F_{6,323} = 0.099$; $p = 0.999$; LD2: $F_{6,323} = 0.999$; $p = 0.998$), indicating weak separation between species. The lack of a clear clustering of species based on the life history traits analysed thus motivated the decision to perform a single PCA for all populations of all species to capture the overall life history variation. To do so, we performed a PCA with and without a body length correction on the combined dataset of life history traits. Because the length-corrected loadings on l-PC2 and l-PC3 differed qualitatively from those in the uncorrected analysis (i.e., traits loaded onto different axes), and l-PC4 was no longer significant (**Table 5**), we proceeded with the length-corrected results. The length-corrected PCA (l-PCA) revealed that three PC axes structured polychaete life history variation, together explaining 72% of the total variance (**Table 5**: l-PCA). l-PC1 was strongly and positively associated with age at maturity, net reproductive rate, and mature life expectancy—traits linked to adult performance (survival and reproduction). Thus, as l-PC1 scores increase, species exhibit higher reproductive output and adult survival (**Figure 2A**). l-PC2 showed a strong positive loading for progressive growth and a strong negative loading for retrogressive growth. Increasing l-PC2 scores indicate greater allocation to growth and development (**Figure 2A**). l-PC3 was positively associated with generation time, indicating that higher l-PC3 scores correspond to longer generation turnover (**Figure 2B**).

3.2 Life history continuums as predictors of performance, resilience, and sensitivity to change (objective 2)

Objective 1 revealed that the first three l-PC were significant (**Table 5**). We thus specified a GLMM with l-PC1, l-PC2, l-PC3, their two-way interactions, and feeding level as a covariate to test their effects on the log-transformed population growth rate, with species identity as a random term. We observed high partial correlations between l-PC2 and feeding level (partial correlation = -0.660), between l-PC2 and the

Life history trait	PC1	PC2	PC3	PC4
Generation time	−0.08	0.02	−0.03	0.92
Age at maturity	0.53	−0.29	−0.17	0.05
Progressive growth	−0.12	0.16	−0.65	−0.31
Retrogressive growth	−0.19	−0.36	0.51	−0.22
Mean recruitment success	0.28	0.44	0.48	−0.09
Degree of iteroparity	0.08	0.70	−0.04	0.03
Net reproductive rate	0.54	0.11	0.13	−0.04
Mature life expectancy	0.54	−0.25	−0.18	0.03
Eigenvalues	$\lambda_K = 1.68$	$\lambda_K = 1.34$	$\lambda_K = 1.20$	$\lambda_K = 1.04$
% variance	35%	22%	18%	13%
Correlation coefficient between PC and feeding level	−0.35	0.08	−0.26	−0.28
Life history trait	l-PC1	l-PC2	l-PC3	
Generation time	−0.07	0.09	0.73	
Age at maturity	0.53	0.16	−0.11	
Progressive growth	−0.18	0.51	−0.41	
Retrogressive growth	−0.09	−0.59	−0.15	
Mean recruitment success	0.34	−0.37	0.27	
Degree of iteroparity	−0.07	0.43	0.43	
Net reproductive rate	0.52	0.03	0.04	
Mature life expectancy	0.53	0.19	−0.10	
Eigenvalues	$\lambda_K = 1.75$	$\lambda_K = 1.36$	$\lambda_K = 1.08$	
% variance	38%	23%	11%	
Correlation coefficient between PC1 and feeding level	0.36	0.31	−0.28	

Table 5.
Life history traits across all species and their loadings onto the principal components (PC) axes (top) and length-corrected l-PC axes (bottom). Only those PC axes were retained which eigenvalues $\lambda_K > 1$ (Kaiser criterion [63]). In bold are loadings higher than 0.50 or lower than −0.50 that load significantly onto their PC axis. Four PC axes are retained for each PCA as their eigenvalues were higher than unity ($\lambda_K > 1$).

interaction between l-PC2 and l-PC3 (partial correlation 0.795) and between l-PC3 and the interaction between l-PC1 and l-PC3 (partial correlation 0.933). We therefore removed feeding level and the latter two-way interactions from the full model. The reduced model revealed a significant interaction between l-PC1 scores (adult performance) and l-PC2 scores (growth) in predicting log-transformed population growth rate λ_0 ($t = -8.606$, $p < 0.001$). Specifically, the population growth rate increased with higher l-PC1 scores, indicating that species with greater adult survival and reproductive output had higher rates. However, this positive effect was more pronounced in species with lower l-PC2 scores (i.e., lower growth) than in those with higher l-PC2 scores (**Figure 3A**). l-PC1 scores on their own significantly impacted population growth rate ($t = -25.225$, $p < 0.001$), as did l-PC2 scores ($t = 2.576$, $p = 0.011$). l-PC3 scores did not significantly affect the log-transformed population growth rate ($t = 1.429$, $p = 0.154$).

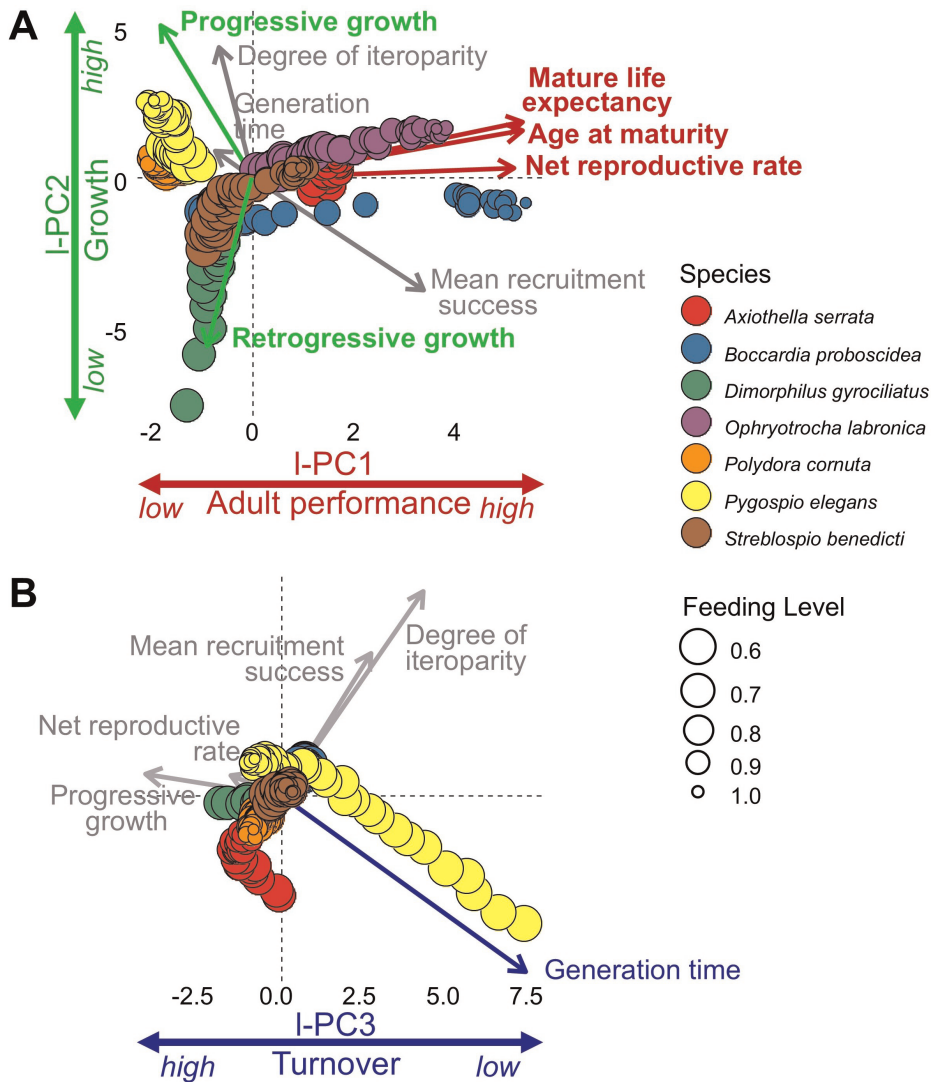


Figure 2.

Length-corrected, l-PCA plots including populations of seven polychaete species across feeding levels E(Y) that ranged from 0.6 (largest dots) to 1 (smallest dots) (see main text). Coloured symbols denote species (see legend); note reverse relationship between symbol size and feeding level. Labelled single-headed arrows correspond to the life history traits (Table 2) used in the l-PCA analysis and show the magnitude and direction of the loadings (Table 5). In (A), red arrows significantly align with the adult performance axis (l-PC1), and green arrows significantly align with the growth axis (l-PC2). In (B), blue arrows significantly align with the turnover axis (l-PC3). Grey arrows in (A) and (B) do not significantly align with any l-PC axis. Note that some non-significant arrows in (B) are covered by the symbols. A jitter was added.

Next, we specified a GLMM with l-PC1, l-PC2, l-PC3, their two-way interactions, and feeding level as a covariate to test their effects on the damping ratio as a measure of resilience to disturbance, with species identity as a random term. We observed high partial correlations between l-PC2 and feeding level (partial correlation = -0.660),

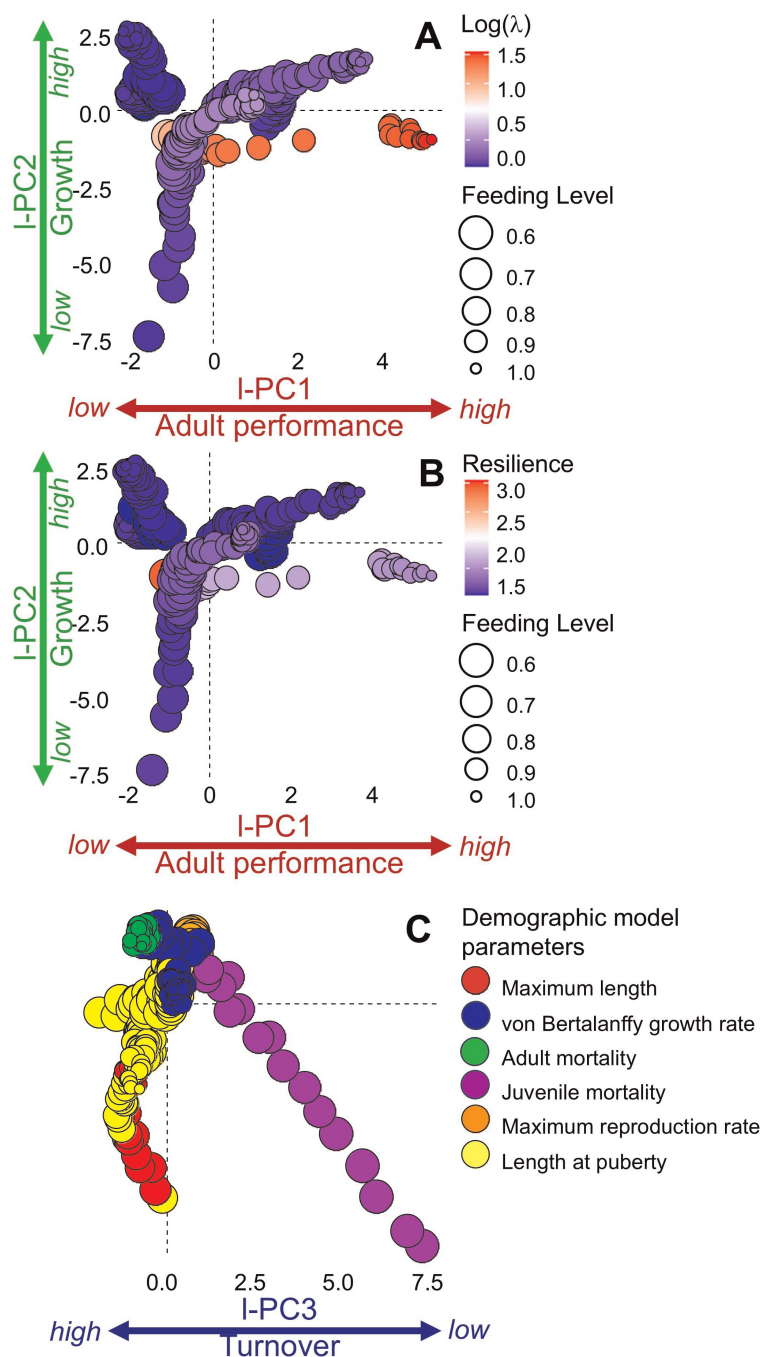


Figure 3. Overlays on the I-PCA figure that includes populations of seven polychaete species across feeding levels $E(Y)$ that ranged from 0.6 (largest dots) to 1 (smallest dots) for (A) log-transformed population growth rate λ_0 , (B) resilience to disturbance, and (C) the results of the elasticity analysis showing for each population which demographic model parameter (Table 3) had the largest impact on the population growth rate. A jitter was added.

between l-PC2 and the interaction between l-PC2 and l-PC3 (partial correlation 0.795), and between l-PC3 and the interaction between l-PC1 and l-PC3 (partial correlation 0.933). We therefore removed feeding level and the latter two-way interactions from the full model. The reduced model revealed that resilience was significantly affected by the interaction between l-PC1 scores (adult performance) and l-PC2 scores (growth) ($t = 7.257, p < 0.001$). Resilience decreased with increasing l-PC1 scores (increasing adult performance) but only for moderately negative l-PC2 scores (lower growth) associated with *B. proboscidea*; otherwise, resilience was low (Figure 3B). l-PC1 scores on their own significantly impacted population growth rate ($t = -12.226, p < 0.001$), as did l-PC2 scores ($t = 8.286, p < 0.001$). l-PC3 scores did not significantly affect resilience ($t = 0.488, p = 0.626$).

Finally, we found that l-PC3 (generation time) had a significant positive impact on the ranking of the DEB-IPM demographic parameters that a population's population growth rate was most sensitive to $t = 5.369, p < 0.001$. At the lowest generation times, population growth rates are most sensitive to perturbation of maximum length and length at puberty. As generation time increased, populations became more sensitive to perturbation of adult mortality, von Bertalanffy growth rate, and maximum reproduction rate (Figure 3C). At the highest generation times, length is only most sensitive to perturbation of juvenile mortality (Figure 3C). Neither l-PC1 ($t = -1.417, p = 0.156$) nor l-PC2 score ($t = -0.916, p = 0.359$) significantly affected the ranking of the DEB-IPM demographic parameters that a population's population growth rate was most sensitive to.

4. Discussion

The aim of this study was to investigate to what extent life history continuums can be applied as trait-based indicators of population performance and resilience in estuarine polychaetes. By employing a DEB-IPM, we quantified eight life history traits across varying feeding levels, reflecting environmental quality, for seven polychaete species. Our analyses revealed that the total life history variation could be effectively decomposed into an adult performance and a growth continuum that significantly predicted population growth rate and resilience to disturbance across all species. These findings underscore the potential of integrating trait-based approaches into environmental assessments to enhance the management and conservation of estuarine ecosystems.

The effectiveness of trait-based approaches is highly dependent on the quality and availability of life history data. If life history data like growth and adult performance are sparse or lacking, which is often the case for species in marine and estuarine environments [70–72], scholars frequently resort to using the maximum intrinsic rate of population increase, $r_{\max}$, as a proxy for resilience [73–76], assuming that species with higher $r_{\max}$ values have higher resilience (cf. [71, 75, 77]). The intrinsic rate of population increase (r) is mathematically equivalent to the log-transformed population growth rate (i.e., $\log(\lambda) = r$). However, our results show that species with the highest population growth rates do not necessarily exhibit the greatest resilience to disturbance. For example, while *S. benedicti* demonstrated a positive relationship between population growth rate and resilience, *B. proboscidea* populations with the highest growth rates displayed only intermediate resilience. This underscores the need for caution when using summary metrics such as $r_{\max}$ to predict species' resilience (cf. [19]). Despite these differences in resilience, *S. benedicti* tolerates broad salinity ranges

and organic pollutants [78], and *B. proboscidea* thrives in polluted areas [79]. Both species recover quickly after disturbances through fast recolonisation [80] rather than local population growth. Nevertheless, fast colonisation requires high adult performance, aligning with our findings and enhancing confidence in the application of trait-based approaches to predict polychaete population growth.

