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Ecological characterisation of peatlands and coastal lagoons in Europe

Knowledge Synthesis for Policy

Cetinic, K.A., Pérez-Ruzafa, Á., Boix, D., Cravo-Laureau, C., Klimkowska, A., Mallast, U., Poursanidis, D. and Carrasco, A.R.

Editors: Blasi, M., Korcheva, A., Vasilakopoulos, P., Velasco Gomez, D.M.

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Prepared by: bioagora

Contact information
Name: EC Knowledge Centre for Biodiversity
Address: Joint Research Centre of the EC
TP 124, Via E. Fermi 2749 I-21027 Ispra (VA)
Email: **EC-Biodiversity-KC@ec.europa.eu**

Joint Research Centre
https://joint-research-centre.ec.europa.eu

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Contents

Abstract

This report is the response to a policy request submitted by DG Environment to the EC Knowledge Centre for Biodiversity (KCBD) through the KCBD ticketing system. KCBD assigned this to the BioAgora project, to be answered by the Science Service for Biodiversity (SSBD).

European wetlands, including peatlands and coastal lagoons, are among the most valuable yet degraded ecosystems, with up to 80% lost over the past century and more than half of peatlands already drained. Despite their limited extent, peatlands store nearly one-third of global soil carbon, while coastal lagoons occupy ~13% of the world's coastline and sustain highly productive and biodiverse systems. The report identifies hydrology as the key control on ecosystem functioning, showing that drainage, land-use change, and nutrient enrichment trigger cascading and often irreversible degradation, turning carbon sinks into major emission sources. It highlights that no single indicator can capture ecosystem condition, and that effective assessment requires integrated, multi-scale frameworks combining Earth Observation with in-situ data.

These findings provide a critical scientific basis for implementing EU policies — including the Habitats Directive (92/43/EEC), Water Framework Directive (2000/60/EC), and Nature Restoration Regulation (2024/1991) — supporting harmonised monitoring, restoration prioritisation, and evidence-based decision-making.

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<u>Process Coordination and Facilitation</u>

This knowledge synthesis was coordinated through the BioAgora project. The request handling and expert coordination were overseen by:

Request focal points: Frédéric Gosselin (National Research Institute for Agriculture, Food and Environment- INRAE, France); Maria Blasi (Centre for Ecological Research and Forestry Applications – CREAF, Spain); Sibylle Schroer (Leibniz-Institute of Freshwater Ecology and Inland Fisheries - IGB, Germany); Mihai Cristian Adamescu (University of Bucharest, Romania)

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The focal point(s) served in a coordination and process facilitation capacity, ensuring adherence to ethical standards, facilitating communication between the expert working group and requesters, overseeing the peer review process, and maintaining the integrity of the knowledge synthesis methodology. Their role was specifically designed to support the independence and objectivity of the expert working group while ensuring procedural rigor and transparency.

KCBD contact points: Paris Vasilakopoulos (European Commission, Joint Research Centre, Ispra, Italy), D. Marcela Velasco Gómez (Piksel Srl, EC consultant, Ispra, Italy)

DG ENV contact point: Bruno Combal (European Commission, DG Environment (ENV.D3), Brussels, Belgium)

Editors: Maria Blasi (Centre for Ecological Research and Forestry Applications – CREAF, Spain); Alexandra Korcheva (Pensoft Publishers, Bulgaria); Paris Vasilakopoulos (European Commission, Joint Research Centre, Ispra, Italy); Marcela Velasco Gomez (Piksel Srl, EC consultant, Ispra, Italy)

<u>Expert Contributors</u>

Authors

Katarina Ana Cetinic (Norwegian Institute for Water Research - NIVA, Norway)

Ángel Pérez-Ruzafa (University of Murcia, Spain)

Dani Boix (University of Girona, Spain)

Cristiana Cravo-Laureau (Université de Pau et des Pays de l'Adour – UPPA, France)

Agata Klimkowska (Wetlands International, Netherlands)

Ulf Mallast (Helmholtz Centre for Environmental Research - UFZ, Germany)

Dimitris Poursanidis (Foundation for Research and Technology Hellas – FORTH, Greece)

Ana Rita Carrasco (University of Algarve, Portugal).

Reviewers: Martina Gaglioti (Italian Ministry of Education and Research, Italy); Bruna Gumiero (University of Bologna, Italy); Noemí Silva-Sánchez (Universidad de Santiago de Compostela, Spain); Meriam Lahsaini (consultant European Commission, Joint Research Centre, Ispra, Italy), Christelle Vancutsem (European Commission, Joint Research Centre, Ispra, Italy).

Conflicts of Interest and Independence

All expert contributors, methods experts, and focal points completed conflict of interest declarations. The focal points did not contribute to the substantive content of this output to maintain process independence and objectivity, while the methods expert participated as a full member of the expert working group with full intellectual contribution rights. Complete conflict of interest documentation is available upon request.

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For questions regarding the coordination process or to provide feedback on this output, please contact: Frederic Gosselin <frederic.gosselin@inrae.fr>

Executive Summary

Wetlands in Europe: Status and pressures

Wetlands, including peatlands and coastal lagoons, are among the most ecologically significant ecosystems in Europe. Despite covering a small fraction of the European land surface, they provide ecosystem services that are disproportionate to their extent, including climate regulation, water purification, flood regulation, coastal protection, fisheries production, and biodiversity support. Europe has experienced substantial wetland loss and degradation, and remaining wetland systems continue to face sustained and interacting pressures from land-use change, altered hydrology, nutrient enrichment, pollution, and climate change. These pressures compound one another, making degradation difficult to detect, attribute, and reverse. At the same time, approaches to wetland assessment, monitoring, and restoration evaluation remain inconsistent across Member States, highlighting the need for harmonized, operationally feasible frameworks grounded in current scientific understanding.

Report scope and policy context

This report is the response to a policy request submitted by DG Environment to the EC Knowledge Centre for Biodiversity (KCBD) through the KCBD ticketing system. KCBD assigned this to the BioAgora project, to be answered by the Science Service for Biodiversity (SSBD).

This document is intended to support the monitoring and assessment of ecological condition under the Habitats Directive (HD; 92/43/EEC), Water Framework Directive (WFD; 2000/60/EC), and Nature Restoration Regulation (NRR; 2024/1991). It provides an ecological characterization of two contrasting Annex I HD wetland habitat groups: *inland peat-forming wetlands* within Group 7 "Raised bogs, mires and fens", and *coastal lagoons*, corresponding to habitat type H1150. For each habitat group, it delivers an ecological characterization of ecosystem structure and functioning, a description of primary pressures and their interactions, an analysis of degradation pathways, and a tiered indicator framework distinguishing essential from complementary variables, with explicit guidance on the integration of Earth Observation and in-situ measurements.

A common methodological principle underlies both assessments: effective ecological characterization requires the assessment of ecosystem processes and functioning, not only habitat extent and species composition. Natural spatial and temporal variability must be explicitly accounted for when interpreting wetland condition, degradation or recovery. Without this context, assessment frameworks risk confusing intrinsic ecosystem dynamics with anthropogenic impacts.

Peatlands

European peatlands store disproportionately large quantities of soil carbon relative to their extent and have experienced extensive loss and degradation, primarily through drainage for land conversion and peat extraction. Hydrology is the dominant control on peatland functioning. A persistently high and stable water table maintains the anoxic conditions that suppress decomposition, sustain peat-forming vegetation and preserve long-term carbon stocks. Its loss can initiate self-reinforcing degradation cascades involving peat oxidation, subsidence, vegetation and microbial shifts, bare peat formation, and increased fire risk. These processes can convert peatlands from long-term carbon sinks into significant greenhouse gas sources, and are difficult and costly to reverse.

Although Group 7 habitat types range from acidic ombrotrophic bogs to mineral-rich fens and spring systems, they share a common dependence on waterlogged conditions that justifies a common assessment framework. However, differences in hydrology, chemistry, vegetation and spatial organization require habitat-specific interpretation of indicators and thresholds. Robust assessment requires a multi-indicator approach combining hydrological condition, peat properties, vegetation composition, and information on pressures. Earth Observation can support landscape-scale assessment of selected peatland condition metrics, particularly those related to hydrology, vegetation and surface change, but should be combined with in-situ measurements for calibration, validation and process-level interpretation.

Coastal lagoons

Coastal lagoons are highly productive and heterogeneous transitional ecosystems characterized and shaped by shallow depths, restricted connectivity and confinement, where multidirectional environmental gradients driven by marine exchange and freshwater inputs regulate sediment dynamics, nutrient availability, biological productivity and stochastic colonization processes. They support biodiversity and ecosystem services linked to fisheries, water purification, carbon storage, coastal protection, tourism and cultural value, while also influencing adjacent coastal and marine systems. These same drivers generate substantial natural variation in community composition even in the absence of anthropogenic pressure, making condition assessment challenging, particularly where standard ecological indices do not adequately distinguish natural variability from anthropogenic degradation. Lagoon degradation is driven by interacting pressures, including nutrient enrichment and eutrophication, contaminant inputs, habitat loss or physical alteration, coastal development, changes in inlet dynamics and climate change. Because responses are often non-linear and context-dependent, assessment requires multi-scale sampling designs, appropriate reference conditions, and indicator frameworks that integrate physico-chemical, biological and functional variables across relevant spatial and temporal scales.

Implications for monitoring and assessment

Across both habitat groups, degradation typically proceeds through interacting pathways rather than a single cause, and no single indicator is sufficient for robust condition assessment. Monitoring programs must be supported by robust experimental designs incorporating controls, replication, and hierarchical spatial sampling frameworks that capture natural variability across multiple ecological scales. At the same time, effective monitoring requires integrated, multi-indicator frameworks that combine Earth Observation with field-based measurement and that are applied within the ecological context specific to each habitat type. The indicator frameworks developed in this report provide a structured basis for that approach, though translating them into fully operational assessment protocols – including calibrated, habitat-specific thresholds – remains a critical next step beyond the scope of this report.

1 Introduction

In November 2024, DG Environment submitted a policy request to the EC Knowledge Centre for Biodiversity (KCBD) through the KCBD Ticketing System (Ventocilla et al., 2024), for a comprehensive ecological characterization of two representative wetland habitat groups: (1) inland peat-forming wetlands (bogs) and (2) coastal wetlands (coastal lagoons), as well as a blueprint methodology for wetland assessments and identification of knowledge gaps. The KCBD assigned this task to the Science Service for Biodiversity (SSBD), which is currently being developed by the BioAgora project.