Generation time is considered an important indicator trait in polychaetes for assessing population performance and environmental impacts in toxicological tests due to its sensitivity to sublethal stress and its role in rapid population turnover [81]. If we link the outcomes of toxicological studies to demographic results, we can unravel how generation time can be applied in polychaete trait-based approaches. For instance, toxicological tests have shown that pollutants like heavy metals and pesticides can alter life history traits, including generation time, growth, and reproduction [82, 83]. By comparing these results with demographic studies, we can assess whether the impacts observed in controlled environments align with those in natural populations. We did find a generation time (turnover) continuum, but it only explained 11% of the total life history variation, after the two main continuums of adult performance and growth. Moreover, generation time did not map onto population growth rate or resilience. If the demographic model used in this study accurately represents polychaete life history trajectories, generation time may not be the optimal trait to focus on in a trait-based approach. However, we did find that generation time predicted which demographic model parameter a population's growth rate was most sensitive to. Thus, while generation time is a key factor in determining the demographic parameters that population growth rate is most sensitive to, it does not predict population growth rate itself, or resilience to disturbances. It is therefore crucial to consider multiple traits and their interactions in trait-based approaches to effectively predict population responses to environmental changes.

The identification of adult performance and growth continuums as the primary axes of life history variation in estuarine polychaetes is a novel finding that diverges from the traditionally recognised fast-slow and parity continuums [10, 12, 84]. The fast-slow continuum reflects the trade-off between reproduction versus growth and survival [84, 85], whereas the parity continuum reflects the trade-off between current and future reproduction [10, 12]. Life history variation along these continuums is thus a result of how populations or species trade-off their resource investment along either continuum. Here, we created life history variation by varying feeding level as a proxy of environmental quality. This means that populations differed in their resource or energy acquisition but not in how they allocated their energy, as this was kept constant in our demographic model. As a result, the life history performance of each individual species, as assessed in the species-specific PCAs, increased with increasing feeding level (which was highly correlated with each first principal component axis, onto which most life history traits loaded). Because the loadings in each of these PCAs, when squared, add up to unity, we can express the proportion of each species' total life history variation associated with growth versus reproduction traits (Table 2). This confirmed a negative relationship between traits associated with growth versus reproduction (correlation coefficient: -0.40), but it was non-significant ($t = -0.986$, $p = 0.377$) (see Appendix in Figshare repository [35]: Table A1). It explains why we did not find any clustering of species in the 2D life history space defined by adult performance and growth. But it does raise the question of which life history continuum framework would be most appropriate to apply as a trait-based indicator framework for polychaetes in estuarine ecosystems? The absence or presence of certain polychaete species, or their abundance, is frequently used to

indicate environmental degradation or pollution [21, 22, 81]. In those cases, a framework centred on polychaete performance over an environmental quality gradient, like we presented here, can provide a robust framework for assessing estuarine systems compared to the traditional fast-slow and parity ones that are based on variation in how species allocate energy.

Finally, some of the polychaetes in this study, including *P. elegans*, *S. benedicti*, *B. proboscidea*, and *P. cornuta*, exhibit poecilogonous development, demonstrating more than one developmental mode (**Table 1**) [27, 86, 87]. Such poecilogonous development allows these species to adapt to varying environmental conditions by switching between different developmental modes that provide multiple pathways for survival and reproduction, enhancing their performance and resilience [27, 86, 87]. For instance, larvae that develop directly by ingesting nurse eggs may have a higher survival rate in stable environments, while planktotrophic larvae may disperse more widely in disturbed habitats, facilitating recolonisation and population recovery. Here, we did not include poecilogony in our modelling approach. In matrix projection models, juvenile development time can be varied (e.g., [88]), and integral projection models can include plasticity *via* trait-environment functions (e.g., [89]). However, both modelling approaches assume fixed developmental rates or transitions, and we still lack a fully integrated, development-centric model of population dynamics [90]. Yet the fact that the main life history continuum of adult performance comprises net reproductive rate suggests that incorporating poecilogony, or developmental plasticity more generally, into trait-based indicator frameworks can provide a more comprehensive understanding of polychaete population dynamics and resilience.

5. Conclusion

The findings of this study highlight the importance of considering context-specific continuums and mechanistic models in life history analyses to better understand and predict population responses to environmental changes. Given the unique life history continuums identified here for estuarine polychaetes, composite traits such as adult performance and growth emerge as robust indicators of their performance and resilience across a gradient of environmental quality. By integrating these (composite) traits into a trait-based indicator framework, we can enhance the accuracy and effectiveness of environmental assessments for estuarine ecosystems that typically focus on the presence, abundance, and community structure of polychaete species. However, caution should be exercised if resilience is inferred from population performance when (composite) trait data are not available. Future research should continue to explore and validate the indicators identified in this study across a wider range of polychaete taxa and environmental contexts to ensure their broad applicability and reliability if life history variation across species is also structured by variation in energy allocation.

Conflict of interest

The authors declare no conflict of interest.

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Chapter 4

Living Links: Coastal Vegetated Ecosystems in UK Estuaries as Blue Carbon Sinks and Food Web Foundations

Mariana do Amaral Camara Lima

Abstract

This chapter examines the ecological and climate regulation functions of coastal vegetated ecosystems (CVEs), with a particular focus on seagrass meadows and saltmarshes within UK estuaries. Drawing on examples from Solent and Mersey, it explores how these habitats support biodiversity and store carbon while also considering the pressures they face from nutrient enrichment, sea level rise and habitat fragmentation. By incorporating methods such as stable isotope analysis and remote sensing, the chapter demonstrates how long-term monitoring can reveal patterns in food web dynamics and carbon cycling. It also considers how recent initiatives, including the development of blue carbon policy tools like the UK Saltmarsh Code, are shaping restoration and management strategies. Rather than presenting CVEs solely as carbon stores, the chapter situates them within a broader framework of estuarine resilience, ecological connectivity and adaptive coastal planning. In doing so, it provides a practical and integrated perspective on the role of these systems in responding to environmental change.

Keywords: estuarine ecology, coastal vegetated ecosystems, blue carbon, seagrass and saltmarsh restoration, climate adaptation

1. Introduction

Coastal vegetated ecosystems (CVEs) encompass a range of habitats shaped by regular, seasonal or episodic inundations and are highly responsive to changes in adjacent terrestrial and marine environments [1]. These ecosystems, primarily saltmarshes, seagrasses and mangroves, occupy the dynamic interface between land and sea and are widely recognised for their ecological significance and multifunctionality. In estuarine contexts, which are the focus of this chapter, CVEs serve as ecotonal buffers that host distinctive communities of salt-tolerant and inundation-adapted plant species. Their structural complexity and functional diversity underpin a broad

suite of ecosystem services, supporting both biodiversity conservation and socio-economic well-being [2]. Among their many benefits, CVEs contribute to nutrient cycling, water filtration, coastal protection and the regulation of flood risk [3]. Of growing interest is their role in carbon storage (blue carbon). These habitats can not only fix carbon through photosynthesis but also retain substantial organic carbon within their sediments, often more efficiently than many terrestrial ecosystems [4]. As a result, they are increasingly valued as nature-based solutions for climate mitigation and have become central to coastal carbon policy initiatives at local, national and global levels [5]. In parallel to their biogeochemical roles, CVEs are also critical components of estuarine food webs. Their high primary productivity, ability to retain organic matter and physical structure support a diverse array of trophic interactions [6]. Numerous studies have linked the spatial extent of estuarine CVEs with local fishery yields, owing to their function as biogeochemical hotspots and their provision of structurally complex habitats that facilitate foraging, reproduction and refuge from predation [7]. Hence, the ecological and economic importance of CVEs extends well beyond their boundaries. These habitats support livelihoods, cultural practices and recreational activities while contributing to shipping, aquaculture and broader coastal economies [8].

CVEs are widespread across European estuaries, particularly along the Atlantic coast. The United Kingdom (UK) alone accounts for roughly one-quarter of the estuarine area in north-western Europe, with vegetated ecosystems distributed across more than 90 estuaries [9]. Within these systems, saltmarshes and seagrass meadows provide critical breeding and feeding grounds for a variety of species, including many flatfish and European seabass (*Dicentrarchus labrax*), highlighting their role in both conservation and food production. The UK's estuaries also play an integral part in supporting global bird migration, as part of the East Atlantic Flyway, a vital migration route for birds, connecting various parts of the world [10]. However, despite their importance to the environment and society, environmental fluctuations, sea level rise, human activity along the coastline and adjacent land management have resulted in significant habitat alteration/degradation of estuaries worldwide [11]. A recent review has found that approximately 2500 km² of intertidal habitat has been lost from estuaries across England and Wales since 1843, primarily because of human-induced pressures [11]. Moreover, rising Global Mean Sea Level (GMSL) could lead to a decline in intertidal zones, as sea-level rise may exceed the rate at which marshes accumulate sediment, reducing available habitat and potentially alter species distributions [12]. Thus, this chapter considers the dual ecological functions of CVEs, mainly saltmarshes and seagrass meadows, as they occur within UK estuaries. It focuses on two interlinked yet often separately studied roles: the contribution of these ecosystems to carbon sequestration (blue carbon) and their capacity to underpin estuarine food webs. Both functions are vital to the ecological integrity and long-term resilience of estuaries, particularly in temperate zones where vegetated habitats are extensive but increasingly under pressure.

Drawing on examples from the Solent and Mersey estuaries, the chapter highlights how CVEs mediate biogeochemical processes while simultaneously providing structural and trophic support to a diverse assemble of organisms. The structure of the chapter is as follows: First, we outline key mechanisms of blue carbon storage and their dependence on habitat condition. We then consider the food web dimensions of these systems, exploring how vegetation supports primary productivity and trophic linkages. Two case studies, one in the relatively well-monitored Solent region and one from the more industrially modified Mersey Estuary, offer grounded illustrations

of these processes in action. Finally, we examine the tools available for monitoring and managing these systems and reflect on policy opportunities to safeguard their ecological and climate-regulating functions.

2. Coastal vegetated ecosystems and blue carbon

The term blue carbon refers to the carbon stored in the plant biomass and, more significantly, the sediments of CVEs [13]. Although they occupy less than 3% of the global coastal area, these habitats are estimated to store up to half of the ocean's organic carbon [14]. CVEs waterlogged sediments act as long-term carbon sinks, with per-hectare sequestration rates estimated to be at least an order of magnitude greater than those of many terrestrial forest systems [15]. In addition to carbon fixed through photosynthesis, CVEs trap sediments and accumulate organic detritus, both from *in situ* vegetation and allochthonous inputs transported by tides, rivers and wave action [16].

The efficiency of carbon sequestration within CVEs varies according to several ecological and geomorphological factors. Sediment characteristics (including grain size and mineral content), vegetation type and local hydrodynamic regimes all influence carbon storage potential [17]. Seagrasses, for example, form dense root mats capable of trapping fine sediments and detritus, while saltmarshes are particularly effective in capturing suspended organic matter in the upper intertidal zone. As such, blue carbon dynamics are highly context-dependent, shaped by both biological processes and physical environmental conditions [18].

Recognition of the climate mitigation potential of CVEs has led to their inclusion in international policy frameworks such as the UNFCCC's Nationally Appropriate Mitigation Actions (NAMAs), REDD+ and LULUCF [19]. These habitats are also reflected in the UN Sustainable Development Goals, particularly SDG 14, which highlights the importance of ocean and coastal conservation [20]. Nevertheless, the amount and form of carbon stored can vary significantly between regions. For instance, Mediterranean seagrass meadows report average sediment carbon stocks of $372.4 \text{ Mg C ha}^{-1}$, considerably higher than those found in many temperate systems [7]. Conversely, temperate saltmarshes, especially in lower salinity environments, often exhibit greater carbon sediment burial rates than seagrass meadows within the same region [21]. These differences reflect the influence of sediment properties, dominant vegetation and tidal regime, reinforcing the need for region-specific assessments when identifying blue carbon hotspots and setting conservation priorities [22].

CVEs are, however, not only climate assets but also vulnerable carbon reservoirs. Anthropogenic pressures such as land reclamation, dredging, aquaculture conversion, upstream damming, nutrient loading and urban expansion threaten to disturb these systems [16]. Under such pressures, previously buried carbon may be re-exposed and released into the atmosphere as CO_2 , effectively reversing the carbon sink function [23]. Globally, between 0.23 and $2.25 \times 10^9 \text{ Mg}$ of CO_2 are estimated to be released annually as a result of coastal ecosystem degradation [4, 16]. At current rates of loss, projections suggest that up to 30–40% of saltmarshes and seagrass meadows, and nearly all remaining mangroves, could be lost within the next century [24].

As such, a deeper understanding of the biological, ecological and sedimentary processes underpinning carbon storage is essential to support the identification, protection and restoration of key blue carbon habitats, particularly in vulnerable estuarine systems. While blue carbon has emerged as a compelling justification for

the protection and restoration of CVEs, these systems are ecologically significant in broader ways. Their role in supporting food webs, through both direct and detrital pathways, links plant productivity to estuarine biodiversity and to wider ecosystem function. In the following section, we examine this trophic dimension, considering how CVEs support primary consumers and how these effects cascade across trophic levels.

3. Coastal vegetated ecosystems as food web foundations

CVEs are among the most productive marine and estuarine systems globally, particularly when compared with offshore or open ocean habitats [25]. These ecosystems benefit from nutrient-rich conditions and diverse inputs that support the rapid growth of primary producers and sustain high densities of juvenile fish and invertebrates during early-stage development [26]. Many of these juveniles subsequently form part of offshore food chains, functioning as prey in wider trophic relays that help transfer energy from coastal to marine systems [27].

At the landscape level, estuarine CVEs are closely interconnected, with materials and organisms moving between adjacent habitats through tidal action, river discharge and seasonal climatic variations [28]. These exchanges allow nutrients, detritus and biota to flow across ecological boundaries, contributing to productivity far beyond the habitat in which they originated. Traditionally, CVEs were understood primarily through their contribution to detrital food webs, often referred to as ‘detritus-based ecosystems’ [29]. This concept highlighted the breakdown of plant material and the subsequent support of secondary production. However, more recent research has shown that epiphytic and benthic microalgae can dominate primary production in some systems, suggesting that energy pathways are more varied than once believed [30]. Regardless of the dominant source, the foundational plant species within CVEs remain essential for maintaining energy flow and trophic structure [31].

CVEs in the UK, more specifically, provide critical habitat for a wide range of species, from planktonic organisms to larger vertebrates [32]. UK saltmarshes, for instance, support bird species such as redshank (*Tringa totanus*) and shelduck (*Tadorna tadorna*), using plant communities like glasswort (*Salicornia* spp.) and common reed (*Phragmites australis*) for feeding and nesting. Reedbeds in estuaries like the Humber, on the north-east coast of England, are known to host breeding populations of marsh harrier (*Circus aeruginosus*), bearded tit (*Panurus biarmicus*) and bittern (*Botaurus* spp.) [32]. Widespread eelgrass beds (*Zostera* spp.) offer important nursery and foraging habitat for species such as the short-snouted seahorse (*Hippocampus hippocampus*) and the spiny seahorse (*Hippocampus histrix*), as well as a variety of wrasse (family Labridae) [33]. These meadows also support diverse invertebrate assemblages of amphipods, polychaetes, bivalves and echinoderms, which play key roles in sediment turnover and nutrient cycling through bioturbation [34].

Hence, the condition of CVEs has a profound influence on the composition and complexity of local food webs [35]. Loss or degradation of key habitats, such as submerged aquatic vegetation, can reduce the availability of shelter and food for juvenile fish, ultimately affecting higher trophic levels, including piscivorous birds and larger fish species. Conversely, systems with greater habitat heterogeneity and connectivity tend to support more complex and resilient trophic structures [36]. Reduced water quality, fragmentation and physical disturbance can result in simplified food webs, with lower biodiversity and less energy transfer

across trophic levels [37]. For example, nutrient enrichment from agricultural runoff may lead to phytoplankton dominance and the displacement of benthic producers through eutrophication, altering base-level food sources and cascading through the ecosystem [38]. Thus, understanding the relationship between habitat conditions and food web dynamics is critical to effective conservation. Managing structural complexity and connectivity can help maintain ecosystem resilience and ensure the continued delivery of essential services, including biodiversity support and fisheries productivity.