Wetlands rank among the planet's most productive ecosystems, on par with rainforests and coral reefs, and sustain rich biodiversity, supporting species across all major groups from microbes to mammals (Adam, 1990). Their biological communities are shaped by physical and chemical factors such as climate, topography, geology, nutrient availability, and hydrology (Ramsar Convention on Wetlands, 2018; UNEP, 2022; UNEP, 2024). As multifunctional natural infrastructure, wetlands provide vital ecosystem services, including water purification, flood regulation, carbon storage, and habitat for wildlife (e.g. Costanza et al., 1997; Zedler & Kercher, 2025).

Wetlands are generally classified in scientific literature into three main categories: inland (non-tidal), coastal (tidal and non-tidal), and human-made (constructed) wetlands, based on hydrology, vegetation, and soil (Junk, 2024). About 80 % of the European wetlands that existed 100 years ago have been lost leading to a substantial decrease in the number, size and quality of large bogs and marshes, and small or shallow water bodies. The trend continues, albeit more slowly (Verhoeven, 2014). The various factors contributing to this trend of degradation include land use changes, increased water extraction, alterations in flooding patterns due to human infrastructure like dams and levees, pollution, the introduction of invasive species, and shifts in temperature and rainfall, amongst others.

Indicators are essential for monitoring, assessing, and managing wetland health by tracking changes in ecosystem functions, composition and structure over space and time, guiding restoration efforts, and informing policy decisions. Several studies have developed indicators of wetland health, ranging from animal and plant species to soil and water quality, to fieldwork and remote sensing (e.g. Géant et al., 2025). These studies continue to grow, increasingly integrating new approaches and innovations (e.g. Herlihy et al., 2019). Unfortunately, studies that integrate direct drivers of wetland degradation and human-related landscapes in a local setting context- dependent, are limited. A need remains to consolidate indicators and establish unified, consensual methodologies for assessing habitat quality to evaluate resistance-resilience trade-offs and adaptation pathways to isolated or compounded effects. This would also facilitate monitoring, standardised research practices, alignment with environmental policies, and, above all, support to local management and governance.

The current policy-oriented review details the main features of two selected wetland habitats in Europe (extracted from the Annex I of the EU Habitats Directive; HD) as well as an assessment of knowledge gaps (including lack of representativity of the information and knowledge available). Peatlands (bogs, fens, raised and blanket mires) are permanently waterlogged ecosystems in which accumulation of plant material under anoxic conditions forms peat, making them major terrestrial carbon stores (Scharlemann et al., 2014; Leifeld & Menichetti, 2018; UNEP, 2022). Hydrological alteration, particularly drainage for agriculture, forestry, peat extraction and infrastructure, is widely recognised as the single most important direct driver of degradation in peatlands (Andersen et al., 2017; Swindles et al., 2019; Ramsar Convention on Wetlands, 2021). Coastal lagoons (transitional ecosystems) are shallow, sheltered coastal water bodies, often separated from the ocean by barrier islands or reefs; they include adjacent wetlands that function as blue carbon systems, storing carbon in living biomass and in anoxic sediments. Land-use change, pollution and eutrophication, urban development, and climate change (including sea-level rise and altered hydrological regimes), amongst others, drive their degradation (Newton et al., 2014; Pérez-Ruzafa et al., 2019a). Degradation of peatlands, wetlands, and coastal lagoons can convert these systems from carbon sinks into net sources by releasing CO_2 and CH_4. Conversely, conservation and restoration measures (e.g. rewetting, halting peat extraction, sediment management, and re-vegetation) can restore carbon sequestration capacity and support climate mitigation and adaptation (Ramsar Convention on Wetlands, 2021).

Although all types of ecosystems that can be grouped under the concept of wetlands are considered special areas of conservation (Annex I of the HD), included in Natura 2000, and some, such as coastal lagoons (H1150) or inland salt meadows (H1340), are even considered priority habitat type, in practice, some of them, such as coastal lagoons, have a clear lack of general guidelines for their management, conservation or restoration. For others, including habitat mudflats and sandflats not covered by seawater at low tide (H1140) or Estuaries (H1130), several documents with technical guidelines are available (Gubbay & Garcia-Herrero, 2025; BISE website, 2026).

For each habitat group included in this study, we provide the general characterisation, pressures and degradation pathways, key assessment parameters, and replicable assessment approaches for selected habitat types, based on thresholds and indicator matrices. The report aims to deliver a comprehensive ecological characterisation of these wetland habitats and to identify knowledge gaps and representativity limits in existing information. Emphasis is placed on functional ecological components, degradation indicators, and sensitivity to pressures and threats. This ecological assessment is expected to form the foundation for future monitoring.

2 Inland wetlands: peat-forming habitats

Key message

Peat-forming inland wetlands are water-controlled, carbon-dense ecosystems whose functioning depends on the maintenance of a persistently high and stable water table. Despite sharing this common functional principle, no two peatlands are identical: bogs, fens, and transitional mires differ fundamentally in their water source, chemistry, vegetation, and ecological dynamics, and these differences require type-specific indicator thresholds and assessment approaches. What unifies all peat-forming habitats is that hydrological alteration – principally drainage – is the dominant driver of degradation, and that degradation proceeds through self-reinforcing cascades linking hydrology, vegetation, peat chemistry, and carbon emissions that become progressively harder to reverse over time. Controlling spatial variability (across habitat types, gradients, and landscape scales) and temporal variability (seasonal water table fluctuations, interannual drought cycles, and successional change) is therefore essential to distinguish anthropogenic degradation from natural peatland dynamics. The identification of key variables, the selection of indicators, and the interpretation of thresholds must all be placed in this context of structural and functional heterogeneity.

2.1 Introduction to inland peat-forming habitats

Peatlands are globally significant ecosystems that, despite only covering around 3-4% of the land surface, store close to one-third of the world's soil carbon (Scharlemann et al., 2014; Leifeld & Menichetti, 2018; UNEP, 2022). This disproportionate carbon stock means that apparently small changes in their condition have outsized consequences for climate mitigation and long-term carbon budgets (UNEP, 2022). In Europe, peatlands cover tens of millions of hectares (Fig. 1), yet more than half of this area has been drained or converted, primarily for agriculture, forestry, and peat extraction (Joosten, 1997; Andersen et al., 2017; Nordbeck & Hogl, 2024), making Europe the continent with the highest proportional loss of mires (Tanneberger et al., 2021).

In this report, peat-forming inland wetlands are understood as ecosystems underlain by peat soils, irrespective of peat thickness, whose vegetation and hydrology are (or were until recently) capable of sustaining net peat accumulation. This usage follows the terminology adopted by the Ramsar Convention on Wetlands and recent global assessments (e.g. UNEP, 2022; 2024). From an ecological and hydrological perspective, peat-forming systems are often grouped into fens and bogs (Annex 1). Generally, bogs are rain-fed (ombrotrophic) systems that are characteristically acidic and nutrient-poor, typically dominated by *Sphagnum* and other peat-forming mosses (Ramsar Convention on Wetlands, 2018; UNEP, 2022; 2024). These conditions, under quasi-permanent water saturation, strongly suppress microbial decomposition and underpin the very high, long-term carbon accumulation that makes peatlands such an immense carbon stock. Fens, by contrast, are systems fed by groundwater and surface waters (minerotrophic), often exhibiting higher base status and supporting more diverse vascular plant communities, including sedges and brown mosses (Ramsar Convention on Wetlands, 2018; UNEP, 2022; 2024). However, this broad division is not absolute. Certain peat-forming habitats, such as aapa mires, do not fully fit within this strict classification. Furthermore, in field conditions, these habitats rarely occur in isolation; instead, they form ecological gradients or sequences. For instance, transitional fens are frequently found along the borders of bogs, while close to spring fens or alkaline fens, one often finds *Alnus* (alder) carr woods – wet forests characterized by *Alnus* growth on peat. In Europe, these various ecosystem types are represented among the Habitats Directive Annex I priority habitats.

Figure 1. The European Peatland Map 2025 ('EPM 2025'). A composite map of peatland and organic soil distribution in Europe, compiled from best-available national datasets.

Source: Tegetmeyer et al. (2025).

Beyond the broad fen–bog distinction, peatland habitat types also differ in the spatial scale at which their ecological condition is most effectively assessed. Blanket bogs (7130) are extensive landscape-scale systems whose condition depends strongly on whole-mire and catchment-scale hydrology, including drainage patterns, topography, and wider land-use influences (European Commission, 2013). In contrast, active raised bogs (7110*) and many expressions of transition mires and quaking bogs (7140) often require closer attention to fine-scale surface patterning, because variation in hummocks, hollows, floating carpets, pools, and associated vegetation can be an important expression of hydrological and ecological integrity (European Commission, 2013). However, these are not strictly separate spatial categories: blanket bogs may also contain important microtopographic patterning, while raised bogs and transition mires must often be understood in their broader hydrological setting. Accordingly, robust assessment should match indicators to the dominant scale of organisation and pressure, combining landscape-scale indicators such as drainage density, burn extent, and hydrological connectivity, with finer-scale indicators such as *Sphagnum* composition, wet microform extent, and local water-table variation.

Given their importance and widespread degradation, systematic assessment and monitoring of peatland conditions have become a core requirement under international and European policy frameworks. Countries increasingly need to know not only where peatlands remain, but also how they are functioning, i.e., whether water tables are still high and stable, whether peat-forming vegetation persists, whether peat is accumulating or is being lost, and how far individual sites have progressed along degradation or recovery trajectories (FAO, 2020; UNEP, 2022). Robust indicators are therefore needed to distinguish intact, vulnerable, and degraded sites, to prioritise restoration and protection, and to follow the effects of management over time.

This chapter addresses inland peat-forming wetlands by treating bogs, fens, and transitional mires together as peatland habitats that share a common set of functional components: a persistently high-water table and characteristic hydrological regime; vegetation dominated by peat-forming mosses and other hydrophilic species; carbon-rich, waterlogged peat soils with a functioning acrotelm; and distinctive microtopography and landscape connectivity. Starting from this functional perspective, the chapter provides a concise, policy-oriented blueprint for assessing peatland conditions. It describes what constitutes a good-functioning state in terms of hydrology, vegetation, soil properties, and landscape context, summarises the main pressures and degradation pathways, and outlines typical ecological response cascades from early alteration to severe and long-lasting decline. Lastly, this chapter introduces a set of candidate indicators of habitat quality and degradation organised around hydrology, vegetation composition and structure, surface morphology and erosion, disturbance and land use, and carbon dynamics and subsidence. For a subset of these indicators, particularly where the evidence base is relatively consistent, such as water table depth or the extent of bare peat, we suggest indicative threshold ranges that can help distinguish early-warning changes from clear signs of degradation or long-term cumulative decline. These thresholds are presented as illustrative and transferable starting points to support qualitative assessments. Developing fully quantified, region-specific thresholds, as well as operational assessment tables, indicator matrices and Earth-Observation-based monitoring tools, will require a dedicated follow-up phase with additional data and calibration.

Overall, the following peatland section is designed to support conservation planning and future technical work by clarifying which ecological components are most critical to track, why they matter for peatland functioning, and how they can be organised into a transparent, qualitative hierarchy of early-warning, urgent, and long-term degradation indicators that can later be refined and operationalized within more detailed monitoring frameworks.

2.2 Definition and interpretation of habitats covered

In this report, peat-forming inland wetland habitats correspond to the mire- and fen-related habitat types under Annex I, Group 7 “Raised bogs, mires and fens” (European Commission, 2013; EUNIS, 2021).

Group 7 is often subdivided into:

- *Sphagnum acid bogs* (7110*, 7120, 7130, 7140, 7150, 7160)
- *Calcareous fens* (7210*, 7220*, 7230, 7240*)
- *Boreal mires* (7310*, 7320*)

These habitats share a common functional basis, and they often occur as mosaic complexes and gradients rather than as isolated units (Fig. 2). Therefore, we do not separate them into sub-groups for assessment; instead, we treat them together as one peat-forming inland wetland habitat group. For clarity and consistency with EU reporting, we list below the 12 Annex I habitat types that make up Group 7, together with brief functional descriptions (and their corresponding EUNIS Level 3 habitat types where applicable) (European Commission, 2013; EUNIS, 2021). However, it is important to note that not all habitats within Group 7 are strictly peat-forming to the same degree. While Group 71 habitats (Active bogs and related habitats: 7110*, 7120, 7130, 7140, 7150) are defined by active peat accumulation under ombrotrophic or minerotrophic conditions, Group 72 habitats (Calcareous fens and petrifying springs: 7210*, 7220*, 7230, 7240*) include systems where peat formation may be limited or absent — notably 7220* (Petrifying springs) and some 7230 (Alkaline fens) sites, where the substrate is more mineral-dominated. This distinction has practical implications for assessment and monitoring: hydrological and peat-specific indicators (e.g. water table depth, peat subsidence, *Sphagnum* cover) are most directly applicable to Group 71 habitats, while Group 72 habitats may require additional or alternative indicators reflecting their distinct hydrochemistry and bryophyte communities. Within Group 71 itself, care should also be taken to distinguish between macrotope-scale assessments (e.g. for 7130 blanket bogs, where catchment-scale hydrology is paramount) and microtope-scale assessments (e.g. for 7110* active raised bogs and 7140 transition mires, where fine-scale water table variation and microtopographic heterogeneity are most critical).

7110* *Active raised bogs* - ombrotrophic, acidic, nutrient-poor bogs sustained mainly by rainwater, typically with a high-water level and *Sphagnum*-dominated hummocks. (**Active:* includes sites still supporting peat-forming vegetation, including cases where peat formation is temporarily stalled, e.g. after fire or during drought cycles).

7120 *Degraded raised bogs are still capable of natural regeneration* - raised bogs with disrupted hydrology (usually anthropogenic) causing surface drying and/or species change; still retain a realistic restoration trajectory where hydrology can be repaired and peat-forming vegetation is expected to re-establish within ~30 years under appropriate management.

7130 *Blanket bogs* (*priority only where “active” bog*) - extensive bog landscapes on flat or sloping ground with poor drainage in oceanic, high-rainfall climates; largely ombrotrophic despite some lateral flow. *Sphagnum* remains important, but sedges/other cyperaceous species are typically more prominent than in raised bogs; priority status applies where peat-forming vegetation remains significant.

7140 *Transition mires and quaking bogs* - peat-forming communities on oligotrophic to mesotrophic water surfaces, intermediate between soligenous (receiving water and nutrients from surface run-off or groundwater) and ombrogenous (receiving water and nutrients from precipitation) types. Often expressed as swaying swards/floating carpets/quaking mires formed by small–medium sedges with *Sphagnum* or brown mosses, frequently accompanied by aquatic and amphibious communities.

7150 *Depressions on peat substrates of the Rhynchosporion* - stable pioneer communities on humid, exposed peat (sometimes sand), forming on stripped/eroded bog surfaces, in flushes, and in the fluctuation zone of oligotrophic pools. Typically includes *Rhynchospora* spp. and sundews (*Drosera*) and is closely linked ecologically to shallow bog hollows and transition mire communities.

7160 *Fennoscandian mineral-rich springs and spring fens* - groundwater-fed springs and spring fens with continuous flow; cold, oxygen- and mineral-rich water due to rapid percolation. Spring fens involve groundwater seeping through peat, supporting specialised vegetation; these systems can remain active in winter and often support specialised invertebrate communities.

7210* *Calcareous fens with Cladium mariscus and species of the Caricion davallianae* - *Cladium mariscus* beds in lake margins, fallows, or successional stages of extensively used wet meadows, typically in contact with *Caricion davallianae* (rich-fen) vegetation and related fen communities.

7220* *Petrifying springs with tufa formation (Cratoneurion)* - hard-water springs with active formation of travertine/tufa, generally small (point/linear) and dominated by bryophytes (*Cratoneurion*). Often embedded within calcareous spring-fen mosaics, and treated here as an associated spring component within peat-forming wetland complexes.

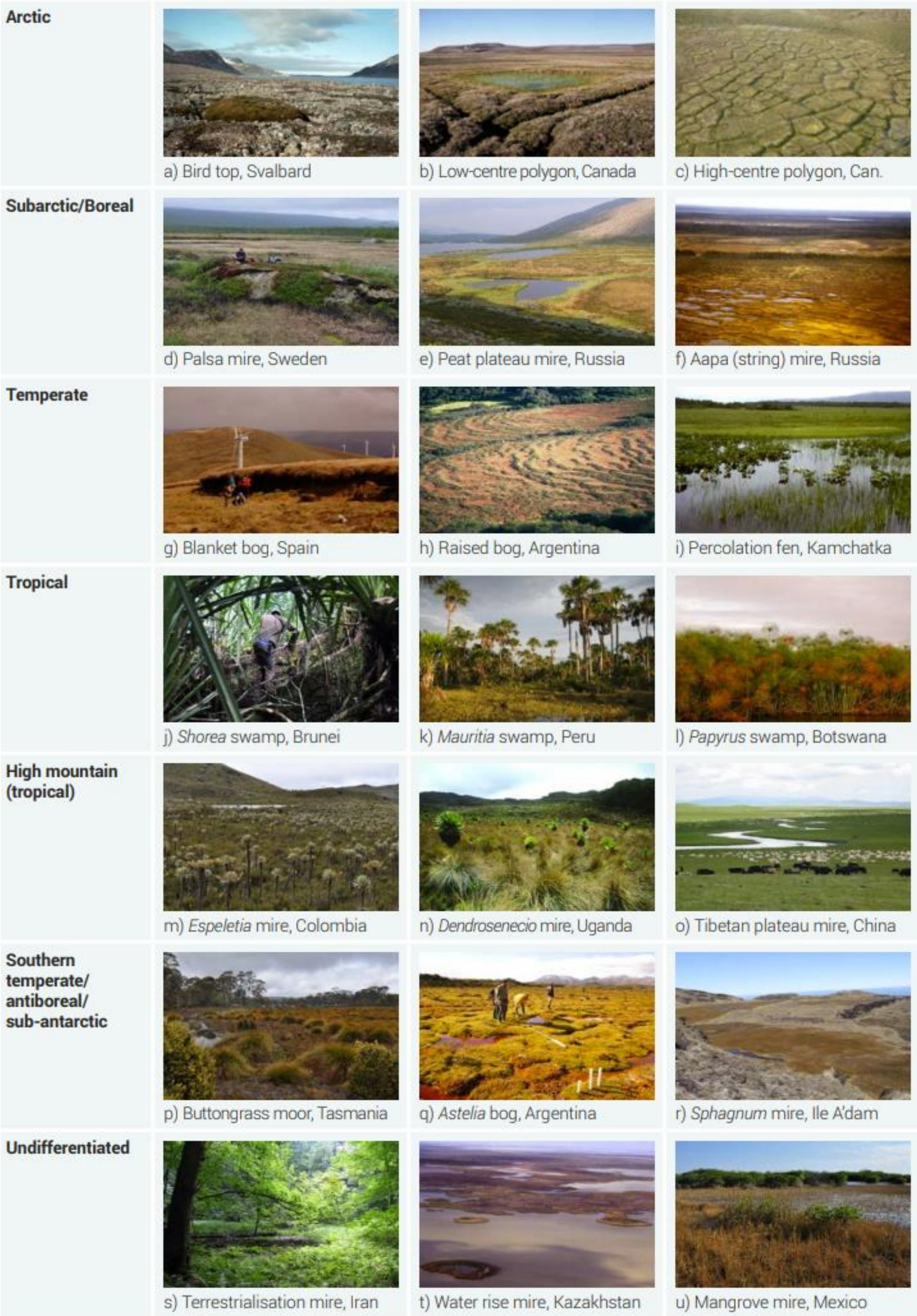
7230 *Alkaline fens* - permanently waterlogged, base-rich (often calcareous) systems dominated by small sedges and brown moss communities, with the water table at/near the surface. Peat (when present) forms under very wet conditions (often infra-aquatic). These fens are typically species-rich and can host many specialised, restricted taxa, but have undergone severe decline in many regions.

7240* Alpine pioneer formations of the Caricion bicoloris-atrofuscae - alpine/subalpine, cold-water-soaked substrates (sometimes peaty) along springs, rivulets, glacial streams and related settings, where long-lasting soil frost is important. Vegetation is low and dominated by *Carex*/*Juncus* communities characteristic of the *Caricion bicoloris-atrofuscae*.

7310* *Aapa mires* - boreal mire complexes characterised by minerotrophic fen vegetation in central parts and a range of hydrotopographical units (e.g. string- and flark-fens, lawns/carpets, mixed mire units). Poor *Sphagnum* fens are common; brown moss fens may be frequent in some areas; mire margins often grade into pine/spruce mire and thin-peat variants.

7320* *Palsa mires* - northern boreal to alpine mire complexes where mean annual temperature is typically below −1°C; mires are mainly minerotrophic except for palsas, which are peat mounds with sporadic permafrost (often a few metres high).

Figure 2. Examples of representative peatland types from different world regions, illustrating the structural, hydrological and ecological diversity of peatlands.



Source: UNEP (2022).