To this purpose, a variety of analytical techniques are used to investigate food web structure and function in estuarine environments [39]. Among the most widely applied is stable isotope analysis, which offers insights into assimilated diets and trophic positioning over time [40]. Isotopic signatures of carbon ($\delta^{13}\text{C}$) can help distinguish between marine and terrestrial carbon sources, while nitrogen isotopes ($\delta^{15}\text{N}$) provide a measure of an organism's trophic level, with higher values indicating a higher position in the food web [41]. Meanwhile, gut content analysis remains a valuable tool for understanding recent feeding activity, offering high-resolution snapshots of prey consumed by fish and invertebrates [39]. While this method provides fine-scale dietary information, it is temporally limited and may not reflect assimilated diet over longer periods.

More recently, environmental DNA (eDNA) metabarcoding has been emerging as a transformative tool for food web and biodiversity assessments [42]. By extracting DNA fragments from water or sediment samples, researchers can detect a wide array of organisms present in an ecosystem, including rare or cryptic species [43]. When applied to trophic studies, eDNA can identify predator-prey relationships, monitor changes in species assemblages and track the spread of invasive species [44]. This method is increasingly adopted across aquatic systems, including estuaries, and has demonstrated performance on par with or exceeding that of traditional sampling techniques [45]. However, while eDNA offers considerable promise, its accuracy depends on the availability of comprehensive reference databases and careful interpretation of results [46]. Continued development of genomic libraries and methodological protocols will be key to maximising their potential in food web research and long-term monitoring of CVEs [47].

Although general principles offer a useful lens through which to understand blue carbon dynamics and food web structure, their real-world expression is context dependent. In this chapter, these will be further explored by the two site-specific case studies presented below. The first, covering the Solent Estuary, with its mosaic of saltmarsh and seagrass habitats, provides a useful case through which to examine how these functions interact in practice. This case study also allows for reflection on how anthropogenic pressures and targeted management responses are shaping CVEs in southern England.

4. Case study 1: The Solent estuaries - a coastal hotspot in Southern England

Location: South Coast of England, between the mainland and the Isle of Wight.

Type: Mesotidal estuarine strait.

Length: ~40 km - Width: 4 km (west) to 10 km (east).

Habitats: Saltmarsh, intertidal mudflats, coastal grazing marsh, seagrass beds.

Designations: 2 SACs, 3 SPAs (forming the Solent European Marine Sites).

This case study begins by examining Solent, one of Europe's most prominent sheltered channel systems. Located in the central English Channel, the Solent is a mesotidal estuarine strait that separates the south coast of mainland England from the Isle of Wight [48]. Its formation is linked to the end of the last Ice Age, when retreating glaciers flooded the former Solent River system, creating the current water body [49]. Characterised by shallow waters and low wave energy, the Solent extends roughly 40 km in length, with the strait narrowing to about 4 km in the west and widening to approximately 10 km at its eastern end [48].

The Solent is widely recognised as one of the UK's most ecologically significant coastal areas, comprising a mosaic of natural and modified landscapes that support rich biodiversity [50]. Tidal processes dominate the region, shaping a network of estuarine and coastal habitats including extensive intertidal mudflats, seagrass beds, sandy flats, scattered saltmarshes and valuable coastal grazing marshes. These diverse environments, found throughout areas such as Portsmouth, Langstone, Chichester and the Isle of Wight, are crucial for many species of national and international conservation importance [49]. Conservation initiatives such as the Solent Waders and Brent Goose Strategy, the Solent Seagrass Restoration Project and the Solent Oyster Restoration Project have been implemented to safeguard these habitats.

The region's complex hydrography and habitat variety underpin its designation as a key Marine Protected Area (MPA) within the UK. Notably, it includes two Special Areas of Conservation (SACs), the Solent and Isle of Wight Lagoons and the Solent Maritime, and three Special Protection Areas (SPAs): Solent and Southampton Water, Portsmouth Harbour, Chichester and Langstone Harbours. Together, these designations form the Solent European Marine Sites (EMS), offering formal protection under UK and EU conservation frameworks [51, 52]. Solent's strategic coastal position and strong transport connections have led to significant human development, including port infrastructure and growing urban centres. Population growth, combined with seasonal tourism, has intensified human activity in the region [50]. These pressures are compounded by land subsidence and rising sea levels, which amplify the impacts of global environmental change on sensitive habitats such as seagrass meadows [53].

Sediment transport dynamics also shape the region's coastal morphology. In areas from the central Solent eastward to Chichester Harbour, finer materials such as mud and sand tend to accumulate in deeper channels, particularly entering from the eastern end [54]. Within the sheltered estuaries and harbours, wide intertidal mudflats are common, with saltmarsh plants colonising higher elevations [55]. While the Solent's extensive subtidal sediments represent the largest carbon stocks overall, CVEs, particularly saltmarshes and seagrass meadows, are notable for their high rates of carbon accumulation per unit area, making them important contributors to carbon sequestration in the region [22]. For example, intertidal *Zostera* spp. meadows in the Solent store considerable amounts of carbon in their sediments, with levels comparable to other seagrass-rich systems globally (**Table 1**, [56]).

Recent estimates have suggested an average sediment carbon stock of $103.12 \pm 71.45 \text{ Mg C ha}^{-1}$ across seagrass habitats in the Solent (**Table 1**, [56]). This varies considerably by sediment type, ranging from $20.76 \pm 2.72 \text{ Mg C ha}^{-1}$ in sandy areas along the Isle of Wight to $119.92 \pm 10.00 \text{ Mg C ha}^{-1}$ in muddier sediments in Chichester Harbour (**Table 1**, [56]). These patterns reflect broader trends, with carbon storage potential strongly correlated with sediment texture and particle size, where finer, muddier sediments typically retain higher amounts of organic carbon (**Table 1**, [17]). The average carbon sequestration rate for seagrass meadows in the Solent has been reported at $75.12 \text{ g m}^{-2} \text{ yr}^{-1}$, a figure

Feature	Solent Seagrass	Solent Saltmarsh	Mersey Saltmarsh	UK Saltmarsh (General)
Carbon Stock (Mg C ha ⁻¹)	Avg: 103.12 ± 71.45 Range: 20.76 ± 2.72 (sandy) to 119.92 ± 10.00 (muddy)	Avg: 99.6 ± 2.98 Max: 101.2 ± 49.4	~115.5 (national avg); range: 22.4–405.1	65–74
Sequestration Rate (g m ⁻² yr ⁻¹)	75.12	69.88 ± 11.50	Estimated ~110.9	120–150
Sediment/particle size	Finer sediments = higher C retention	Fine-grained; particle size influences retention	Predominantly silt; erosion-prone	Fine sediments associated with higher C
Total C storage/loss	Subtidal sediments hold largest stocks	69,030–78,588 Mg C; ~50% loss since 1946	—	—
Economic value (per ha/yr)	—	£34.56–£118.26 ~ £10 million projected by 2100	—	—
Carbon origin (%)	—	~66.1% terrestrial/autochthonous	—	Range: 10.1–99% (Devon–Scotland)
Carbon content (%)	—	Surface: 5.1–6.5% Deep: 2.7–3.0%	Similar to national average	Surface ~7.94% Deep ~2.96%
Restoration potential	—	—	62 ha = ~68.8 Mg C/year potential	—
Main functions	High carbon stock and active sequestration	Long-term carbon sink; shoreline stability	Blue carbon; flood defence	Similar benefits as above

Data highlight the variability in carbon storage and sequestration potential.

Table 1.
 Summary of carbon stocks, sequestration rates, sediment characteristics and ecosystem functions for seagrass meadows and saltmarshes in the Solent, Mersey, and across the UK.

well within the global range and consistent with other temperate seagrass systems (**Table 1**, [57]). These rates appear to exceed those recorded for regional salt-marshes, which average $69.88 \pm 11.50 \text{ g m}^{-2} \text{ yr}^{-1}$, and fall below the reported UK national mean of $104.61 \pm 9.61 \text{ g m}^{-2} \text{ yr}^{-1}$ (**Table 1**, [58]).

Despite this, saltmarshes remain significant long-term carbon sinks. Nationally, UK saltmarshes are estimated to store between 65 and 74 Mg C ha⁻¹ (**Table 1**, [59]), with Solent saltmarshes aligning closely, showing average sediment carbon stocks of $99.6 \pm 2.98 \text{ Mg C ha}^{-1}$ and reaching up to $101.2 \pm 49.4 \text{ Mg C ha}^{-1}$ at sites such as Newton Harbour (**Table 1**, [60]). Current estimates suggest that remaining salt-marshes in the Solent collectively store between 69,030 and 78,588 Mg C, substantial figures, though markedly lower than historical levels [61]. For instance, in 1946, five major Solent sites, Chichester Harbour, Southampton Water, Langstone Harbour, Portsmouth Harbour and the Lymington Estuary, were estimated to hold approximately 157,738 Mg C in total, indicating a decline of over 50%, primarily due to habitat loss and fragmentation (**Table 1**, [61]). UK saltmarshes also demonstrate strong annual sequestration capacity, with rates ranging between 120 and 150 g m⁻² yr⁻¹ (**Table 1**, [59, 62]). In the Solent, more specifically, current sequestration is estimated at 1274.4 to 1593.0 Mg C annually (**Table 1**, [61]). The economic value of this service was estimated in 2009 at £34.56 to £118.26 ha⁻¹ yr⁻¹, suggesting an annual contribution of approximately £35,710 to £122,198 for the current extent (**Table 1**). If this area is maintained by 2100, the total estimated economic value of saltmarsh carbon sequestration could reach £10 million (**Table 1**, [63]).

In addition to quantity, the origin of the carbon stored in saltmarsh sediments is also ecologically relevant. Isotope mixing models suggest that an average of $66.1 \pm 15.0\%$ of organic carbon stored in UK saltmarshes originates from terrestrial or autochthonous (in situ) sources (**Table 1**, [58]). This proportion varies geographically from just 10.1% in the Arne marshes of Devon, where marine-derived material dominates, to nearly 99% in fjord-head systems such as the Kyle of Tongue and Loch Laich in Scotland (**Table 1**, [58]). These contrasts underscore the influence of hydrological and tidal regimes on carbon origin, transformation and retention potential. CVEs in the Solent also play a crucial role in the food web by providing nursery grounds and supporting various species. Seagrass meadow's dense vegetation provides shelter and ample food sources for many species, including commercially important ones like cod (*Gadus morhua*), plaice (*Pleuronectes platessa*) and whiting (*Merlangius merlangus*) [64]. Intertidal saltmarshes and mudflats also serve as nurseries, supporting a range of invertebrates and fish species. The range of elevations across saltmarsh habitats in the Solent creates important high-tide refuges for bird species that forage on adjacent mudflats [61]. These elevated areas allow birds to feed nearby while remaining close to safe roosting zones, protected from tidal flooding [61]. This is particularly vital for the 25 wildfowl species found in the Solent, some of which appear on the amber and red lists of conservation concerns. Among the birds that rely on saltmarsh vegetation for winter grazing are the black-tailed godwit (*Limosa limosa*), dark-bellied brent goose (*Branta bernicla bernicla*) and teal (*Anas crecca*) [61]. Notable bird species in the area include the common and roseate terns (*Sterna hirundo* and *Sterna dougallii*), the little tern (*Sternula albifrons*), the Mediterranean gull (*Ichthyaeetus melanocephalus*) and the sandwich tern (*Thalasseus sandvicensis*). In addition, seeds from annual halophytes like *Salicornia* spp. provide a critical food resource for various wildfowl [32]. The Solent saltmarshes also play a crucial ecological role as nurseries for juvenile fish, offering warm, shallow and complex habitats rich in food and relatively safe from predators [61]. Young non-predatory fish such as mullet (*Chelon* spp.) and herring

(*Clupea harengus*) thrive here, efficiently converting high-energy food into mobile biomass. This, in turn, supports larger predatory species and wading birds through a process known as the trophic relay [65]. As they grow, larger fish and nekton migrate to deeper creeks and nearby seagrass beds, linking saltmarsh productivity with the broader coastal ecosystem [66]. Although the precise contribution of saltmarshes in the Solent estuary to commercial fisheries is difficult to quantify due to the species' use of multiple habitats throughout their life cycles, modelling studies provide useful insight. One such study estimated that saltmarshes produce around 31.0 g of finfish $\text{m}^{-2} \text{year}^{-1}$, with approximately 12–20% of this biomass feeding piscivorous fish, 8–13% supporting fish-eating birds, and 18–29% exported across the estuary [67]. When applied to Solent, this equates to an estimated 320 tonnes of finfish produced annually, with a potential market value of £50.3 million, based on UK fisheries statistics [68]. However, this figure is likely a significant decline compared to 1946, reflecting the considerable loss of saltmarsh habitat over time.

Nevertheless, despite the presence of important CVEs, water quality concerns remain prominent in the Solent. Multiple studies report eutrophication risks across Langstone, Chichester and Portsmouth Harbours, as well as the Eastern Yar [52]. Nutrient enrichment, particularly from nitrates and phosphates, has been linked to harmful algal blooms and the proliferation of opportunistic benthic algae that can smother and degrade seagrass habitats [52]. As a result, these estuaries have been designated as nutrient-sensitive or polluted areas under the Nitrates Directive and the Urban Wastewater Treatment Directive [52, 69]. Meanwhile, the region is also facing increasing environmental pressures linked to climate change. A study by reference [70] reported that land in the Solent area is slowly sinking, while reference [71] estimated sea level rise at 4–5 mm per year since the early twentieth century, substantially higher than earlier figures of 1–2 mm per year [72]. Looking forward, projected sea level rise under climate change scenarios poses further risk to the integrity of CVEs in the Solent [73]. For this reason, recent conservation efforts in the Solent region have brought increased focus to the restoration and protection of saltmarshes and seagrass meadows, with two major initiatives, the *ReMEDIES* (Reducing and Mitigating Erosion and Disturbance Impacts affecting the Seabed) project and the *Solent Seascape Project*, playing leading roles (Table 2). A recent report commissioned by Natural England synthesised conservation actions under the ReMEDIES project in the Solent Maritime Special Area of Conservation (SAC) [74]. According to their report, while the mapped extent of seagrass in the Solent has increased slightly, likely due to improved survey methodologies, evidence of declining condition has been recorded in some areas (Table 2). Reductions in leaf length and density have also been observed in meadows around the Isle of Wight [74].

Working at a broader scale, the Solent Seascape Project (2022–2026) brings together conservation organisations, researchers and stakeholders to deliver seascape-scale ecosystem recovery across the region. Led by the Hampshire & Isle of Wight Wildlife Trust, the project integrates restoration efforts across multiple habitats, including seagrass meadows, saltmarshes, native oyster beds and seabird nesting sites (Table 2). As part of this work, the project has continued mapping and monitoring efforts for seagrass while promoting public engagement, citizen science and stronger policy integration [64]. It complements the targeted efforts of ReMEDIES by addressing cross-habitat interactions and building support for the long-term stewardship of Solent coastal ecosystems. Nevertheless, the combined efforts of ReMEDIES and the Solent Seascape Project demonstrate that coordinated restoration, grounded in ecological evidence and supported by public engagement, can contribute meaningfully to

Estuary	Habitat type	Key projects	Observed outcomes	Main challenges	Future goals
Solent	Seagrass	ReMEDIES (2019–); Solent Seascape (2022–)	New growth (e.g. Beaulieu); improved awareness; minor extent gains	Turbidity, recreation pressure, long-term viability	Scale-up planting; integrate citizen science
	Saltmarsh	Solent Dynamic Coast; Solent Seascape; historic SSSI	~50% loss since 1940s; stable carbon stocks in remaining sites	Coastal squeeze, sea-level rise, fragmentation	Support managed realignment; restore connectivity
Mersey	Saltmarsh	EA-led since 2000s; sediment recharge; monitoring	Recolonisation in areas like Eastham; increased biodiversity; improved sediment quality	Legacy pollution; gaps in long-term monitoring	Expand restoration zones; link to carbon funding
	Seagrass (low)	No active restoration yet; feasibility under review	Anecdotal presence; no confirmed meadows	High turbidity; lack of baseline ecological data	Map distribution; trial small-scale restoration efforts

Table 2. *Summary of recent and ongoing coastal vegetated ecosystem (CVE) restoration projects in the Solent and Mersey estuaries, highlighting the primary habitat types targeted, key interventions, observed outcomes, prevailing challenges and future restoration.*

the protection and recovery of seagrass ecosystems in a heavily used and ecologically complex estuarine system (**Table 2**).