2.3 Ecohydrology and functioning of peat-forming habitats

2.3.1 Ecological and economic importance of peatlands

Peatlands provide a wide range of ecological and economic services that are disproportionate to their spatial extent and are widely recognized as critical components of the Earth system (Ramsar Convention on Wetlands, 2018; Bonn et al., 2014). Ecologically, they are among the most important terrestrial ecosystems for climate regulation, hydrological control, and biodiversity conservation (Ramsar Convention on Wetlands, 2018). Globally, peatlands store more carbon than any other terrestrial ecosystem or soil type, accumulating organic matter over millennia under waterlogged and predominantly anaerobic conditions. It is important to distinguish two related but separate dimensions of this carbon role: (i) the carbon stock: the quantity of carbon already stored per hectare of peatland as accumulated peat, typically hundreds to thousands of tonnes of C ha^{-1}; and (ii) the carbon flux: the net annual rate at which carbon is removed from the atmosphere and added to the peat body per hectare per year, which is much smaller (typically 0.2–1 t C ha^{-1} yr^{-1} in intact systems) but ecologically and climatically significant when aggregated over large areas. Intact peatlands function as long-term carbon sinks, whereas drainage, extraction, or land-use conversion rapidly transform them into significant sources of greenhouse gas emissions, particularly carbon dioxide and nitrous oxide, thereby amplifying climate change (Intergovernmental Panel on Climate Change, 2014).

In addition to their role in climate regulation, peatlands play a fundamental role in hydrological processes at catchment and landscape scales. Their high water-holding capacity allows them to retain precipitation, attenuate flood peaks, and gradually release water, thereby sustaining stream baseflows during dry periods. Peat soils further act as effective natural filters, reducing sediment transport, nutrient loads, and pollutant fluxes, which contributes to erosion control and improved downstream water quality (Holden, 2005).

Peat-forming habitats support distinct biological communities that are strongly structured along hydrological and chemical gradients (in particular pH and base cations) and are characterised by a high degree of ecological specialisation (Vitt et al., 2022). Plant assemblages are composed primarily of species adapted to permanently or periodically waterlogged, nutrient-poor, and often acidic conditions, resulting in clearly differentiated types and a strong filtering of species pools. This ecological specialisation leads to the occurrence of numerous habitat-restricted and regionally rare species, making peatlands important reservoirs of biodiversity at regional scales (Bonn et al., 2011; Vitt et al., 2022). In addition, peatlands provide essential habitats for a range of animal groups, including invertebrates, amphibians and birds, which use these systems for breeding, foraging, or as refugia within intensively managed landscapes (Desrochers & van Duinen, 2006). Owing to their structural heterogeneity and relative isolation, peatlands play a key role in sustaining specialised biotic communities that are poorly represented in other ecosystem types (Bragg & Lindsay, 2003).

Beyond biodiversity support, peatlands play a central role in nutrient cycling and ecosystem connectivity. Slow decomposition rates and high retention capacity lead to efficient nutrient storage and internal recycling, thereby regulating fluxes between terrestrial, aquatic, and atmospheric compartments. Through these processes, peatlands influence biogeochemical dynamics well beyond their immediate spatial boundaries and act as integrative components of landscape-scale ecological networks (Laiho, 2006).

Peatlands also hold substantial cultural, spiritual, and scientific value. The stratified peat matrix preserves pollen, macrofossils, geochemical signals, and archaeological artefacts with exceptional temporal resolution, providing unique archives of past climates, vegetation dynamics, and human land-use history. These long-term records are of high relevance for palaeoecological, archaeological, and Earth system research (Ramsar Convention on Wetlands, 2018).

From an economic perspective, peatlands deliver a broad range of provisioning, cultural, and regulating services. They supply freshwater resources, grazing land, and locally harvested products such as berries, reeds, or biomass. In some regions, peat has historically been extracted for fuel or horticultural use; however, such practices are increasingly recognised as environmentally unsustainable due to their long-term ecological and climatic impacts (IPCC, 2014). Peatlands further provide important cultural services, including opportunities for recreation, education, and nature-based tourism, which are gaining socio-economic relevance in many regions (Bonn et al., 2014).

The regulating services of peatlands are of particularly high economic value. Climate regulation through carbon storage, flood mitigation, water purification, and erosion control generates substantial avoided costs and long-term societal benefits that frequently exceed the short-term economic gains associated with drainage or extractive land use. These benefits are increasingly acknowledged in international policy and assessment frameworks, highlighting peatlands as critical natural assets for climate mitigation, climate adaptation, and sustainable development (IPCC, 2014; Evans et al., 2021).

2.3.2 Geomorphology and hydrology of peat-forming habitats

The initiation, development, and persistence of peat-forming habitats are fundamentally controlled by interactions between geomorphology and hydrology. Peat accumulation typically begins in landscape positions characterised by low relief, impeded drainage, or persistent water saturation, where organic matter production exceeds decomposition. Such conditions commonly arise in depressions, valley bottoms, gently sloping uplands, or areas with limited hydraulic gradients, allowing waterlogged conditions to be maintained over long periods (Evans & Warburton, 2007). The spatial distribution and thickness of peat deposits are therefore closely linked to the pre-existing topography and its influence on surface and subsurface water flow.

Hydrological pathways within peatlands are often inherited from the underlying mineral landscape. Drainage routes, seepage zones, and flow convergence patterns established prior to peat formation may continue to influence water movement after peat accumulation has commenced, shaping both peatland morphology and long-term hydrological stability. This inheritance contributes to spatial heterogeneity within peatlands and affects their sensitivity to disturbance, such as drainage or erosion (Evans & Warburton, 2007).

Based on their dominant water sources and hydrochemical characteristics, peatlands are commonly distinguished into bogs, fens, and transitional mire systems. Bogs are ombrotrophic ecosystems that receive water and nutrients exclusively from precipitation and are therefore typically acidic and nutrient-poor. In contrast, fens are minerotrophic systems that receive groundwater or surface inflows enriched with dissolved minerals, resulting in higher nutrient availability and more diverse chemical conditions. Mires represent transitional forms along this gradient, reflecting varying degrees of hydrological and geochemical influence from surrounding mineral substrates.

Vertical stratification within the peat profile exerts a major control on peatland hydrology and ecosystem functioning. Two principal layers are commonly distinguished: the acrotelm and the catotelm (Ingram 1978; Holden 2005). The acrotelm forms the upper, biologically active layer and is characterised by relatively high hydraulic conductivity, fluctuating water tables, and periodic aeration. This zone supports rapid exchanges of energy and matter and hosts high microbial activity under both aerobic and anaerobic conditions. Beneath the acrotelm lies the catotelm, which constitutes the permanently saturated main body of peat. The catotelm is characterised by very low permeability, persistently anaerobic conditions, and slow rates of decomposition, making it the principal long-term carbon store within peatlands (Holden, 2005). See Table 1.

Table 1. Characteristics of acrotelm and catotelm.

	Acrotelm	Catotelm
Water table	Oscillating	Continuously saturated
Aerobic status	Periodically aerobic	Anaerobic
Moisture content	Variable	Continuously saturated
Hydraulic conductivity	High	Very low
Exchange of energy and matter	Rapid	Slow
Microbial activity	High – aerobic and anaerobic	Low - anaerobic

Source: Adapted from Ingram (1978).

Beyond internal structure, peatland hydrology is strongly influenced by interactions with the surrounding landscape, which affect both water quantity and water quality. Precipitation-derived water is typically mineral-poor and weakly acidic, but its chemical composition may change substantially upon contact with mineral soils or bedrock. The extent of these changes depends on catchment characteristics, including climate, geology, soil properties, vegetation cover, land use, and the residence time of water within the catchment, which is itself controlled by catchment size, permeability, and relief (Holden et al., 2005).

As a result, peatlands may receive water of highly variable chemical composition, and waters of different origin and quality can coexist within different zones of the same peatland. Conversely, similar hydrochemical conditions may arise from distinct hydrogeological settings, underlining the inherent complexity of peatland hydrological systems and the need for site-specific understanding when assessing peatland functioning and vulnerability (Holden & Burt, 2003; Vitt et al., 2022).

2.3.3 Composition and structure of peatland biota

The biotic composition of peat-forming habitats is strongly shaped by hydrological, chemical, and microtopographic gradients, resulting in distinct and highly structured biological communities. Plant assemblages are dominated by species adapted to permanently or seasonally waterlogged, nutrient-poor, and often acidic conditions, leading to a high degree of ecological specialisation and clear differentiation among peatland types. Such environmental filtering restricts species occurrence to narrow ecological niches and produces characteristic vegetation assemblages that differ markedly from those of surrounding mineral-soil ecosystems (Wheeler & Proctor, 2000; Vitt et al., 2022).

Bryophytes, particularly *Sphagnum* species, play a central structural and functional role in many peatlands, especially in ombrotrophic bogs (Fig. 3). Their growth forms, water-retention capacity, and influence on acidity and nutrient availability contribute to the development of distinct microtopographic features such as hummocks and hollows. In minerotrophic fens, vegetation is more strongly influenced by groundwater chemistry and is typically dominated by sedges, grasses, and brown mosses. Across peatland types, vascular plants contribute organic inputs of varying quality, influencing litter decomposition rates and peat properties (Wheeler & Proctor, 2000; Rydin & Jeglum, 2013; Roccio et al., 2025).

Figure 3. Examples of *Sphagnum* species from Romanian peatlands, illustrating the morphological diversity of peat-forming mosses that are important in bog formation and peat accumulation: (A) *Sphagnum medium*; (B) *Sphagnum riparium*; (C) *Sphagnum inundatum* var. *graveții* and *S. denticulatum*.

Source: Adapted from Ștefănuț et al. (2026).

Animal communities in peatlands are similarly structured by hydrological conditions and vegetation patterns. Peatlands provide essential habitats for a range of invertebrates, amphibians, and bird species, many of which are dependent on wet, open, or structurally heterogeneous conditions for breeding, foraging, or refuge. Due to the relative isolation and persistence of peat-forming environments, peatlands often support habitat-restricted and regionally rare fauna, contributing disproportionately to biodiversity conservation at regional scales (Bragg & Lindsay, 2003; Desrochers & van Duinen, 2006).

Belowground, microbial communities play a key role in peatland functioning and are closely linked to redox conditions and substrate quality. Anaerobic conditions prevailing below the water table favour slow-growing microbial assemblages adapted to low oxygen availability, which contribute to reduced decomposition rates and long-term peat accumulation. Vertical differentiation between the acrotelm and catotelm is reflected in strong contrasts in microbial activity and community composition, reinforcing the functional stratification of peatland ecosystems (Ingram, 1978).

Overall, the composition and structure of peatland biota reflect tight coupling between hydrology, vegetation, and soil processes. This close integration results in biologically distinct systems with limited functional redundancy, rendering peatland communities particularly sensitive to changes in water regime, nutrient inputs, or land use (Wheeler & Proctor, 2000; Bragg & Lindsay, 2003).

2.3.4 Key ecological processes in peat-forming habitats

Peatland functioning is governed by a set of tightly coupled ecological processes that regulate organic matter production, decomposition, and long-term carbon accumulation. Primary productivity in peatlands is generally moderate compared to more nutrient-rich ecosystems, but it is persistent over long timescales and dominated by mosses, sedges, and dwarf shrubs adapted to waterlogged and nutrient-poor conditions (Moore, 1989; UNEP, 2022). Peat formation occurs where this sustained organic matter input exceeds decomposition, a balance that is maintained under permanently or seasonally saturated conditions (Holden, 2005; Limpens et al., 2008; Maltby & Barker, 2009).

Decomposition processes in peatlands are strongly constrained by environmental conditions. Low oxygen availability, low temperatures, and often acidic conditions suppress microbial activity, resulting in slow organic matter turnover (Pastor et al., 2002; Wieder & Vitt, 2006; Lindsay, 2010; Andersen et al., 2013a; 2013b; Leifeld & Menichetti, 2018). As a consequence, plant remains are only partially decomposed and accumulate as peat. This imbalance between production and decomposition underpins the long-term carbon sink function of intact peatlands and explains their role as major terrestrial carbon reservoirs (Drollinger et al., 2020; Limpens et al., 2008).

Hydrology plays a central role in regulating these processes and generates strong feedbacks that stabilise peat-forming conditions. The high water-holding capacity of peat soils and peat-forming vegetation helps maintain saturated conditions close to the surface. In many systems, positive feedbacks between a high-water table, dominance of peat-forming mosses (particularly *Sphagnum*), and reduced decomposition rates reinforce peat accumulation. Conversely, hydrological disturbances such as drainage or prolonged drought lower water tables, increase aeration, and can rapidly accelerate decomposition and carbon losses (Holden, 2005; Limpens et al., 2008; Ramchunder et al., 2009; Fluet-Chouinard et al., 2023).

Nutrient cycling in peatlands is characterised by low availability and slow turnover (Moore, 1989; van Breemen, 1995). Limited external nutrient inputs and reduced mineralisation rates result in oligotrophic conditions, particularly in ombrotrophic systems. Despite this, productivity is sustained through efficient internal recycling, close coupling between plants and microbial communities, and the dominance of species with conservative nutrient-use strategies. These mechanisms further contribute to low decomposition rates and long-term peat accumulation (Limpens et al., 2008).

Peatlands exchange greenhouse gases with the atmosphere, primarily carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Intact peatlands typically act as long-term CO_2 sinks, although this uptake is partially offset by methane emissions under anaerobic conditions, in particular during drainage, but also under disturbance or warming conditions (Moore & Knowles, 1989; Maljanen et al., 2010; Szajdak et al., 2020; Gan et al., 2024; Müller et al., 2025). These conditions can alter this balance, often leading to increased CO_2 and N_2O emissions and a shift from net carbon sinks to net carbon sources (Hooijer et al., 2010; Leifeld et al., 2011; Fluet-Chouinard et al., 2023; He et al., 2023; Meļņiks et al., 2023).

At smaller spatial scales, peatland processes are influenced by pronounced microtopographic patterning. Features such as hummocks, hollows, lawns, and pools create fine-scale gradients in water level, aeration, and vegetation composition (Keiser et al., 2025). These patterns emerge through self-organising interactions between hydrology and vegetation and contribute to spatial heterogeneity, habitat diversity, and functional stability within peatland ecosystems (Limpens et al., 2008; Ramchunder et al., 2009; Van der Ploeg et al., 2012; Lindsay et al., 2014a).

Over longer timescales, peatlands undergo successional dynamics driven by peat accumulation, vegetation change, and shifts in hydrological connectivity. In many regions, these processes lead to transitions from minerotrophic fens to increasingly ombrotrophic bog systems, reflecting progressive isolation from groundwater inputs and greater dependence on atmospheric water and nutrient sources (Väliranta et al., 2017).

Peatland resilience depends on the maintenance of these coupled processes. Disturbances such as fire, drainage, permafrost thaw, and land-use change disrupt the balance between production and decomposition and can initiate long-lasting degradation (Dawson et al., 2010; Leifeld et al., 2011;

Grzywna, 2017; Meļņiks et al., 2023; Lu et al., 2025). Recovery is typically slow, as it requires the re-establishment of hydrological conditions and recolonisation by peat-forming species, highlighting the sensitivity of peatland carbon dynamics to environmental change (Holden, 2005; Limpens et al., 2008; IPCC, 2014; Zak & McInnes, 2022).

2.3.5 Good-functioning state

A good-functioning peatland is characterised by the sustained operation of key ecological processes that collectively maintain peat accumulation, carbon storage, and ecosystem stability (UNEP, 2022). Central to this state is a hydrological regime that supports persistently waterlogged conditions. In intact systems, the water table remains close to the surface for most of the year, with seasonal fluctuations largely confined to the acrotelm. These fluctuations do not result in prolonged drying of the catotelm, thereby preserving permanently saturated and anaerobic conditions essential for peat formation (Fluet-Chouinard et al., 2023). Water inputs from precipitation and, depending on peatland type, surface or groundwater sources are balanced by outputs through evapotranspiration and lateral flow, maintaining hydrological equilibrium over long timescales (Ingram, 1978; Holden, 2005; Lamentowicz et al., 2019; Fluet-Chouinard et al., 2023).

Vegetation structure and composition provide a further defining feature of a well-functioning state. Plant communities are dominated by peat-forming species adapted to waterlogged and nutrient-poor conditions, with *Sphagnum* mosses prevailing in ombrotrophic bogs and brown mosses, sedges, and other hydrophilic vascular plants dominating minerotrophic fens (Moore, 1989; Rydin et al., 2006; UNEP, 2022). Vegetation assemblages typically exhibit pronounced microtopographic patterning, such as hummocks, hollows, and lawns, which reflect fine-scale hydrological gradients and contribute to spatial heterogeneity. These plant communities are self-regenerating and support a range of habitat-specialist fauna, indicating functional integrity and ecological resilience (Wheeler & Proctor, 2000; Limpens et al., 2008; Vitt et al., 2022).

Soil and peat properties are closely linked to hydrological and vegetation characteristics. In a well-functioning peatland, peat is waterlogged throughout most of the profile, with a relatively fibrous structure in the acrotelm and increasingly compacted, highly decomposed material in the deeper catotelm. Anaerobic conditions prevail below the water table, resulting in low decomposition rates, high organic matter content, and stable long-term carbon storage (Clymo, 1987). The physical structure of peat promotes high water retention and capillary rise, buffering short-term climatic variability and reinforcing hydrological stability (Ingram, 1978; Holden, 2005; Lindsay et al., 2014b).

Biogeochemical functioning in intact peatlands is characterised by tight internal cycling and low rates of nutrient loss. External nutrient inputs are generally limited, particularly in ombrotrophic systems, and nutrient availability is regulated through slow mineralisation and efficient internal recycling. Under natural conditions, long-term carbon sequestration exceeds carbon release, maintaining peatlands as net carbon sinks. Although methane emissions occur as a result of anaerobic decomposition, these are typically balanced over longer timescales by sustained carbon dioxide uptake associated with peat accumulation (Lai, 2009; Limpens et al., 2008; IPCC, 2014; Gan et al. 2024).

A good-functioning peatland is also embedded within, and connected to, its surrounding landscape. Hydrological and biogeochemical exchanges with the catchment remain largely unaltered, allowing appropriate water inputs and maintaining characteristic chemical conditions. Intact transition zones to mineral soils or adjacent wetland types, as well as buffer areas with low-intensity land use, help regulate inflows, reduce external pressures, and sustain biodiversity and ecosystem functioning at the landscape scale (Holden, 2005; Evans & Warburton, 2007).

Finally, resilience and self-regulation are defining attributes of a good-functioning state. Intact peatlands are capable of maintaining their key ecological processes and recovering from moderate natural disturbances such as short-term droughts, flooding, or grazing. This resilience is underpinned by feedbacks between hydrology, vegetation, and peat accumulation that stabilise system functioning over long timescales. Disruption of these feedbacks, particularly through sustained drainage or land-use change, compromises this self-regulating capacity and leads to long-lasting degradation (Holden, 2005; Limpens et al., 2008).

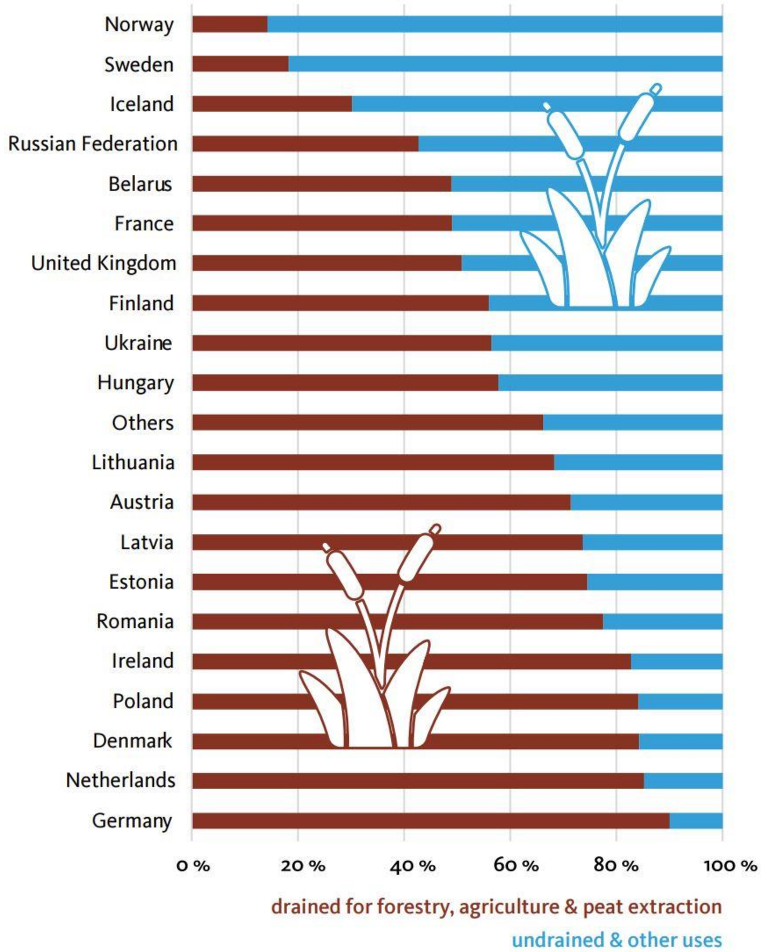
2.4 Primary pressures and threats

Peatlands are shaped by long-term internal dynamics (succession, microtopographic development, disturbance-recovery cycles) and by external drivers. In a conservation and restoration context, pressures and threats are typically framed as direct anthropogenic pressures (e.g. drainage, conversion, extraction, burning, nutrient inputs), while climate change acts as a pervasive indirect driver that increasingly amplifies hydrological stress, fire risk, and vegetation shifts (Renou-Wilson & Wilson, 2018; UNEP, 2022). To reflect this, the subsections below cover the dominant direct pressures, and a final subsection explicitly addresses climate-driven and quasi-"natural" stressors and feedbacks.