While the Solent represents a relatively data-rich estuarine system, contrasting it with a more industrially influenced estuary provides an opportunity to examine how historical degradation affects ecological function [75]. The Mersey Estuary, discussed in the following case study, has been long subjected to heavy industrial use and has undergone extensive remediation efforts over the past four decades. This case study illustrates the ways in which restoration efforts influence carbon cycling and enhance biodiversity support in CVEs.

5. Case study 2: The Mersey estuary - an urban estuary of international importance

Location: Northwest England, from Liverpool Bay to Warrington and the Wirral Peninsula.

Type: Macrotidal estuary.

Length: ~113 km (entire river), estuarine section ~30 km.

Width at mouth: ~3 km.

Key Habitats: Intertidal mudflats, saltmarsh, freshwater marsh, reedbeds.

Designations: SSSI, SPA, Ramsar site.

Conservation Initiatives: Mersey Gateway Environmental Trust, former Mersey Basin Campaign.

The Mersey Estuary in Northwest England, UK, has a long and complex history of industrialisation, with major urban and industrial development along its banks, particularly between Liverpool and Warrington [76]. During the twentieth century, the river became heavily polluted by untreated industrial waste, prompting national concern. By the 1970s, the Mersey was widely regarded as one of the most polluted rivers in Europe [77]. Decades of recovery began with the launch of the Mersey Basin Campaign in 1985, a 25-year initiative aimed at improving water quality and restoring degraded ecosystems. This has since led to measurable improvements in dissolved oxygen levels, fish returns and the re-establishment of intertidal habitats. However, legacy pollutants and habitat loss remain key concerns [78].

The River Mersey is the principal freshwater source for the estuary, where it meets saline water from the Irish Sea to form a highly dynamic brackish system. The river's mean annual discharge is approximately 37 m³/s, with typical low flows occurring 95% of the time, dropping to around 9.4 m³/s [79]. These variable hydrological conditions shape sediment transport, nutrient cycling and the spatial structure of biological communities across the estuary. Geographically, the estuary is divided into four main sections, each with distinct ecological and hydrodynamic roles. The Upper Estuary, stretching from Warrington to Runcorn, narrows dramatically at the Runcorn Gap, a constriction shaped by underlying sandstone ridges. Downstream lies the Inner Estuary, where broad intertidal flats and developing saltmarshes dominate both shorelines. In this zone, sediment channels have historically migrated across the estuary, reshaping the landscape in response to tidal and sedimentary processes [80]. Seaward of this lies The Narrows, a deep, linear channel where powerful tides scour the bed, maintaining depths of up to 20 metres. Finally, the Outer Estuary opens into Liverpool Bay and is largely composed of extensive intertidal sandbanks [80].

Once biologically rich and home to Atlantic salmon (*Salmo salar*), European eel (*Anguilla anguilla*) and European smelt (*Osmerus eperlanus*), the Mersey suffered a dramatic ecological collapse during the height of industrial activity. By the mid-twentieth century, the estuary was deemed virtually lifeless due to decades of untreated pollution [77]. However, this trend began to reverse in the 1980s with the introduction of tougher environmental regulations and the Mersey Basin Campaign, which addressed sewage management, industrial discharge and public engagement [78]. Signs of ecological recovery have since emerged. Atlantic salmon were recorded returning to the Mersey by 2001, a key indicator of improved water quality, as the species requires clean, oxygen-rich conditions for upstream migration and spawning [81]. Similarly, the European eel, which travels more than 5000 km to spawn in the Sargasso Sea, was once abundant in the Mersey but declined sharply by the 1960s due to pollution and overfishing [82]. Recovery efforts for eel populations began in 2003, but the species remains vulnerable and under close observation. In addition, smelt populations, once prolific, were almost absent by the 1950s. A monitoring study between 2003 and 2007 recorded just three individuals, underscoring the fragility of this population [83]. Efforts to track fish populations in the Mersey have historically relied on methods such as electrofishing, fry netting and angler records, which, while informative, can be imprecise and intrusive [80]. More recently, habitat surveys have been used to identify key areas for spawning and juvenile recruitment, particularly for migratory species [84]. To improve data accuracy and reduce ecological disturbance, researchers are now trialling the use of eDNA monitoring as a non-invasive method capable of detecting species presence through genetic material in the water column [85].

Although water quality has improved dramatically since industrial times, historical contamination still affects the estuary. Heavy metals such as lead (Pb), zinc (Zn) and iron (Fe) remain detectable in sediment and biota, particularly in the upper reaches near Warrington [84]. These pollutants have long-term effects on invertebrate assemblages and may hinder recovery in more sensitive species. In addition to metal contamination, nutrient enrichment from surrounding agriculture and urban runoff contributes to algal growth and can reduce oxygen availability, affecting both benthic life and juvenile fish survival [86].

A recent field study conducted at Widnes has shown the potential of restored and remnant saltmarshes to support carbon sequestration. Sediment cores ($N = 15$) from three monitored transects over 3 years (2022–2024) showed carbon content ranging from 2.1 to 6.5% (analysed using an elemental analyser - Elementar Vario EL Cube – at the University of Salford, UK), with surface layers (0–5 cm) being most carbon enriched, ranging from 5.1 to 6.5%. Deeper layers (20–40 cm), in contrast, contained between 2.7 and 3.0% (**Table 1**). These values compare favourably with wider UK saltmarsh datasets and are consistent with national data, which estimate that UK saltmarshes store on average $115.5 \text{ Mg C ha}^{-1}$, with a reported range from 22.4 to $405.1 \text{ Mg C ha}^{-1}$ across 26 sites (**Table 1**, [60]). Scottish marshes show mean surface concentrations of 7.94% organic carbon, with deeper sediments averaging around 2.96%, closely matching values observed in the Mersey (**Table 1**, [60]).

While carbon burial rates for the Mersey have not been directly quantified, applying the UK mean sequestration rate of $1.11 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ (equivalent to $110.9 \text{ g C m}^{-2} \text{ yr}^{-1}$ [58]) suggests that restoring 62 hectares of saltmarsh could potentially sequester around 68.8 tonnes of carbon per year (**Table 1**). These values highlight the blue carbon potential of the Mersey's recovering marshes and underscore the importance of habitat protection and expansion in climate mitigation strategies. Moreover, particle size analyses conducted using a laser diffraction particle size analyser (LA 950v2) suggested that the dominant sediment type in the saltmarsh sediment cores was fine, poorly sorted silt. These results could raise concerns over erosion and carbon stock vulnerability during storm events or under sea-level rise scenarios. Analysis of particle size and carbon content showed a statistically significant relationship ($p < 0.01$), confirming that coarser, better-aerated strata tend to support greater carbon accumulation (**Table 1**). These findings provide a crucial baseline for blue carbon accounting in urban estuarine systems, supporting the case for nature-based flood defence strategies. Dense saltmarsh vegetation not only traps sediments and slows water flow but also contributes to wave attenuation and shoreline stability, critical functions in a low-lying estuary like the Mersey [32].

A recent field study, conducted in collaboration with the Mersey Gateway Environmental Trust, has explored macroinvertebrate communities at three sites within the Upper Mersey Estuary: West Bank in Widnes, Fiddler's Ferry near Penketh and Howley in Warrington [85]. Sampling was conducted during the full flood tide and part of the ebb to capture the tidal variability in invertebrate distribution. A plankton net with a $250 \mu\text{m}$ mesh size and a 250 mm diameter frame was used to collect aquatic invertebrates [87]. Following a 10-minute deployment, the net was hauled in, and the mesh filter located at the cod end was gently detached [88]. Spanning 5 weeks, the study took place between July 23 and August 22, 2024, encompassing a total of 12 sampling days. Results demonstrated how intertidal CVEs in the Mersey provide crucial habitat for a wide range of fauna, revealing over 477 individual organisms across key taxa, including *Neomysis*, *Crangon*, *Palaemon*, *Gammarus* and *Chironomidae*. These species form a vital part of the estuarine food web, supporting

higher trophic levels such as fish and birds [89]. Statistical analysis showed significantly higher Shannon diversity at Fiddler's Ferry and West Bank compared to Howley ($p < 0.05$), suggesting better ecological conditions in mid- and lower-estuarine zones along the Mersey (**Table 1**).

The estuary has also been known to support fisheries of non-resident species like Dover sole (*Microstomus pacificus*) and Atlantic Cod (*Gadus morhua*), and with large populations of wintering waders and migratory fish such as European eel (*Anguilla anguilla*) and smelt (*Osmerus eperlanus*) [90]. These habitats are home to species whose presence is well-established, leading to their formal designation as a Site of Special Scientific Interest (SSSI), a Special Protection Area (SPA) and a Ramsar site, designations that reflect both national (UK) and international conservation importance (**Table 2**) [91]. In addition to their ecological value, these areas support a wide range of local activities, including commercial and artisanal fishing, aquaculture (such as oyster farming) and leisure pursuits like birdwatching and paddleboarding, making them integral to the cultural and economic life of nearby communities [91]. Recently, the Mersey Estuary has also been considered by Natural England as a potential site for seagrass restoration (limited by the lack of up-to-date data) (**Table 2**) [92].

In summary, the Mersey Estuary is subject to ongoing management and restoration programs focused on improving water quality, enhancing habitats and adapting to climate change. These efforts involve collaboration between local authorities, environmental organisations and community groups, aiming to balance ecological health with human activities and development. For example, the Mersey Gateway Environmental Trust is the leading body involved in active restoration efforts along the estuary. While large-scale saltmarsh restoration has yet to be fully implemented, plans are underway to restore up to 62 hectares of intertidal habitat, including both saltmarsh and reedbed systems. These efforts aim to benefit key bird species, bolster biodiversity and enhance the estuary's role as a carbon sink (**Table 2**) [85].

Given the estuary's recovering fish populations, re-emerging vegetated habitats and role in climate mitigation, the Mersey is increasingly recognised as a high-value system for nature-based restoration. Long-term ecological monitoring and adaptive management remain essential to ensure that restoration success translates into measurable ecological function and climate resilience (**Table 2**). As restoration targets become increasingly ambitious, the role of monitoring and assessment tools, such as environmental DNA and remote sensing, that are improving our ability to track changes in carbon stocks and food web dynamics over time, becomes central to understanding success (**Table 2**). In the next section, we consider how management strategies and policy instruments are evolving to reflect the multifunctional value of these habitats.

6. Management, conservation and policy implications

Effective management of coastal vegetated ecosystems (CVEs) depends on long-term ecological monitoring to assess condition, detect degradation and support adaptive decision-making [93]. Monitoring tools such as stable isotope analysis have proven especially valuable for tracing carbon and nitrogen sources, identifying trophic linkages and understanding how organic matter and energy move between estuarine habitats [52, 94]. These methods provide a window into processes not captured by habitat mapping alone, revealing, for example, the

degree to which fish and invertebrates use seagrass beds, saltmarshes and adjacent mudflats at different life stages.

Despite their recognised importance, many UK CVEs remain in decline. Seagrass beds and saltmarshes are protected in national and EU policy frameworks [95], yet they continue to suffer from eutrophication, physical disturbance and poor water quality. Current monitoring efforts, typically based on six-year cycles, often emphasise spatial extent over ecological condition or functional resilience [93]. In this context, a stronger evidence base is urgently needed to assess habitat quality, carbon storage capacity and long-term ecosystem service delivery.

A growing area of promise lies in the integration of advanced monitoring technologies. In addition to isotope tracing, eDNA analysis now allows for rapid detection of species presence, trophic links and invasive species from water or sediment samples. This technique, coupled with next-generation sequencing (NGS), enables cost-effective biodiversity monitoring across multiple taxa and has been shown to outperform traditional sampling methods in some estuarine systems. Remote sensing, including satellite and drone-based tools, can complement *in situ* surveys by capturing spatial patterns of vegetated habitat change over time [96]. For blue carbon assessments, tools such as static flux chambers and time-series sediment sampling are also important for quantifying greenhouse gas emissions, especially methane (CH₄) and nitrous oxide (N₂O), which may increase under warmer, wetter conditions [7, 97].

In terms of policy, the UK's *Net Zero Strategy* identifies nature-based solutions as critical to meeting national climate goals. The emergence of blue carbon finance, which supports habitat restoration by generating tradable carbon credits, has begun to bridge conservation and climate agendas [98–100]. The UK Saltmarsh Code, developed by the Centre for Ecology & Hydrology, offers a structured approach to verifying carbon sequestration from restoration activities, establishing protocols for monitoring, permanence and compliance [101]. While initially focused on saltmarshes, the underlying principles can inform similar frameworks for seagrass, mangroves and other CVEs. Real-world applications of these frameworks are beginning to take shape. The Seagrass Ocean Rescue project, for example, has attempted large-scale seed-based restoration. Early trials revealed key challenges, such as seed loss on mobile substrates and tidal exposure, highlighting the importance of environmental thresholds and site-specific restoration strategies [102]. Adaptive management and long-term planning are essential to anticipate natural feedback such as storm impacts, grazing or altered salinity regimes and to prevent restored habitats from reverting to degraded states.

Crucially, restoration must not only target ecological metrics but also address social and economic dimensions. Projects in the Solent and Mersey estuaries have demonstrated how local communities benefit directly from CVE recovery through improved water quality, flood mitigation, fisheries enhancement, eco-tourism and public engagement. These outcomes are summarised in **Table 3**, which illustrates the multifunctional role of CVEs and their potential to contribute to regional well-being and climate resilience. Looking ahead, future priorities should include expanding baseline data on both organic and inorganic carbon stocks (e.g. calcium carbonate), quantifying greenhouse gas fluxes across seasons and developing resilience indicators linked to habitat condition and landscape connectivity. Restoration success must be measured not only by area restored but also by how effectively these systems function ecologically and integrate with policy, finance and local knowledge systems. In this sense, CVE management must be interdisciplinary, long-term and responsive, grounded in science, supported by communities and enabled by robust governance frameworks.

Region/project	Key benefits	Scale/outcomes	Stakeholder engagement
Solent – ReMEDIES & Solent Seascape	Seagrass restoration, behaviour change, eco-tourism, citizen science	Increased awareness, reduced anchoring impacts and early signs of habitat recovery	NGOs, local authorities, schools and volunteers
Mersey – CONVERSE & NFM projects	Natural flood mitigation, habitat recovery and community co-design	Flood risk reduction; increased saltmarsh coverage	Local residents, councils, EA and academic partners
Saltmarsh Code (UK-wide)	Carbon finance for habitat restoration: links to Net Zero policy	Formal certification framework; potential for carbon credit generation	UKCEH, policymakers and private investors
Seagrass Ocean Rescue (Pilot, UK)	Blue carbon restoration: public awareness	Large-scale seed trials: site-specific challenges highlighted	Conservation NGOs, researchers and local stakeholders
Ribble Estuary (Lancashire)	Coastal defence, wetland access, biodiversity benefits	£1 m saltmarsh creation; 140 properties protected	RSPB, NE, public-sector funders
Waldringfield (Deben Estuary, Suffolk)	Erosion control, fish nursery habitat, improved access	£750 k raised locally; 1 km polder restored	Local community groups, Environment Agency

These examples illustrate the multifunctional value of CVE recovery and its relevance to climate, biodiversity and regional planning objectives.

Table 3.
Socio-economic benefits from coastal vegetated ecosystem (CVE) restoration in the Solent and Mersey estuaries. Outcomes are summarised alongside principal stakeholder groups involved in design, delivery or stewardship.

7. Conclusions

Coastal vegetated ecosystems (CVEs), particularly saltmarshes and seagrass meadows, are critical components of UK estuaries. They serve a dual role as carbon sinks and foundational habitats that support complex food webs. Their ability to store carbon, attenuate wave energy, reduce flood risk and sustain biodiversity places them at the intersection of climate adaptation, conservation and coastal resilience. These ecological and biogeochemical functions are closely interlinked: intact plant communities support sediment stability and trophic interactions, which in turn reinforce long-term carbon storage. The case studies of the Solent and Mersey estuaries highlight how differing histories can still yield restoration potential when met with investment and long-term vision. The Solent, despite its high conservation status, remains vulnerable to eutrophication, disturbance and sea-level rise. However, integrated projects such as ReMEDIES and the Solent Seascape Project show that ecological recovery is possible through combined efforts in science, outreach and regional policy. Meanwhile, the Mersey, once a symbol of industrial degradation, is now a site of ecosystem regeneration, illustrating how monitoring, remediation and stakeholder collaboration can re-establish key estuarine functions.

Given the dynamic nature of estuarine systems, restoration must be adaptive. These landscapes are shaped by tidal movement, sediment transport and fresh-water inflows, all of which are shifting under climate change. Restoration should therefore focus on enhancing resilience, not replicating static past conditions. New tools, including stable isotope tracing, eDNA analysis and remote sensing, now offer unprecedented capacity to monitor food web dynamics, carbon cycling and habitat

change. Policy mechanisms such as the UK Saltmarsh Code provide promising pathways to align habitat restoration with climate finance. Embedding CVEs within broader Net Zero and biodiversity frameworks will be vital to scale up efforts and ensure that nature-based solutions are fully realised. Nonetheless, challenges remain. Poor water quality, habitat fragmentation and weak enforcement continue to constrain progress. Importantly, the rising interest in blue carbon must not overshadow the broader ecological and social value of CVEs. Effective policy and planning should account for carbon alongside biodiversity, fisheries and community use. In conclusion, CVEs are not just carbon stores or wildlife habitats; they are ecological infrastructure for resilience. Their protection must be central to future estuarine planning, supported by long-term, interdisciplinary action.

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Perspective Chapter: The Significance of Characterising Geologic and Hydrogeologic Settings of Aquifers for Submarine Groundwater Discharge and Seawater Intrusion Estimations

Haile Arefayne Shishaye

Abstract

The significance of geologic and hydrogeologic characterisations of aquifer settings seems to be overlooked in submarine groundwater discharge (SGD) estimates and seawater intrusion (SWI) evaluations, particularly in critical areas. The geological and hydrogeological context of an aquifer greatly affects the groundwater movement. Nonetheless, most SGD estimation approaches, including geophysical and geochemical tracers and water balance methods, fail to consider geological heterogeneities, even though aquifer geology and hydraulic conductivity can vary significantly. This variation indicates that SGD estimates might be inaccurate. Likewise, SWI assessments are unlikely to be dependable if the associated geology is not thoroughly understood. This brief review underscores the importance of incorporating geologic and hydrogeologic characterisations into SGD and SWI estimation techniques to minimise uncertainties and promote well-informed decisions and effective management strategies for coastal aquifers.

Keywords: geologic complexity, groundwater flow modelling, hydrologic processes, hydrogeology, submarine groundwater discharge, coastal aquifers

1. Introduction

The accuracy and precision of groundwater flow models ultimately rely on a thorough understanding of the geological settings of aquifers. Groundwater flow is mainly affected by the structural configurations of an aquifer. Geological factors such as compartments [1], heterogeneity [2, 3], and fault orientations significantly influence the groundwater movement. Therefore, a comprehensive understanding of geological

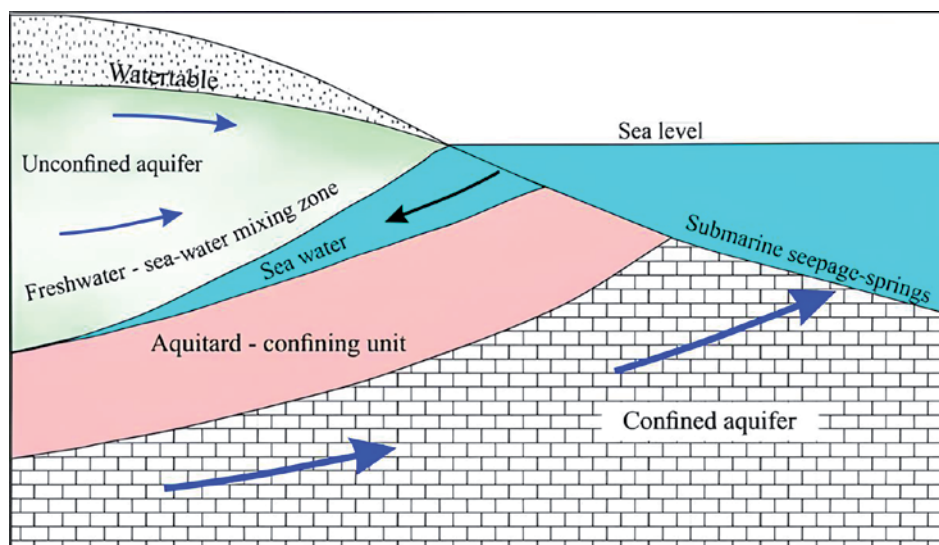


Figure 1.
Coastal aquifers, seawater-freshwater mixing and groundwater seepage at the sea floor (source: *Geological Digression*).

settings—including 3D lithological and stratigraphic mapping—is vital for accurately characterising the structural and lithological complexity of hydrogeological units and for understanding and quantifying groundwater flow in a region [4]. Tartarello et al. [5] used geological techniques to investigate the underground reservoirs in Sardinia (Italy), where they identified an alluvial plain overlaying low-grade metamorphic sandstones and carbonate rocks. They reported that the potential reservoir was a heterogeneous, dual-porosity and permeability system, represented by a matrix and a fracture network. Israde-Alcantara et al. [6] also employed geological methods to characterise the underground reservoirs within a municipal landfill and to evaluate the related environmental implications in Central Mexico, where they found that the site lacks impermeable subsurface layers.

The approach is equally vital in coastal aquifers ([7]; **Figure 1**). Souplos et al. [8] employed geological methods to characterise a coastal aquifer in north-western Crete (Greece) to develop a conceptual model prior to groundwater flow modelling. They identified two separate aquifer systems contaminated by seawater intrusion. Harbison and Cox [9] also used this approach to identify groundwater flow paths in the subtropical coastal plain in Pimpama (Queensland), stressing that geological insight is essential for estimating groundwater flow in coastal aquifers. In a study of hydrogeology and groundwater flow in a shallow coastal aquifer in southern Akwa Ibom State, Nigeria, Edet [10] pointed out that detailed geological characterisation is crucial for estimating SGD and seawater intrusion, as well as for understanding the hydrogeological settings of aquifers (**Figure 1**).

2. Hydrogeologic characterisation

The hydrogeology of an aquifer system is often studied by creating 3D geological models and developing geological histories [11, 12]. 3D geological models

enable direct assessments of the spatial distributions, stratifications, and volumes of lithologic units [2, 3]. This information is then used to define aquifer characteristics, including groundwater flow, potential, connectivity, and geometries [13, 14]. Identifying lithotypes at specific locations and aquifer connectivity can help characterise the possible groundwater flow paths [3]. Furthermore, hydrogeological characterisations also involve modelling groundwater flow directions using fault orientations and hydraulic gradients in fractured aquifers, as well as hydraulic gradient and permeability data in porous aquifers [15, 16]. This indicates that geologic and hydrogeologic methods can be employed for qualitative evaluations of SGD. It also emphasises that understanding geology and hydrogeology is crucial before undertaking any SGD estimations, as it offers a comprehensive view of how the aquifer system operates [17].

In a study evaluating the legacy and factors influencing groundwater nutrients and pesticides in an agriculturally impacted coastal Quaternary aquifer system in Southeastern Queensland, Shishaye et al. [1] identified potential preferential flow paths and perched systems through 3D geological models (**Figure 2**). These models clarified the variability in groundwater age, nutrients, and pesticides that could not be explained by the isotopes and geochemical tracers used. Accordingly, Shishaye et al. [1] emphasised that understanding both the geology and hydrogeology of an aquifer is essential before conducting monitoring or making conclusions about groundwater flow and contaminant transport based solely on isotopic and geochemical data. Similarly, making assumptions about SGD estimates and their variability without considering geological and hydrogeological differences could lead to the misleading outcomes. Without geological and hydrogeological characterisation to identify the groundwater flow paths and paleochannels, it becomes impossible to understand how the subsurface functions or where groundwater discharge originates. Consequently, any SGD estimates based on available techniques should be viewed as approximate, with the management decisions heavily reliant on the geological and hydrogeological context of the aquifer system.

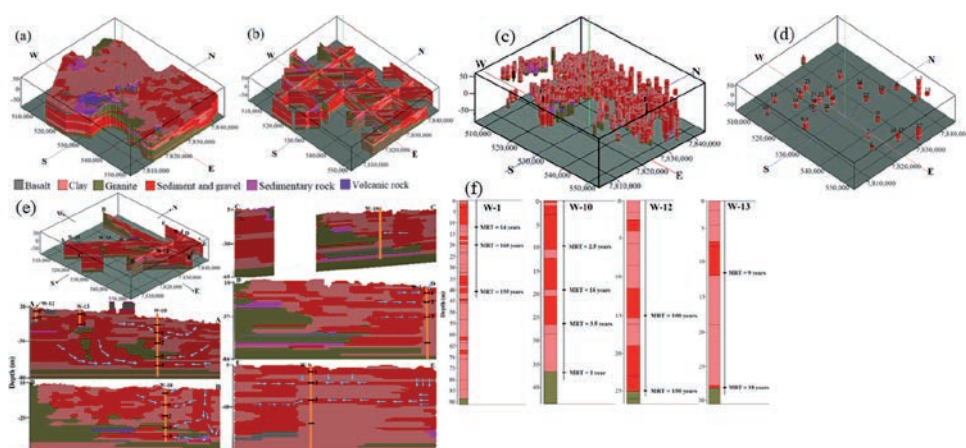


Figure 2.
 3D lithologic model of the Lower Burdekin catchment (Queensland, Australia) (a), geologic cross-sections of the 3D model (b), location and density of wells used to develop the 3D geologic model (c), drilled log data of sampled wells (d), geologic cross-sections of sampled wells (e), and drilled log data coupled with groundwater age of the sampled wells (f). This figure was adapted from Shishaye et al. [1].

In this figure, easting (lower right axis) and south (lower left axis) coordinates are in UTM-meters (WGS-841984, UTM zone 55), while elevation (upright axis) is in metres. The study showed that the geologic setting, characterised by preferential flow, caused younger groundwater to be detected below older groundwater (W-1 and W-10; **Figure 2**, where “w” stands for “well” and MRT indicates “mean residence time”). In a simple system, groundwater is replenished by rainfall and becomes older as it moves down through lithologic layers. However, the presence of clay and other aquitards in an aquifer system can slow groundwater movement and result in high groundwater retention. Clay and other compartmenting units can also cause perched or semi-confined aquifers that are partly isolated from the main aquifer system. This influences groundwater MRT in certain areas for extended periods, leading to older groundwater ages. Areas with younger groundwater detected below older groundwater mostly feature continuous sediment and gravel layers either vertically or laterally to the coastal environment (**Figure 2e**). The coastal borders of the Lower Burdekin catchment contain relatively higher sediment and gravel deposits than the parts further inland (**Figure 2**). These sediment and gravel-rich layers create preferential flow paths and lead to younger groundwater stored beneath older groundwater (groundwater that passes across clay-rich units). The arrowheads in **Figure 2e** indicate the possible directions of groundwater flow. The numbers (1–4) refer to sampled depths, while the additional hyphens indicate areas of potentially different groundwater ages. The arrows in sections C-C suggest that the most likely source of groundwater around W-10 is from the eastern direction, as intruded granites and clay units may impede groundwater flow from other directions. Similarly, unless the geologic settings of an aquifer system are well understood, any estimates of SGD obtained using various techniques may not alone be sufficient to inform monitoring programs, develop management strategies, or draw definitive conclusions about the sources of groundwater.

3. Limitations of SGD estimation techniques

One limitation of most SGD estimation techniques is their limited geological understanding, even though groundwater flow is influenced significantly by geology, as much as by climate and topography [3]. Geochemical tracers, for example, are among the commonly used methods for SGD estimation [18], but their effectiveness drops in heterogeneous geological settings [19]. Additionally, it is important to recognise that aquifers are never homogeneous; they only vary in degrees of heterogeneity [20]. Similarly, water budget approaches are often used to estimate SGD from coastal aquifers [21], but these methods usually assume a homogeneous aquifer system, despite the significant variation in geology and aquifer properties [22]. Therefore, understanding the hydraulic connections between coastal aquifers and the marine environment through detailed geologic and hydrogeologic characterisation is essential for accurate SGD estimates. This emphasises the need for a combined approach, starting with comprehensive geologic assessments, to improve accuracy and lessen the uncertainties linked to relying on single methods.

4. How and where can geologic and hydrogeologic characterisations be used?

The most crucial aspect of geologic and hydrogeologic characterisation is its ability to be applied at both local and regional levels using 2D and 3D assessment

methods [2, 3, 16]. Large-scale 3D hydrogeological models enable the evaluation of transboundary geological influences on groundwater flow and highlight the temporal impacts of human actions and climate change on groundwater resources and dependent ecosystems [23]. These models have been developed across various regions worldwide (e.g., [1–3, 16]). Different datasets can be used to construct 3D geological models of aquifers, including drill logs [2], geophysical survey data such as vertical electrical sounding (VES) [3, 16], and seismic profiling data [24]. Gathering drill log data can be expensive and time-consuming, especially over large areas, and accurately mapping subsurface geology requires a dense data set that captures geological variations. Therefore, deciding the optimal number of observations to create a representative subsurface geological map involves balancing data availability, drilling expenses, model uncertainties, and computational capacity. In this regard, combining data sets from drill logs and VES [3], along with seismic data [24], can help address this challenge.

Combining fault data with 3D geological modelling and interpolated water levels can be used to assess groundwater flow directions in geologically complex areas like fractured aquifer systems [16]. Fault networks and orientations are key factors influencing aquifer connectivity and groundwater movement in fractured aquifers. Fault data are often derived from satellite images and digital elevation models (DEM), which can be validated using fracture data from drill logs. Hassen et al. [2] employed an integrated approach combining a 3D geological model and fault orientations to determine the likely groundwater flow directions in the Kasserine aquifer system in central Tunisia, revealing high connectivity between geological compartments with flow directions aligned along faults rather than topographical gradients. Achtziger-Zupancic et al. [25] used similar methods to study groundwater flow in fractured basement rocks of the Ore Mountains in Germany, finding the significant aquifer permeability due to highly displaced faults. Shishaye et al. [16] combined 3D geological modelling with fault and fracture data in the Great Barrier Reef catchment, identifying possible groundwater flow directions along fault lines, regardless of the surface topography.

In areas with high-resolution seismic data sets, seismic profiling can serve as a standalone qualitative method for SGD evaluation, offering insights into the geological origins of groundwater flow pathways. Taniguchi et al. [24] observed that seismic profiling data show that submarine paleochannels, filled with permeable sediments and sealed by impermeable materials, create a hydrological link between coastal aquifers and offshore discharge locations. However, this method cannot quantify SGD rates unless supported by other quantitative techniques.

5. Conclusion

Geologic and hydrogeologic characterisations should act as benchmarks for any groundwater flow modelling, especially in SGD and SWI estimations. Without characterising the types, number, and nature of aquifers through such methods, and identifying detailed groundwater flow paths, both qualitative and quantitative assessments of SGD and SWI in coastal aquifers should be regarded as rough estimates from areas where subsurface settings and processes are not well understood. This underscores the importance of understanding both the geology and hydrogeology of a region before conducting monitoring studies or making management decisions based solely on different methods.

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Conflict of interest

There is no conflict of interest associated with this manuscript.

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Chapter 6

Changes and Evolution in Positive and Negative Storm Surges Recorded in the Río de la Plata Estuary from 1905 to 2024

María Florencia de Azkue and Enrique E. D'Onofrio

Abstract

One of the most relevant processes in the Río de la Plata involving atmosphere–ocean interaction is the storm surge. This chapter describes both positive and negative storm surges and how they have evolved over the past 120 years in the Río de la Plata estuary. In addition to analyzing changes in the annual number of events, their duration, and their height, the study also examines the contribution of rising mean river level—attributed to global warming—to these changes. Two scenarios are considered: (i) a real scenario, which includes the observed trend in mean river level, and (ii) a hypothetical scenario, in which this trend is removed. Annual trends in the aforementioned variables—number of events, event duration, and height—are presented, along with a seasonal analysis. The results show that trends in positive and negative storm surge variables do not directly follow the rising mean river level, even though the maximum height observations do. However, changes are observed in positive storm surges—such as a greater number of events in recent Southern Hemisphere springs (realistic scenario)—and in negative storm surges, including a deceleration in the number of events and an increasing trend in average duration when moving from the hypothetical to the realistic scenario. Furthermore, positive extreme events occur in the Southern Hemisphere winter, whereas negative ones take place in the Southern Hemisphere autumn.