2.4.1 Hydrological alteration and drainage

Hydrological alteration, particularly drainage for agriculture, forestry, peat extraction and infrastructure, is widely recognized as the single most important direct driver of degradation in peatlands (Andersen et al., 2017; Ramsar Convention on Wetlands, 2018; Swindles et al., 2019) (Fig. 4; Fig. 5). Drainage ditches, deepened channels, and lowered outlet levels reduce the height and stability of the water table, exposing previously saturated peat to oxygen and triggering accelerated aerobic decomposition (Yang et al., 2025). Peatland drainage leads to subsidence and compaction (Dawson et al., 2010; Leifeld et al., 2011; Grzywna, 2017), and largely irreversible changes in the physico-chemical properties of peat, including increased bulk density (Wallor et al., 2018) and higher degrees of humification (Szajdak et al., 2020). As a result, drained peatlands retain less water and generate higher and more rapid downstream runoff, amplifying flood peaks and diminishing their drought-buffering function at the catchment scale.

Figure 4. Status of peatlands in selected European countries, showing the proportion of peatland area drained for forestry, agriculture and peat extraction versus undrained and other uses.



Source: UNEP (2024).

Figure 5. Drains cut across blanket bog in the Flow Country, northern Scotland.

Source: Royal Society for the Protection of Birds (RSPB).

From a biogeochemical perspective, drainage converts peatlands from net carbon sinks to net greenhouse gas (GHG) sources (Müller et al., 2025). This occurs through the lowering of the water table, exposing previously anoxic peat layers to oxygen, and leading to the release of centuries of stored carbon as carbon dioxide (CO_2) and nitrous oxide (N_2O) (Armentano & Menges, 1986; Maljanen et al., 2010; Tiemeyer et al., 2016; Yang et al., 2025). Although drained organic soils occupy only a small share of the agricultural land base, they account for a disproportionate fraction of total agricultural GHG emissions in several EU Member States. At EU scale, drained organic soils represent only around 3% of the agricultural land area but are estimated to generate up to a quarter of agricultural GHG emissions, underlining the high mitigation leverage of rewetting and changed management on peat soils (UNEP, 2022). However, while rewetting is often highly effective for reducing CO_2 emissions from aerobic peat decomposition, ecological recovery and wider biogeochemical outcomes depend strongly on site conditions and management (e.g. nutrient legacy, hydrological connectivity, vegetation trajectory). Drainage also increases the mobilisation of heavy metals (Rothwell et al., 2006) and nutrients such as phosphorus (P) and DOC from degraded peat layers (Tiemeyer & Kahle, 2014), with significant implications for adjacent aquatic systems and water resource management.

Hydrological pressures are particularly acute for fens, whose water and nutrient status depend on the quality and quantity of groundwater and surface inflows (Ramsar Convention on Wetlands, 2018). River regulation, embankment construction, floodplain disconnection, hydropower development, and groundwater abstraction can all reduce inundation frequency and baseflows, leading to fen desiccation, succession to meadow or woodland, and loss of fen-specific flora and fauna (Wassen et al., 1990; Günther et al., 2020; Grodzka-Łukaszewska et al., 2022). At catchment scale, water abstraction, channelisation, and dam construction accelerate runoff and shorten residence times. Conversely, dam construction can impound water and alter downstream flow regimes, potentially cutting off the hydrological connectivity that minerotrophic fens depend upon, reducing the capacity of bogs and fens to buffer both floods and droughts.

2.4.2 Land conversion

Land conversion for agriculture and forestry is closely connected to drainage and represents another major pressure on peatlands highlighted by the Ramsar assessments. The demand for productive land has driven widespread transformation of peatlands into pastures, intensively managed meadows, arable fields and forest plantations, particularly in accessible lowland and valley peatlands (Ramsar Convention on Wetlands, 2018; UNEP, 2022). In many contexts, this includes plantations, but drainage can also facilitate spontaneous tree establishment and secondary succession, shifting open mires toward scrub and woodland even without deliberate planting.

In fens, conversion to improved grassland or arable land typically follows decades of ditching and fertilisation. This process replaces peat-forming vegetation with nutrient-demanding grasses and crops (Tiemeyer et al., 2016). On converted meadows or arable land, repeated fertilizer applications and mechanical disturbance substantially alter peat physico-chemical properties (Tiemeyer et al., 2016; UNEP, 2022; Deng et al., 2025), further accelerating peat mineralisation and nutrient release.

Historically, fen conversion in parts of Europe began as early as the Middle Ages, in some cases creating semi-natural, traditionally managed, species-rich grasslands under relatively low-intensity use. However, the most extensive and ecologically damaging phases often occurred later, during the 19th century and early 20th century, and again after World War II (notably from the 1960s/70s through the 1980s), as effective deep drainage and pumping systems enabled sustained lowering of water tables and facilitated widespread agricultural intensification (Fig. 6). This shift typically moved systems from semi-natural or moderately improved grasslands toward more intensively managed and strongly degraded peatlands.

In bogs, afforestation (primarily with coniferous species) usually requires sustained lowering of the water table and often deep ploughing, both of which disturb peat structure and alter nutrient dynamics to considerable depth (Hancock et al., 2018). Afforestation increases peat decomposition and nitrogen availability, with knock-on effects on GHG fluxes, nutrient export, and biodiversity (Sloan et al., 2018).

These land-use changes increase GHG emissions, including CO_2 and N_2O from drained peat soils (Armentano & Menges, 1986; Martikainen et al., 1993; Maljanen et al., 2010; Lin et al., 2022; Wang et al., 2024), while drainage networks and ditches can be significant sources of methane (CH_4) (Peacock et al., 2021) (Fig. 6).

Figure 6. Environmental changes following conversion of tropical peat swamp forest to oil palm, showing lower groundwater levels, higher soil temperature, and increased net ecosystem methane exchange (NEE–CH_4) across the pre-conversion, land clearing, and young oil palm plantation phases.

	Before conversion (peat swamp forest) January 2014 – February 2017 	During conversion (land clearing) March 2017 – April 2018 	After conversion (young oil palm plantation) May 2018 – December 2020 
Groundwater level (cm)	-20.0 ± 14.2	-102.3 ± 31.6	-96.5 ± 19.3
Soil temperature (°C)	26.5 ± 0.6	28.8 ± 1.0	29.6 ± 0.6
NEE-CH_4 ($mg\ C\ m^{-2}\ d^{-1}$)	12.1 ± 5.3	23.3 ± 8.6	16.3 ± 4.1

Source: Adapted from Wong et al. (2025).

2.4.3 Peat extraction

Peat extraction, historically for fuel and more recently for horticultural substrates, is a highly destructive but spatially more localized pressure (Ramsar Convention on Wetlands, 2018; UNEP, 2022). Industrial extraction fields and traditional cutover areas remove the acrotelm and substantial portions of the catotelm, effectively eliminating peat accumulation and disrupting the hydrological and ecological integrity of bogs and other peatland types (Price & Ketcheson, 2009; Zając et al., 2018). Although extraction sites cover a relatively small proportion of total peatland area, they represent a near-total loss of habitat. Even after extraction ceases, extensive drainage networks and bare peat surfaces continue to promote oxidation and erosion, so that these landscapes emit GHGs for decades and remain highly erodible (Rankin et al., 2023).

2.4.4 Burning and overgrazing

In many upland and blanket bog landscapes, managed burning and grazing are long-standing traditional uses, and may be considered under the Ramsar *wise use* concept when appropriately timed and limited in intensity (Ramsar Convention on Wetlands, 2018). When carried out too frequently or intensively, however, they become major drivers of degradation. Drying and repeated burning remove the protective moss and litter layer, exposing bare peat that is highly susceptible to wind and water erosion. In drained peatlands, fires can extend into the peat itself, causing deep combustion, long-lasting smoke episodes, permanent subsidence, and severe loss of soil volume (Rein et al., 2008; Khakim et al., 2020), as well as increased CO_2 fluxes (Ward et al., 2007).

Overgrazing can suppress regeneration of key peat-forming species and alter peat physical structure through trampling (Ward et al., 2007). Trampling-driven compaction can reduce microtopographic relief and homogenize microsites (e.g. flattening hummock-hollow patterns), which in turn reduces habitat diversity and shifts hydrological niches available for peat-forming plants. Compactions can also promote faster runoff and erosion, further weakening peatland water retention (Ward et al., 2007).

2.4.5 Atmospheric deposition

Peatlands, especially nutrient-poor ombrotrophic bogs, are highly sensitive to atmospheric inputs of biologically reactive nitrogen and acidifying compounds (Ramsar Convention on Wetlands, 2018). Excess nutrient loading disrupts the nutrient balance of bogs and fens, promoting the displacement of peat-forming *Sphagnum* mosses by non-peat-forming, more nutrient-demanding vascular plants and thereby reducing the capacity to accumulate peat (Chapin et al., 2003; Bragazza et al., 2006). Long-term nutrient-addition experiments in ombrotrophic bogs show that these vegetation shifts, together with higher nutrient availability, weaken the peatland carbon sink by reducing net CO_2 uptake and enhancing ecosystem respiration; under high Nitrogen, Phosphorus and Potassium (NPK) loads they can also raise water tables through peat subsidence and substantially increase CH_4 emissions (Larmola et al., 2013; Juutinen et al., 2018). In much of western and central Europe, chronic nitrogen deposition has already altered nutrient balances in ombrotrophic bogs and minerotrophic fens, favouring competitive grasses and shrubs over specialist peatland species, and further eroding peat-forming vegetation.

Fens, which depend on groundwater and surface-water inputs, are additionally exposed to nutrient enrichment from the wider catchment, making eutrophication from agricultural fertilizers and effluents particularly damaging. Nutrient enrichment promotes expansion of tall-growing, highly productive species (e.g. sedges and reeds) at the expense of slow-growing fen specialists and bryophytes adapted to low-nutrient conditions, changing habitat structure and reducing biodiversity (Bollens et al., 2001; Bobbink et al., 2010; Cusell et al., 2014).