Keywords: positive storm surge, negative storm surge, sudestada, rising mean river level, global warming, Río de la Plata estuary, climate change

1. Introduction

The Río de la Plata (RdP) is a large—300 km long—and shallow estuary—between 2 and 25 m deep—with a NW-SE orientation shared by Argentina and Uruguay (**Figure 1**). Along its shores and within its area of influence lie the capitals of both countries, Buenos Aires and Montevideo, as well as numerous ports, seaside resorts,

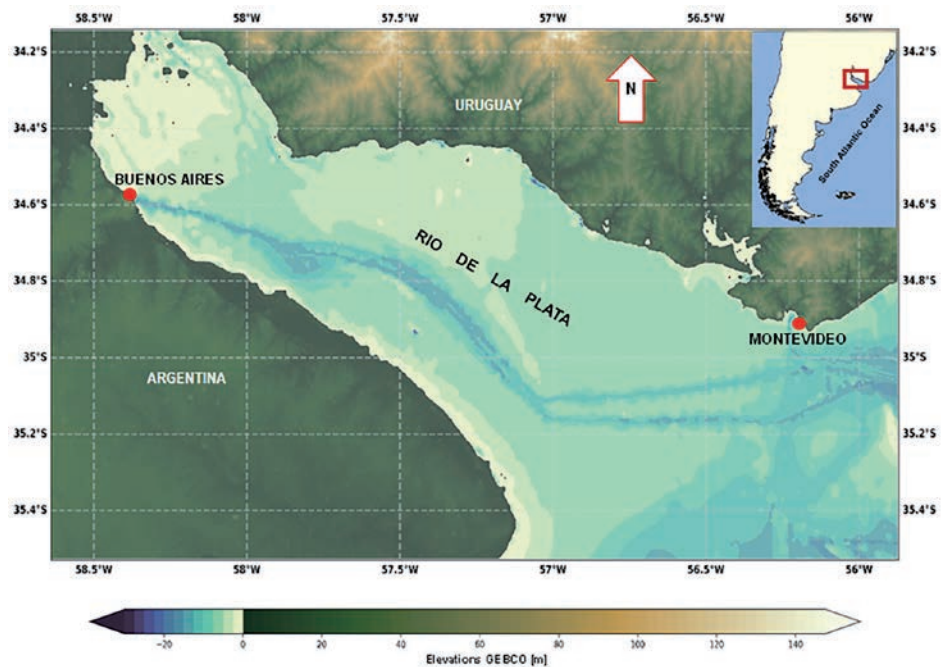


Figure 1.
Location and bathymetry of the Río de la Plata estuary [1].

and industrial centers. This estuary is formed by the confluence of the Paraná and Uruguay rivers, and it drains the second-largest river basin in South America, surpassed only by the Amazon River [2]. The RdP width increases from around 40 km in the upper section to about 220 km at its mouth in the Atlantic Ocean. The total estuarine surface area is 35,000 km².

The water level in the RdP is mainly influenced by a combination of a semidiurnal microtidal regimen with diurnal inequalities [3] and its wind-driven circulation. This circulation can be explained by two modes. In one of these modes, the winds have a dominant component perpendicular to the estuary axis (NE–SW), which is related to the inflow and outflow of water. In the other mode, the winds blow along the estuary axis (NW–SE). Both mechanisms can lead to a fall or rise in the water level in the upper part of the estuary [4, 5].

Thus, the mentioned meteorological conditions can cause differences between observed tides and astronomical tides. These anomalies are known as storm surges. Positive anomalies are associated with water levels higher than the predicted astronomical tide, while negative anomalies correspond to lower levels (**Figure 2**). Both can affect large coastal areas over periods that may extend from some hours to several days.

The estuary is located in one of the most cyclogenic regions of the Southern Hemisphere due to the presence of atmospheric waves travelling along the subtropical latitudes of the South Pacific and South America [6]. Numerical simulations conducted in Ref. [7] indicate that sea level fluctuations in the RdP can be remotely originated or locally forced, due to the geometry of the estuary and its resonant response to winds with a component aligned with its main axis (NW–SE). Regarding the remote forcing, it is characterized by the passage of planetary waves over

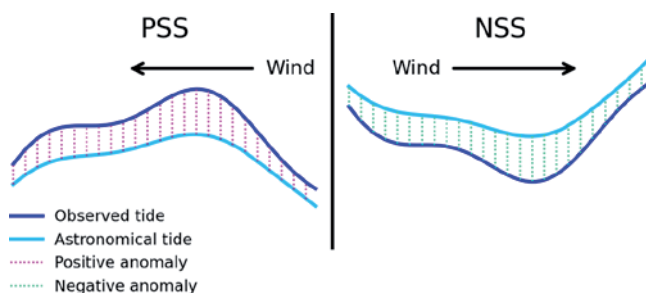


Figure 2.
Diagram of a positive (left) and negative (right) storm surge.

Patagonia, which propagate and reach the estuary. These waves generate winds with a strong alongshore (N–S) component, which induces Ekman transport either offshore or onshore, resulting in negative or positive sea level anomalies, respectively [7]. As a result, the estuary is frequently affected by both positive and negative storm surges [3, 7, 8].

Due to its size, shallow waters, and funnel-like shape, positive storm surges (PSS) in the RdP can push water upriver and cause severe flooding along Buenos Aires. Wind action not only raises river levels but also increases pressure at the outlets of storm drains and streams that flow through the city into the estuary. This pressure obstructs the drainage of accumulated rainwater, leading to flooding along the shoreline and also in the stream valleys within the city, where many of the drainage inlets are located [9]. This frequent phenomenon is locally known as “*sudestada*,” a name that refers to the prevailing wind direction. Conversely, pronounced ebb tides caused by negative storm surges (NSS) can significantly affect navigation safety and the drinking water supply for the City of Buenos Aires [3]. Photographic records of an extreme negative storm surge event from the year 1937 are preserved in the National General Archive. One of them is shown in **Figure 3**. On that occasion, the water level reached -3.58 m below the Tidal Datum, and although the City of Buenos Aires did not fare very well, many men and women walked across the riverbed that was left dry. As noted by Dinápoli et al. [7], negative events have not been extensively studied



Figure 3.
People walking on the dry bed of the Río de la Plata during an extreme negative storm surge event in 1937 (National General Archive).

in this area. Therefore, this work aims to contribute to a better understanding and update of these events by analyzing the effects of mean river level variations in the RdP on storm surges.

In summary, winds from the S, SE, and SSE are associated with the generation of PSS events, whereas winds from the N, NNW, and NNE are linked to the forcing of NSS events.

For their part, significant interactions may occur among tides, storm surges, waves, sea level rise, and flooding; however, the relevance of each interaction mechanism varies considerably depending on the specific location [10]. Consistent with global oceanic trends, the RdP is experiencing a rise in mean river level attributed to global warming (e.g., Refs. [1, 11]) in agreement with the broader South Atlantic sea level pattern [12]. Several studies have shown that the interaction between storm surge and sea level rise can exhibit either linear or nonlinear behavior [13, 14]. It is known that storm surges are influenced by a reduction in bottom friction, which typically tends to amplify their intensity. However, as demonstrated by Ref. [15] in the North Sea, the decrease in bottom friction seems to be offset by the reduced effectiveness of surface wind stress. Specifically, the same wind forcing (surface stress) becomes less effective in driving water, resulting in a smaller surge when the water depth increases. Furthermore, sea level rise will lead to the emergence of new flooded areas in certain locations, which will serve as additional dissipative zones for tides and surges [10]. Building on this concept, this study examines not only the evolution of PSS and NSS events but also investigates whether the rise in the annual mean river level contributes to the observed changes.

2. Data and methods

2.1 Data

The baseline data used in this study consist of hourly water levels recorded in the Río de la Plata estuary, spanning from 1905 to 2024. These observations were obtained using two different tide gauges. The first gauge, operated by the Ministry of Public Works, was located at the mouth of the Riachuelo River—known as *Semáforo del Riachuelo*—and was in service from 1905 to 1959. The second gauge, operated by the Naval Hydrographic Service, is located in the Palermo neighborhood at the Club de Pescadores station, approximately 9 km north of the Riachuelo River, and has been operational from 1957 to the present. Consequently, there are three years (1957–1959) of overlapping data between the two records. The coastline between the two stations presents no significant morphological differences, and the tidal characteristics at both locations are similar, allowing the datasets to be considered comparable [3]. Both tide gauges reference their measurements to the same local datum, defined at the Riachuelo River, which is 0.531 m below the datum of the National Geographic Institute. This local zero was established to closely match the level of the lowest astronomical tides [16]. The resulting dataset spans 120 years, from 1905 to 2024, with a data gap between 1963 and 1964.

2.2 Annual trend analysis of mean river level

Following the methodology proposed in Ref. [1], the observational series was divided into two subsets to estimate the linear trend in the annual mean river level

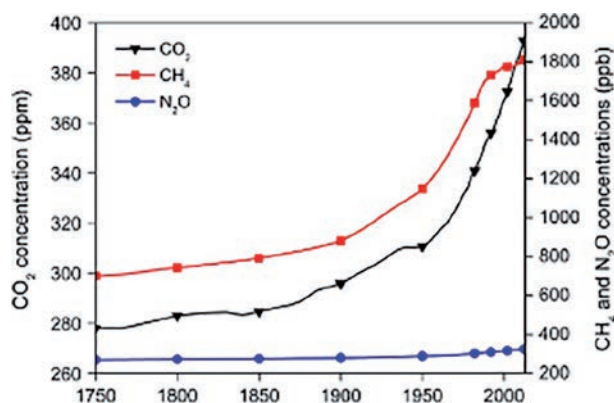


Figure 4.
 Worldwide greenhouse gas emissions from 1750 to 2013 [17].

(AMRL): the first includes data from 1905 to 1962, and the second covers the period from 1965 to 2024. These periods are referred to as P1 and P2, respectively. This division is based on notable changes observed in the global greenhouse gas emission curve (**Figure 4**), which are associated with the onset and intensification of global warming.

As a consequence of global warming, the mean sea level in the RdP has risen, with an acceleration observed since around 1960 [18]—consistent with trends seen across the global ocean. To analyze this change, two series of tide gauge records are examined. For each series, AMRLs are estimated using two approaches: (1) calculating the arithmetic mean of all hourly observations for each calendar year and (2) applying a Savitzky–Golay filter [19] to the annual values, using a window size of 31 years and a third-order polynomial. In the calculated annual arithmetic means, various contributions from long-period astronomical tidal cycles are expected. These include the Metonic cycle of approximately 19 years, among others. Ref. [20] reports that, in regions such as the North Sea and the central Baltic Sea, peak astronomical tides occur at intervals of 74.4 and 55.8 years, multiples of the mentioned 18.6-year lunar nodal modulation cycle. Similarly, Ref. [21], focusing on the Patagonian shelf, concludes that in most cases, a 30-year period is sufficient for the convergence of extreme tidal values. This understanding informed the choice of a 31-year window for the Savitzky–Golay filter. A polynomial order of 3 is selected based on empirical evidence showing that this configuration preserves peak features and fine-scale details in the time series with minimal distortion.

To isolate the nonastronomical river level signal and enable comparison of results, the predicted astronomical tide is subtracted from the hourly height series prior to calculating annual means. This process produces a new time series of AMRLs, which are compared to those derived from the filtered series.

The trend of the filtered series is then estimated using a least squares linear regression, and the Pearson correlation coefficient (r) is calculated to assess the strength of the linear relationship.

2.3 Residuals, storm surges, and events classification

To calculate storm surge events, a series of residuals is required, which is obtained by subtracting the astronomical tide from the observed water level.

The astronomical tide is determined through harmonic least-squares analysis according to Eq. (1). In this equation, the subindex j refers to each tidal constituent, t is the time of each observation, and $h(t)$ is the observed water level. The equation takes the form:

$$h(t) = Z_0 + \Delta Z \cdot t + \sum_{j=1}^n H_j f_j \cos((v+u)_j - g_j) \quad (1)$$

Where Z_0 is the mean river level of the period, ΔZ is the linear trend in the annual mean river level, H_j is the amplitude of each tidal constituent, f_j is the nodal factor, $(v+u)_j$ is the equilibrium argument, and g_j is the modified epoch. This methodology incorporates potential linear variations in mean river level by including the $\Delta Z \cdot t$ term in the tidal prediction. Harmonic constants are obtained directly [22], while equilibrium arguments and nodal factors are estimated using the time scales provided in Ref. [23]. The estimation of error components in the harmonic analysis is based on the variance–covariance matrix derived from the least squares method.

A total of 134 tidal constituents are included in the harmonic analysis to capture a wide range of tidal frequencies. Although harmonic constants are often assumed to be time-invariant, they can be affected by factors such as changes in mean sea level, bathymetry, river discharge, and thermohaline conditions [24, 25]. For example, large tidal amplitudes and energy dissipation occur when the tidal forcing frequency approaches the natural resonant period of an ocean or sea [26]. Rising mean sea levels can therefore modify tidal behavior, shifting a shelf sea or embayment either closer to resonance—resulting in increased tidal range—or further from resonance, reducing it. The sensitivity of tidal response to water depth changes increases as the system approaches resonance. Additionally, greater water depth reduces bottom energy dissipation and alters the tidal wave propagation speed, both of which influence tidal range. To account for these potential changes and the Metonic cycle, harmonic analyses are performed over six separate periods: four 19-year intervals (1905–1923, 1924–1942, 1965–1983, 1984–2002), one 20-year interval (1943–1962), and one 22-year interval (2003–2024), resulting in six distinct sets of harmonic constants.

Astronomical tide predictions are then made in two different ways for each period (P1 and P2). The first one uses: (i) all harmonic constants from the corresponding 19/20/22-year period, (ii) the mean river level for the period, and (iii) the linear trend of the mean river level of the period. The second follows the same parameters as above, but excluding the mean river level trend.

From these, four residual series are computed by subtracting each tidal prediction from the observed water levels. These residuals are then used to identify and characterize storm surge events. The four series of residuals (Eqs. (2)–(5)) include:

$$\begin{aligned} \text{Residuals}_1 = & \text{observations (1905–1962)} \\ & - \text{astronomical tide (}\sim 19 \text{ year harmonic constants)} \\ & - \text{mean river level (1905–1962)} \\ & - \text{mean river level trend (1905–1962)} \end{aligned} \quad (2)$$

$$\begin{aligned} \text{Residuals}_2 &= \text{observations}(1905-1962) \\ &\quad - \text{astronomical tide}(\sim 19 \text{ years harmonic constants}) \quad (3) \\ &\quad - \text{mean river level}(1905-1962) \end{aligned}$$

$$\begin{aligned} \text{Residuals}_3 &= \text{observations}(1965-2024) \\ &\quad - \text{astronomical tide}(\sim 19 \text{ years harmonic constants}) \quad (4) \\ &\quad - \text{mean river level}(1965-2024) \\ &\quad - \text{mean river trend}(1965-2024) \end{aligned}$$

$$\begin{aligned} \text{Residuals}_4 &= \text{observations}(1965-2024) \\ &\quad - \text{astronomical tide}(\sim 19 \text{ years harmonic constants}) \quad (5) \\ &\quad - \text{mean river level}(1965-2024) \end{aligned}$$

Thus, **Residuals**₁ (R1) and **Residuals**₃ (R3) series exclude the mean river level trend and represent a hypothetical scenario. In contrast, **Residuals**₂ (R2) and **Residuals**₄ (R4) retain the influence of the mean river level trend and therefore correspond to a more realistic scenario.

To identify the heights associated with PSS and NSS, the previously estimated residual series are used. However, not all residuals correspond to storm surge events. For a PSS event to be classified as such, the following criteria must be met: (i) residual values must never fall below 0.30 m, (ii) the maximum residual during the event must be equal to or greater than 1.60 m, and (iii) the event must last at least 6 hours. In the case of an NSS, the criteria are: (i) residuals must never exceed −0.30 m, (ii) the minimum residual during the event must be less than or equal to −1.20 m, and (iii) the event must have a minimum duration of 6 hours.

The ±0.30 m threshold is adopted to ensure that the residuals reflect actual storm surge events rather than natural variability. Under calm meteorological conditions, the difference between observed water levels and the astronomical tide is typically around ±0.10 m. Therefore, a threshold three times that value (±0.30 m) provides a conservative filter. The 1.60 m threshold for the PSS is chosen because, when combined with a tidal height near the mean river level (~ 0.79 m above the Riachuelo River database) during a semidiurnal rising tide, it can lead to water levels approaching the warning thresholds for the City of Buenos Aires and surrounding areas [3]. Similarly, the −1.2 m threshold for NSS events corresponds to the drinking water alert level of the RdP for the City of Buenos Aires [8]. Finally, to rule out isolated anomalies and confirm the presence of a sustained storm surge event, a minimum duration of six hours is required.

2.4 Estimating trends in storm surge variables

Once the events are identified, annual trends are estimated for the following variables:

- The annual number of PSS and NSS events,
- The mean and maximum annual duration of PSS and NSS events,

- The annual maximum and minimum observed water levels,
- The annual maximum and mean PSS heights, and
- The annual minimum and mean NSS heights.

To perform this analysis, and following the same methodology used for estimating the AMRL trend, a Savitzky–Golay filter is first applied to the annual data using a third-order polynomial and a window size of 31. This smoothing technique helps to emphasize long-term trends by minimizing the influence of short-term fluctuations. Subsequently, a least-squares linear regression is applied to the periods 1905–1962 and 1965–2024 to determine linear trends. The Pearson correlation coefficient (r) is also calculated to evaluate the strength of the linear relationship. These analyses are conducted using both the R2 and R4 residual series, which include the AMRL trend, and the R1 and R3 series, from which the trend was removed.

Additionally, to provide a more comprehensive understanding, a seasonal analysis of storm surge events is carried out for both periods under the realistic scenario (i.e., using the R2 and R4 residual series).

3. Results of the mean river level trend

As outlined in the preceding 2.2 section, two approaches were implemented to filter the observational time series. The annual river level series processed through the Savitzky–Golay filter and the series obtained by subtracting astronomical tide predictions yielded comparable results, with differences limited to the millimeter scale. Consequently, the filtered series is selected for subsequent trend analysis.

The annual pattern and trend of AMRL are shown in **Figure 5a** for P1 and **Figure 5b** for P2. The corresponding values of the Pearson correlation coefficient (r) are also presented. An acceleration in AMRL is evident between the two periods: the trend increases from 1.1 mm·yr.⁻¹ in P1 to 2.5 mm·yr.⁻¹ in P2. In other words, the AMRL trend in the second period is more than twice that of the first. Additionally, the high r values indicate that both trends are statistically significant.

4. Evolution of storm surge events

To assess the evolution of storm surges over the 120-year observation period and to evaluate the potential contribution of rising mean river levels to this evolution, we analyzed the trends of the previously computed surge-related variables. These include the number of events, maximum and minimum observed water levels, the mean of maximum and minimum observed levels, maximum and mean PSS heights, minimum and mean NSS heights, and the maximum and mean event duration. The results are summarized in **Tables 1** and **2**. In these tables, values of the Pearson correlation coefficient (r) exceeding 0.5 are highlighted in grey. The statistical significance of these variables allows us to identify changes in the behavior of storm surge events. Furthermore, the comparison between the two scenarios and time periods analyzed provides insights into the relationship between storm surge events and the rise in mean river level.

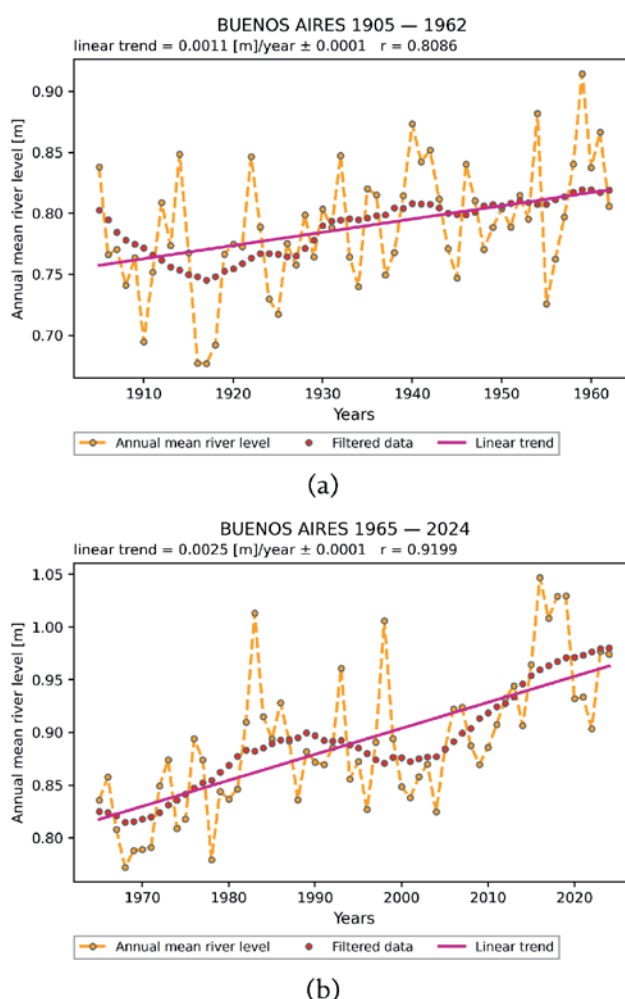


Figure 5.
 (a) AMRL trend during P1 period (from 1905 to 1962) and (b) AMRL trend during P2 period (from 1965 to 2024).

4.1 Trends in PSS-related annual variables

All variables exhibiting statistically significant trends belong to period P2—except for the maximum observed height—indicating that the main changes in PSS behavior have occurred over the past 60 years. These variables include the annual number of events, the annual maximum observed height, the annual mean PSS height, and the annual maximum event duration. The number of events, their maximum duration, and the maximum observed height show positive trends in both the realistic and hypothetical scenarios. In contrast, the trend in mean PSS height is negative in both cases. The trends identified under the hypothetical scenario cannot be attributed to the rising trend in AMRL, as this influence was deliberately excluded from the analysis. Furthermore, the similarity in the trends of event frequency and maximum duration between the hypothetical and realistic scenarios supports the conclusion that the rise in AMRL does not exert a direct influence, despite more pronounced values

Variable (annual)	Period	Hypothetical scenario	r value (hypothetical)	Realistic scenario	r value (realistic)
Number of events [event/year]	P1	0.0115	0.2034	0.0117	0.2154
	P2	0.1411	0.94	0.1649	0.9468
Maximum observation [m/year]	P1	—	—	−0.0015	0.5387
	P2	—	—	0.0029	0.8164
Mean maximum observation [m/year]	P1	—	—	−0.0009	0.591
	P2	—	—	0.0002	0.1402
Maximum PSS [m/year]	P1	−0.0002	0.0671	0	0.0054
	P2	0.0001	0.09	0.0005	0.3024
Mean PSS [m/year]	P1	−0.0006	0.4682	−0.0004	0.3412
	P2	−0.0011	0.8359	−0.0012	0.8529
Maximum duration [h/year]	P1	0.0181	0.0899	0.0187	0.0905
	P2	0.3988	0.8179	0.401	0.7751
Mean duration [h/year]	P1	−0.0353	0.3171	−0.0327	0.2648
	P2	0.0101	0.12	0.0243	0.2414

Values of r greater than 0.5 are shaded in grey to indicate statistical significance.

Table 1.
Trends in storm surge variables associated with PSS, along with their respective Pearson correlation coefficients (*r*).

Variable (annual)	Period	Hypothetical scenario	r value (hypothetical)	Realistic scenario	r value (realistic)
Number of events [event/year]	P1	−0.0337	0.7131	−0.0476	0.8085
	P2	−0.0508	0.862	−0.0508	0.8344
Minimum observation [m/year]	P1	—	—	0.0045	0.431
	P2	—	—	0.0016	0.2276
Mean minimum observation [m/year]	P1	—	—	0.0009	0.4023
	P2	—	—	−0.0002	0.0745
Minimum NSS [m/year]	P1	0.0023	0.2486	0.0025	0.2657
	P2	0.0037	0.4327	0.0032	0.4064
Mean NSS [m/year]	P1	0.0005	0.2212	0.0001	0.0451
	P2	0.0001	0.0496	−0.0003	0.0928
Maximum duration [h/year]	P1	−0.0956	0.2647	−0.0623	0.2308
	P2	−0.016	0.1597	0.0129	0.139
Mean duration [h/year]	P1	−0.0743	0.5018	−0.0612	0.5602
	P2	−0.0216	0.4565	−0.0065	0.2629

Values of r greater than 0.5 are shaded in grey to indicate statistical significance

Table 2.
Trends in storm surge variables associated with NSS, along with their respective Pearson correlation coefficients (*r*).

in the realistic scenario. Regarding event duration, it is noteworthy that the trend is lower in the realistic scenario compared to the hypothetical one, which is contrary to what would be expected if the behavior were directly influenced by AMRL. For the mean PSS height in the realistic scenario, it is particularly notable that the trend is opposite to that of AMRL: while AMRL increases between periods, the mean PSS height decreases. As for the maximum observed height, it is remarkable that its trend is negative during period P1—contrary to the AMRL trend—but it reverses sign in period P2 and reaches a value ($\sim 2 \text{ mm}\cdot\text{yr}^{-1}$) nearly identical to that of the AMRL trend in the same period (**Figure 5b**).

4.2 Trends in NSS-related annual variables

In the case of negative events, only two variables—the number of events and their mean duration—exhibit significant trends, with the mean duration showing a Pearson correlation coefficient (r) greater than 0.5, but only during period P1. As in the analysis of PSS, the trends identified in the hypothetical scenario are not modulated or influenced by the increase in AMRL, since this factor was excluded from that scenario. Furthermore, within a given period, the values across both scenarios are similar, indicating a consistent outcome. Finally, when focusing on the number of events in the realistic scenario and comparing the trends across both periods, we find not only that the trends are similar—thus not reflecting the doubling of the AMRL trend—but also that the number of events increases, contrary to what would be expected in response to a rise in AMRL.

5. Seasonal analysis of storm surges

To complement the annual trend analysis and gain deeper insight into the temporal behavior of storm surge events, a seasonal breakdown of their occurrence is also performed. As the RdP is located in the Southern Hemisphere, seasonal references correspond to this hemisphere. Accordingly, summer includes December, January, and February; autumn comprises March, April, and May; winter includes June, July, and August; and spring encompasses September, October, and November.

5.1 Seasonal variability of PSS

Figure 6 illustrates the seasonal behavior of the variables associated with PSS, evaluated for both periods under the realistic scenario—that is, considering the variation in AMRL.

The number of events (**Figure 6a**) is highest in summer, followed by spring and autumn, with the lowest frequency observed in winter. However, it is during winter that PSS events exhibit the longest duration in both P1 and P2 (**Figure 6f**). With respect to their maximum (**Figure 6d**) and mean amplitudes (**Figure 6e**), the variables exhibit similar patterns regarding which period attains the highest values in each season; however, the seasons corresponding to the highest and lowest values differ between periods. In P2, the highest values for both variables are recorded in spring and winter, whereas in P1, they occur in autumn and winter. This indicates that the most extreme events tend to take place during winter in both periods.

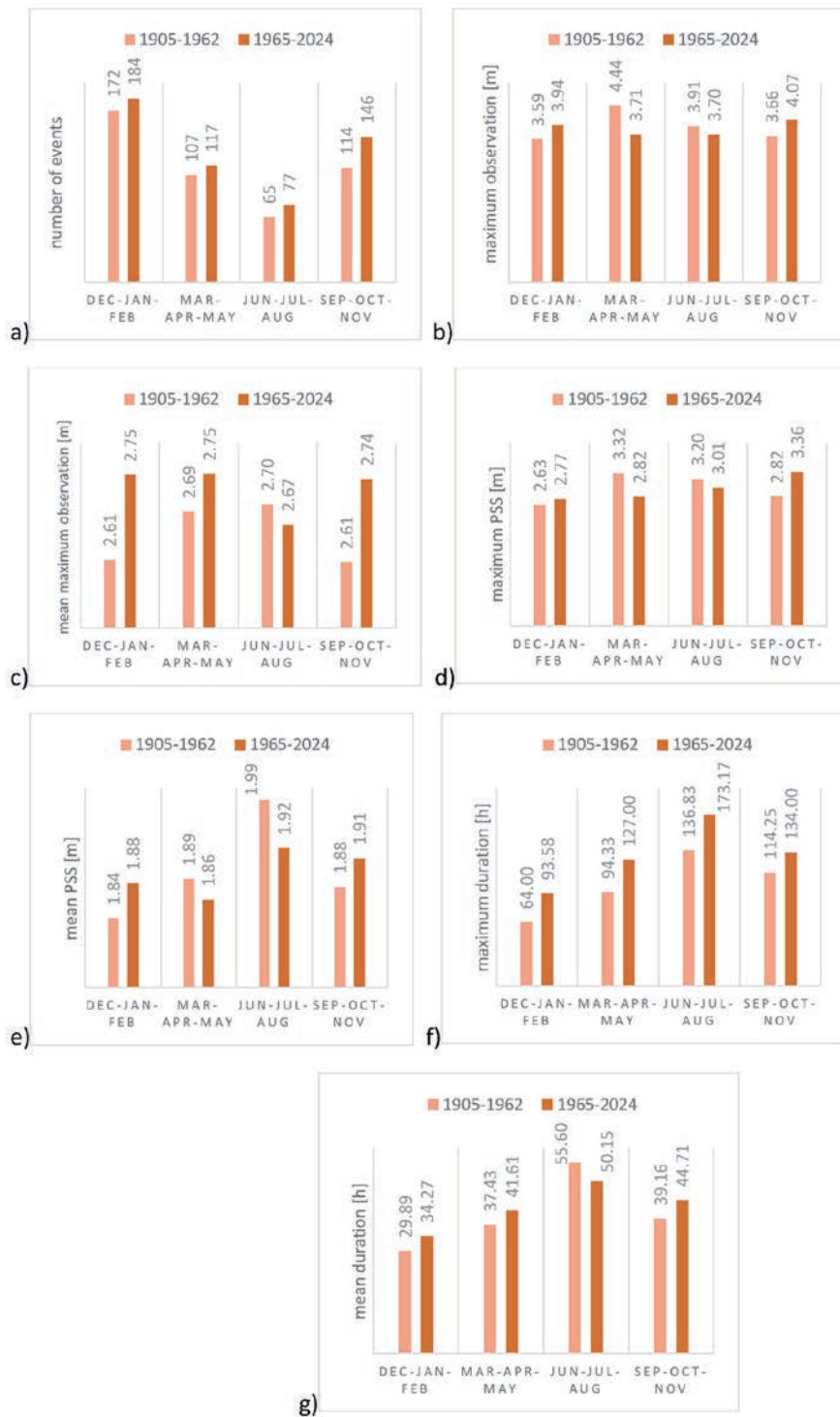


Figure 6. Seasonal analysis for P1 and P2 in the realistic scenario of: (a) number of PSS events, (b) maximum observation, (c) mean maximum observation, (d) maximum PSS, (e) mean PSS, (f) maximum duration, and (g) mean duration.

Another notable feature is that both the maximum observed water level (**Figure 6b**) and the maximum PSS (**Figure 6d**) reach higher values in summer and spring during P2 compared to P1, while the opposite is observed in autumn and winter, where higher values are recorded in P1.

The seasonal patterns of the mean maximum water level (**Figure 6c**) and mean PSS (**Figure 6e**) differ. In P2, the highest mean water levels occur in summer, autumn, and spring, while in P1, they are observed in autumn and winter. In contrast, mean PSS values reach their peaks in winter and spring in P2, whereas in P1, the maximum occurs in winter.

The only variables that show an increase across all seasons from P1 to P2 are the number of events (**Figure 6a**) and the maximum duration of these events (**Figure 6f**).

The mean event duration (**Figure 6g**) behaves similarly to the maximum duration (**Figure 6f**) in all seasons, except during winter, when P1 exceeds P2.

Figures 7–9 display the seasonal distribution of the maximum PSS, the maximum PSS duration, and the number of events, respectively, derived from the R2 and R4 residual series over the years. Seasons without events—due to the application of a threshold criterion of 1.60 m—are shown in gray, while the years 1963 and 1964, for which no data are available, are indicated in blue.