Beyond nutrients, atmospheric contaminants such as heavy metals and persistent organic pollutants can accumulate in peat (Rothwell et al., 2006; Tiemeyer and Kahle, 2014). Importantly, this is not only a 'problem under degradation', since intact peatlands can function as long-term sinks for some contaminants, contributing to regulation of water quality. Under drainage, oxidation, and erosion, however, stored contaminants and DOC can be remobilized and exported to downstream waters (Rothwell et al., 2006; Tiemeyer and Kahle, 2014).

Historically, the magnitude and ecological influence of atmospheric deposition intensified markedly with industrialisation and the expansion of energy production and intensive agriculture. In many regions, nitrogen deposition became a dominant pressure from the 1970s onward and increased further through the 1990s and into the 21st century, compounding existing drainage and land-use impacts. Earlier phases of deposition were often dominated by acidifying compounds and heavy metals associated with industrial emissions, contributing to ecological collapse and severe erosion in some upland and blanket bog systems (e.g. in northern England, a core region of the Industrial Revolution), with slow recovery and long-lasting legacy effects.

2.4.6 Climate change and natural stressors

In addition to direct anthropogenic pressures, peatlands are increasingly threatened by climate change, which acts primarily by altering hydrological regimes (evapotranspiration, drought frequency, snow dynamics, and seasonal water balance), and by increasing the probability of extreme events, e.g. heatwaves and wildfire-conducive conditions. A key ecological consequence of hydrological stress – whether caused by climate change, drainage, or their combination – is vegetation change. Drying favours drought-tolerant shrubs, grasses and trees over peat-forming mosses, contributing to the replacement of *Sphagnum*-rich communities by dwarf shrubs or forest, and contraction of open mire habitats (Lu et al., 2025), with cascading effects on faunal assemblages dependent on wet peatlands (Berg et al., 2009). Tree and shrub encroachment can occur as secondary succession following drainage or burning but can also be accelerated by warmer and drier conditions. Once established, woody vegetation can further exacerbate drying through elevated evapotranspiration and hydrological stress. This feedback is especially important because it can lock peatlands into alternative stable states that are difficult to reverse without sustained hydrological intervention and active revegetation measures. It is worth noting that once a former peatland transitions to mature forest, the new woody system may eventually become a net carbon sink again over decadal timescales; however, this occurs only after a prolonged period of substantial carbon release, and the peatland-specific carbon stock, hydrology, and biodiversity are not recovered.

2.5 Degradation pathways

Peatland degradation is rarely caused by a single pressure acting in isolation. Human interventions typically trigger cascades of linked ecological responses that spread through hydrology, vegetation, soils, GHG fluxes and water-storage capacity over years to decades. Understanding these cascades is key to locating a site along a degradation trajectory, deciding which indicators are most informative at each stage, and judging when restoration is still cost-effective and when damage becomes difficult or impossible to reverse.

Across European peat-forming wetlands, four major and often interacting degradation cascades can be distinguished:

1. Hydrological drainage → drying/oxidation → compaction/subsidence → increased fire and erosion risk
2. Nutrient enrichment and atmospheric deposition → shifts in vegetation and microbial community structure → structural and functional change
3. Physical disturbance → bare peat → erosion and secondary drainage
4. Drying → fuel build-up → fire → rapid carbon loss

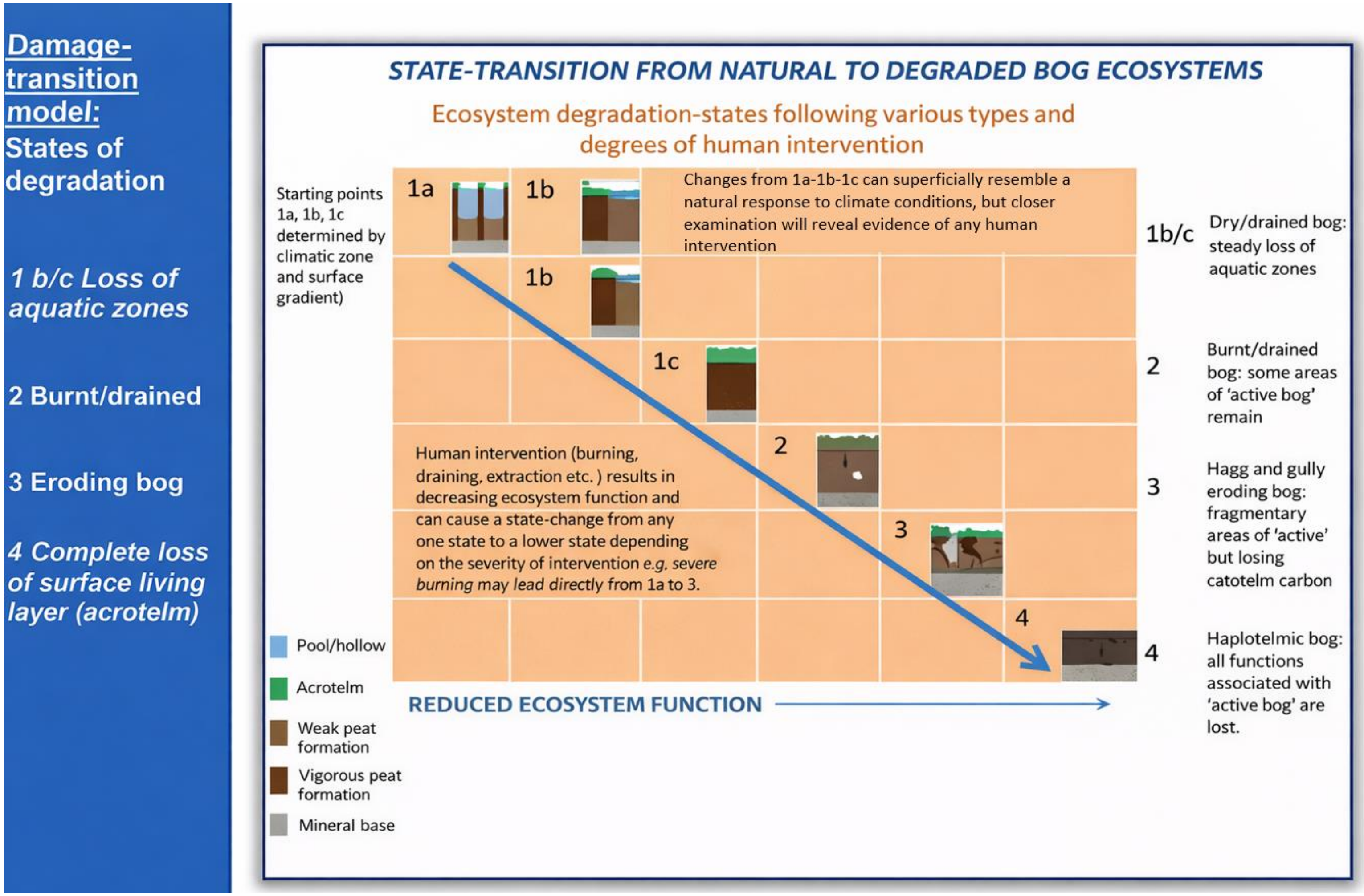
These cascades are driven primarily by hydrological alteration (ditches, channel deepening, outlet lowering, groundwater abstraction, gully incision), compounded by land-use change and management intensity (forestry, agriculture, peat extraction, grazing pressure and trampling), nutrient inputs, and climate stress (droughts, heatwaves, extreme rainfall) (Lindsay et al., 2014a; Leifeld & Menichetti, 2018; UNEP, 2022). Together they produce characteristic structural, hydrological and biogeochemical changes that can be captured by Earth Observation (EO) and complemented by *in situ* functional indicators (e.g. water-table measurements, DOC/POC, $\delta^{15}N$, ^{14}C-DOC) (Trippier et al., 2020).

For the dominant hydrological drainage cascade, three broad degradation stages can be recognized with indicative timeframes:

- Stage 1 (~0–5 years after drainage; often reversible);
- Stage 2 (~5–20 years; recovery possible but slower and costlier);
- Stage 3 (>20 years; often severe or only partly reversible without major restoration).

Each stage corresponds to a different mix of early-warning indicators (subtle shifts in water table and vegetation), immediate/urgent degradation indicators (extensive bare peat, entrenched drainage, recent fires), and long-term cumulative indicators (subsidence, bulk-density increase, land-use conversion) (Fig. 7). EO will mainly be used as an “alarm system” for these stages; the indicator hierarchy and thresholds are developed in the following section. A detailed example is provided for the hydrological drainage → oxidation → subsidence degradation pathway (Section 2.5.1.)

Figure 7. Damage-transition model illustrating the state-transition from natural to degraded bog ecosystems under increasing human intervention, from near-natural starting states (1a–1c) through loss of aquatic zones (1b/c), burnt/drained bog (2), hag and gully erosion (3), and ultimately haplotelmic bog (4) with complete loss of the living surface layer (acrotelm), alongside reduced ecosystem function.



Source: Lindsay et al. (2014b).

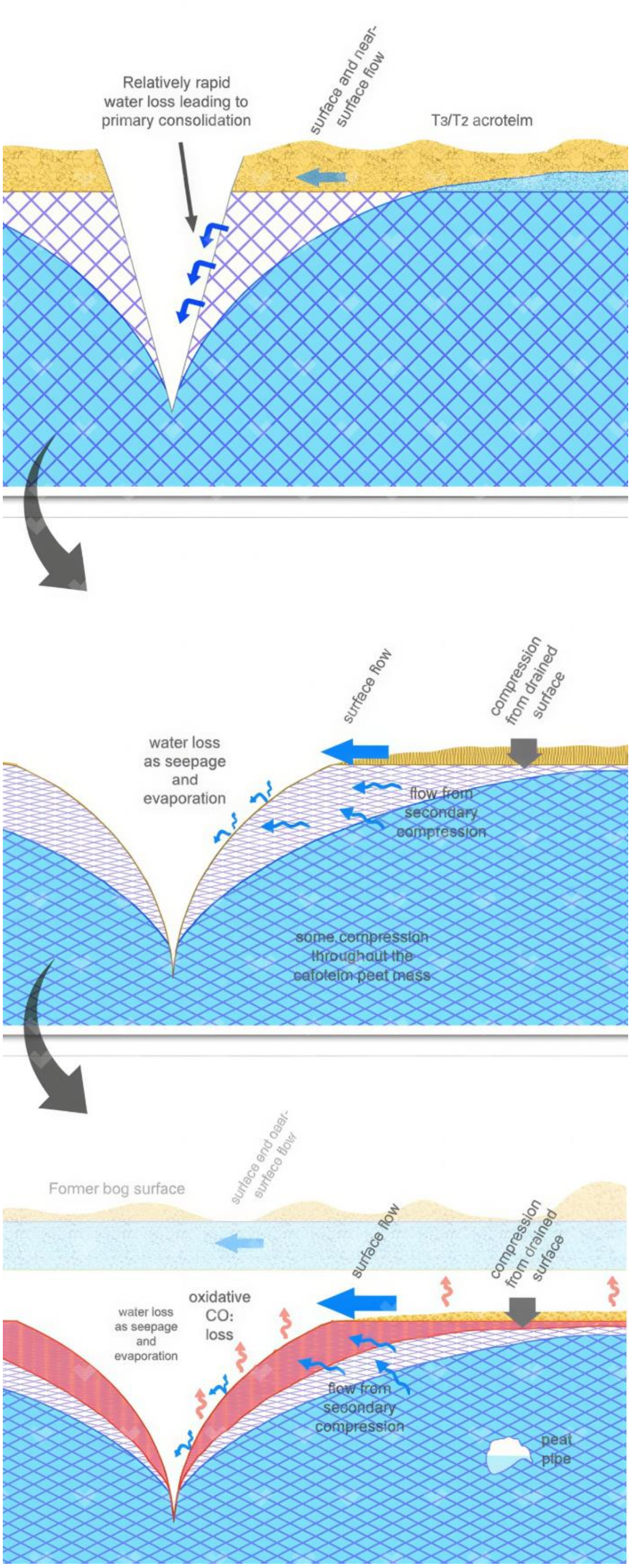
2.5.1 Degradation pathway: Hydrological drainage → oxidation → subsidence