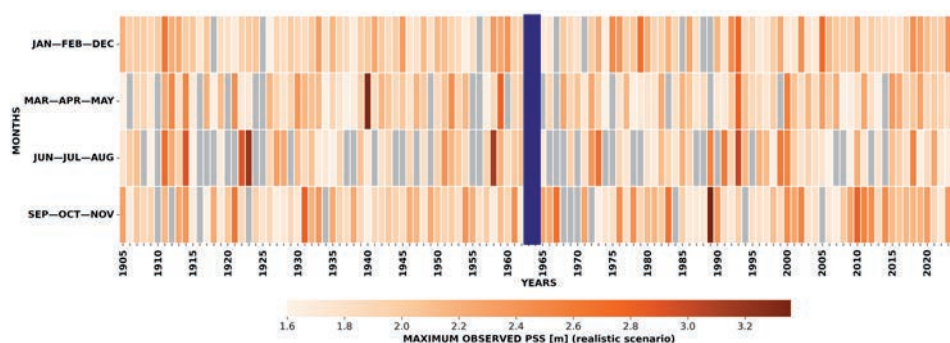


Figure 7.
 Seasonal distribution over the years of the maximum PSS obtained from the R2 and R4 residual series (realistic scenario).

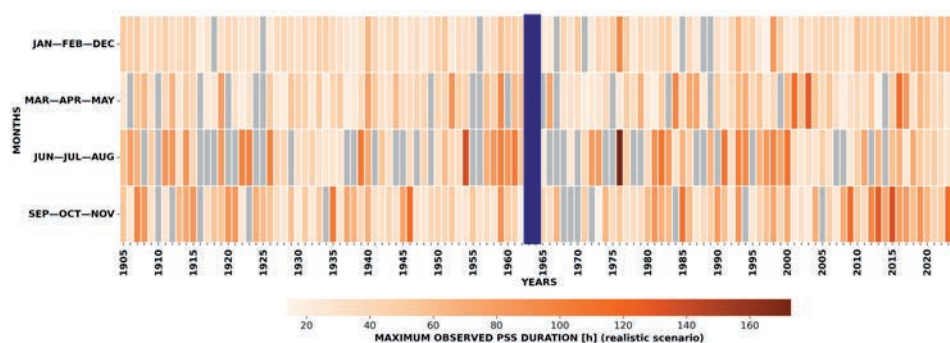


Figure 8.
 Seasonal distribution over the years of the maximum PSS duration obtained from the R2 and R4 residual series (realistic scenario).

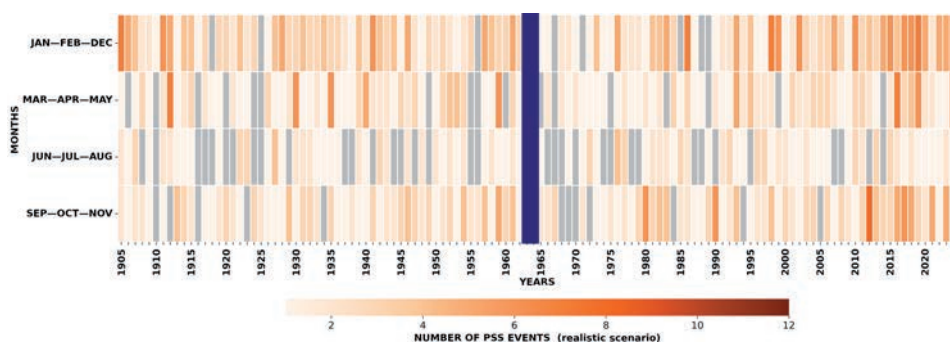


Figure 9. Seasonal distribution over the years of the number of PSS events obtained from the R2 and R4 residual series (realistic scenario).

In **Figure 7**, spring 1989 stands out as the event with the greatest height (3.35 m), followed by autumn 1940 (3.32 m). These events represent the most extreme values recorded in periods P2 and P1, respectively. The figure also shows that, starting in 2008, spring consistently registers the highest values, exhibiting greater persistence compared to earlier years.

Regarding the maximum duration of PSS events (**Figure 8**), the historical maximum corresponds to the winter of 1976, lasting 173 hours. As shown in **Figure 6f**, winters generally exhibit the highest values over the entire record. However, in recent years, spring values have surpassed those recorded during winters.

In **Figure 9**, summer clearly emerges as the modal season, characterized by a markedly higher number of events. This is evidenced by the darker shading, which also highlights a notable increase in event frequency during recent springs.

5.2 Seasonal variability of NSS

Figure 10 illustrates the seasonal behavior of the variables associated with NSS, evaluated for both periods under a realistic scenario that accounts for variations in the AMRL.

The highest number of NSS events (**Figure 10a**) occurs during winter in both P1 and P2, while the lowest values are recorded in summer, in both periods.

The behavior of the minimum observed value (**Figure 10b**) and the minimum NSS (**Figure 10d**) is analogous within each period; however, the minimum values of these variables do not occur in the same season. In P2, the minimums are observed in autumn, whereas in P1, they occur in winter.

The mean of the minimum observations (**Figure 10c**) and the mean NSS (**Figure 10e**) do not exhibit analogous behavior. The minimum observed values are lower in P1 across all seasons, whereas the mean NSS values are lower in P2. Moreover, while the lowest mean observed minimum occurs in winter for both periods, the minimum mean NSS values are recorded in spring during both P1 and P2.

Regarding event duration, the maximum durations (**Figure 10f**) occur in spring during P1 and in autumn during P2, indicating that the most severe events take place in autumn during the latter period. In contrast, the mean duration (**Figure 10g**) shows little seasonal variation, except in summer, when the lowest values are observed.



Figure 10. Seasonal analysis for P1 and P2 in the realistic scenario of: (a) number of NSS events, (b) minimum observation, (c) mean minimum observation, (d) minimum NSS, (e) mean NSS, (f) maximum duration, and (g) mean duration.

Both the number of events (**Figure 10a**) and the mean minimum observed value (**Figure 10c**) are lower in absolute values in P2 across all seasons. In contrast, the mean NSS (**Figure 10e**) is lower in absolute values in all seasons during P1.

When comparing the minimum NSS (**Figure 10d**) and the mean NSS (**Figure 10e**), it can be observed that, for the minimum NSS in P1, values are lower than in P2 during summer, winter, and spring, while in autumn, P2 shows lower values than P1. This latter pattern is consistent across all seasons for the mean NSS.

Figures 11–13 show the distribution of the minimum NSS, the minimum observation, and the number of events over the years in the realistic scenario, respectively. As in Section 5.1, gray indicates seasons without events—due to the application of a threshold criterion of -1.20 m—while blue corresponds to the years 1963 and 1964, for which no data are available.

In **Figure 11**, the historic extreme event of 1937 (-4.22 m) is clearly identifiable, along with others that occurred in 1908 (-3.75 m), 1945 (-3.59 m), and 1948 (-3.65 m)—all belonging to P1—and a more recent event from 2002 (-3.75 m), which belongs to P2.

In **Figure 12**, the events with the longest historical durations can be observed. All of them occurred during P1—in 1920 (83 h), 1923 (88 h), and 1951 (86 h)—and took place in winter and spring. Moreover, as shown in **Figure 10f**, the duration of events

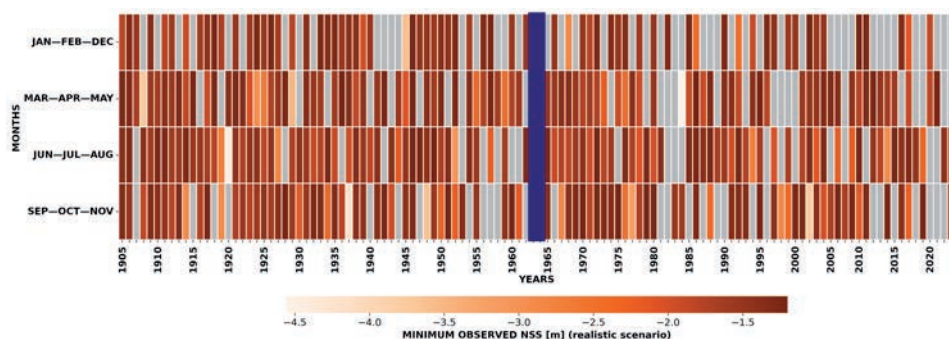


Figure 11. Seasonal distribution over the years of the minimum NSS obtained from the R2 and R4 residual series (realistic scenario).

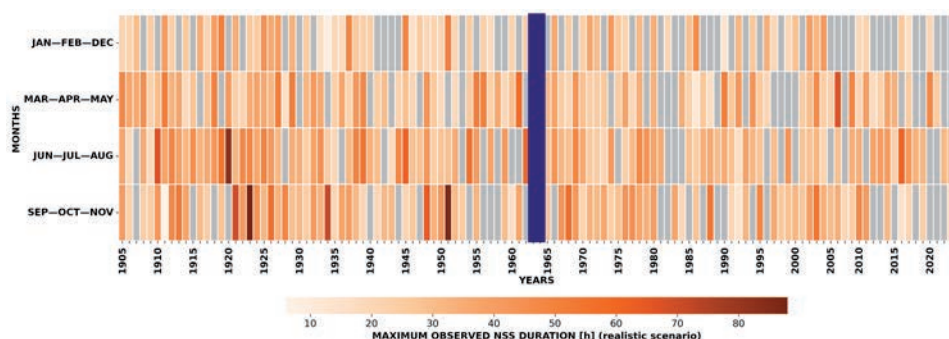


Figure 12. Seasonal distribution over the years of the maximum NSS duration obtained from the R2 and R4 residual series (realistic scenario).

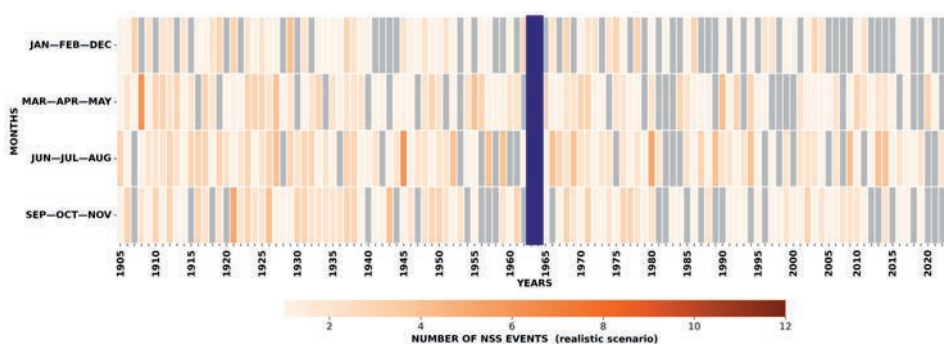


Figure 13.
 Seasonal distribution over the years of the number of NSS events obtained from the R2 and R4 residual series (realistic scenario).

in P2 has decreased compared to P1 at most seasons. However, it should be noted that the maximum duration of NSS did not exhibit a significant trend (**Table 2**).

In **Figure 13**, a decreasing trend in the number of events from P1 to P2 can be observed. Specifically, the mean annual number of events drops from 6.26 in P1 to 4.74 in P2. This decline has been evident across all seasons since 1980. Additionally, it is apparent that, in recent years, winter has become the season contributing most significantly to the statistics.

6. Discussions and conclusions

The analysis of PSS-related variables' annual trends over the two periods, P1 and P2, reveals a marked intensification in recent decades. During P1, trends for most variables are weak or nearly flat, regardless of the scenario, with low correlation coefficients indicating limited temporal coherence. In contrast, P2 exhibits clearer and stronger trends—particularly in the number of events and storm surge duration—with higher correlation values under both scenarios. The number of events and the maximum duration show slightly higher values in the realistic scenario, which incorporates mean sea level rise, suggesting that sea level increase may contribute to the enhanced frequency and persistence of PSS. The transition from P1 to P2 reflects a shift toward more frequent and longer-lasting storm surges, highlighting the growing influence of long-term sea level rise on coastal hazard patterns. However, a meteorological component also contributes to shaping these events, accounting for some of the variability observed throughout the study period. Atmospheric dynamics and their fluctuations play a crucial role in this behavior [27]. The absence of a clear trend in the annual maximum PSS height, across both periods and scenarios, suggests that this variable is not directly influenced by the rise in AMRL. Instead, its behavior may be shaped by interactions with astronomical tides. Sea level rise could slightly suppress tidal heights due to nonlinear interactions between tides and PSS events [28]. Additionally, tidal cycles—such as the 18.61-year cycle driven by the precession of the Moon's orbit—can modulate extreme water levels [20, 29]. These factors may also help explain the observed decreasing trend in mean PSS.

The analysis of annual trends for NSS events indicates that only two variables—the number of events and their mean duration—exhibit statistically significant linear trends. Both display decreasing trends under the hypothetical and realistic scenarios,

particularly in P1. These results are consistent across scenarios, suggesting that the observed behavior is not influenced by AMRL rise. In contrast, other variables, such as minimum observations, minimum NSS values, and maximum durations, show weak or no trends, and their evolution remains largely unaffected by whether the AMRL rise is considered. Overall, the increase in AMRL does not appear to modulate or diminish the intensity of NSS events in the Río de la Plata estuary. Consequently, the observed changes in the number of events may be associated with alterations in atmospheric circulation that exhibit persistence across both periods, rather than with recent global warming directly.

In summary, the influence of rising AMRL differs significantly between positive and negative storm surge events. In the case of PSS, a relationship with the recent evolution of AMRL can be identified, whereas no such link is evident for NSS. Nonetheless, rising AMRL is not the sole influencing factor in PSS; changes in atmospheric circulation—particularly the apparent persistence of easterly and southeasterly winds in winter, and their increased frequency in summer and, more recently, in spring—must also be considered. These atmospheric shifts may, in turn, be linked to broader climate change impacts, including global warming and the resulting AMRL rise.

From the seasonal analysis, we note that the highest frequency of positive events occurs in summer. However, the most extreme events take place in winter, where greater amplitude and longer durations coincide. Negative events, in contrast, show the highest frequency in winter, while the most extreme events of this type have occurred mainly in autumn in recent years (P2), in agreement with Refs. [7, 8].

It is also observed that the season with the highest number of positive events (summer) corresponds to the lowest occurrence of negative events, and vice versa. In winter, when negative events are most frequent, the occurrence of PSS reaches its minimum. This seasonal pattern has remained consistent over the years, although an increase in PSS cases during recent spring seasons has been noted.

Our analysis has provided insights into the evolution of storm surge events over the last 120 years and their connection to the rising mean river level. Furthermore, the findings may support planning and decision-making processes in coastal engineering and development, as well as in early warning systems and alert thresholds concerning water supply and navigation safety.

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Edited by Leonel Pereira

Estuaries are among the most vibrant and complex environments on the planet, places where land, river, and sea converge to create ecosystems of extraordinary productivity, diversity, and change. This volume, *Estuaries – Dynamic Ecosystems at the Land–Sea Interface*, offers a compelling exploration of these transitional systems through a multidisciplinary lens, revealing how physical forces, ecological processes, and human pressures interact to shape estuarine futures. Across six richly detailed chapters, leading researchers examine estuaries from hydrodynamic, ecological, geological, and socio-economic perspectives. The book opens with a foundational overview of estuarine structure and function, establishing the scientific context needed to understand their inherent dynamism. A century-long analysis of storm surges in the Río de la Plata follows, uncovering subtle but significant shifts in extreme events under changing climatic conditions. Subsequent chapters highlight the ecological importance of coastal vegetated ecosystems in UK estuaries, demonstrating their roles as blue carbon sinks, biodiversity hotspots, and essential components of resilient food webs. A trait-based investigation of estuarine polychaetas provides new insights into population performance and environmental quality, while a focused perspective on aquifer geology underscores the often-overlooked influence of subsurface processes on submarine groundwater discharge and seawater intrusion. The volume concludes with a regional case study from West Africa, emphasizing the indispensable role of estuaries in sustaining coastal fisheries, livelihoods, and food security. Together, these contributions paint a vivid picture of estuaries as dynamic, interconnected systems whose health is vital to both nature and society. This book is an essential resource for researchers, students, coastal managers, and anyone seeking a deeper understanding of the processes that govern estuarine environments and the challenges they face in a rapidly changing world.

Maurizio Lazzari, Earth Sciences Series Editor

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