Hydrological drainage is the dominant degradation pathway in peat-forming wetlands, initiated by pressures such as drainage ditches, channel deepening, outlet lowering, groundwater abstraction and forestry/agricultural drainage linked to conversion of peat to cropland, pasture and plantations (Lindsay, 2010; Lindsay et al., 2014a; Leifeld & Menichetti, 2018; UNEP, 2022). These interventions lower the water table from near-surface conditions to ~10–20 cm, and eventually >30 cm, below ground for long periods, aerating the formerly saturated acrotelm and upper catotelm (Fig. 8).

Aerobic decomposition accelerates, CO_2 (and often N_2O) emissions rise, and the system shifts from a net carbon sink to a net source (Moore, 1989; Moore & Knowles, 1989; Hooijer et al., 2010; UNEP, 2022). Peat shrinks and compacts, and transitions towards more mineral-like soil, with bulk density increasing, and from centimeters to tens of centimeters of surface subsidence accumulating over decades (McCarter & Price, 2015; Word et al., 2022). As storage capacity declines, peatlands lose their “sponge” function, with reduced baseflow support, flashier runoff, and higher downstream flood peaks and associated water-quality impacts (e.g. increased DOC/POC and nutrient export, water colour) (Lindsay et al., 2014a; Leifeld & Menichetti, 2018).

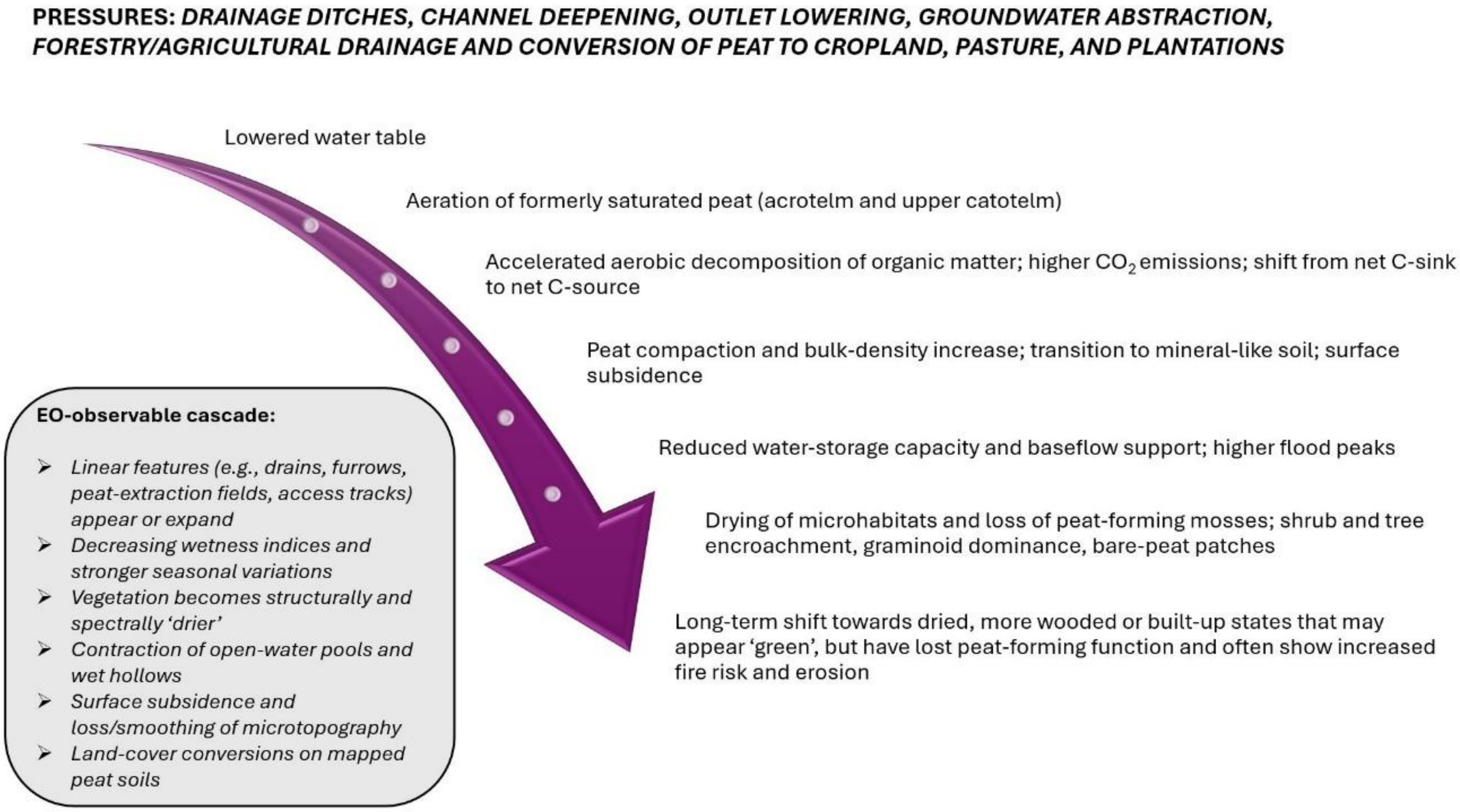
At the surface, hollows, lawns and pools dry out; peat-forming *Sphagnum* and brown mosses are progressively replaced by graminoid dominance, shrub and tree encroachment, and bare-peat patches. These changes greatly increase fire risk and erosion and tend to lock the system into a degraded hydrological and carbon state (Lindsay et al., 2014a; UNEP, 2022) (Fig. 9). In lowland agricultural peat, continued drainage and oxidation cause severe subsidence, sometimes bringing peat surfaces below surrounding rivers or sea level, with long-term dependence on pumping and heightened flood and salinisation risk.

Figure 8. Schematic representation of peat bog response to artificial drainage, illustrating primary consolidation, secondary compression, and oxidative wastage as linked processes driving long-term peat subsidence and carbon loss.



Source: Lindsay et al. (2014a).

Figure 9. Hydrological drainage – oxidation – subsidence cascade in peat-forming wetlands. Pressures from artificial drainage and land-use conversion lower the water table, trigger peat aeration, oxidation, compaction, and subsidence, reduce water-storage and flow-regulation capacity, and drive vegetation shifts that increase fire and erosion risk. The EO-observable cascade on the left highlights the main remote-sensing signals (drainage features, surface wetness changes, vegetation structural shifts, subsidence, bare peat) that can act as early-warning and degradation indicators along this pathway.



Source: Own elaboration.

While the detailed trajectory is site-specific, the drainage cascade in many European peatlands can be described in four broad stages:

Stage 0 – Intact / near-intact state

Hydrology: Water table fluctuates within a narrow band close to the surface, typically around 0–10 cm below ground for much of the year in bogs and many poor fens; seasonal amplitude is modest.

Soils: Peat is saturated, with a well-developed acrotelm–catotelm structure; bulk density is low and porosity high.

Vegetation: Dominated by peat-forming mosses (e.g. *Sphagnum*) and specialist fen species; shrub and tree cover is low in open mire types; hummock–hollow microtopography and pools are well expressed.

Function: The system acts as a net carbon sink and maintains strong water-storage and flow-regulation functions, damping both floods and droughts.

Stage 1 – Early drainage and vegetation shift (~0–5 years; often reversible)

Typical pressures: installation of shallow drains or small ditches, local outlet lowering, small-scale land levelling or road construction on peat.

Hydrology: Mean water table drops by roughly 10–30 cm compared to intact conditions; periods with water table at or above the surface shorten; seasonal amplitude increases.

Soils: The acrotelm dries more frequently; upper peat layers become warmer and more oxygenated, initiating peat oxidation but with limited depth penetration.

Vegetation: Sensitive mosses and specialist bog/fen species start to decline in hollows and pool margins; graminoids and dwarf shrubs gain a competitive advantage, but peat-forming communities are still present and microtopography is largely intact.

Function: Water-storage capacity is slightly reduced, and early increases in DOC and nutrient export can occur, but the peatland may remain a weak carbon sink or near-neutral.

Stage 2 – Prolonged low water table, peat oxidation and functional decline (~5–20 years; recovery possible, but slower and costlier)

Pressures: deeper or denser drainage networks, channel straightening, continued outlet lowering, intensification or expansion of forestry and productive grassland, sometimes fertilisation and liming, plus cumulative effects of prolonged droughts.

Hydrology: Mean water table typically >30–50 cm below the surface for large parts of the growing season; seasonal amplitude can exceed 40–50 cm. Deep summer drawdowns become common.

Soils: Extended exposure to oxygen accelerates decomposition and CO_2 emissions, with rising N_2O in fertilized systems; peat shrinks, consolidates and compacts, leading to measurable subsidence (often 1–3 cm yr^{-1} in drained temperate/boreal systems) and formation of moorsh-like horizons in the upper peat.

Vegetation: Peat-forming mosses and fen specialists decline strongly or disappear from large parts of the surface; more productive sedges, grasses, reeds, shrubs and trees dominate; open mires transition towards scrub or plantation forest; hummock–hollow structure collapses as hollows and pools infill or dry out.

Function: Water-storage capacity is substantially reduced, hydrographs become “flashier”, downstream flood peaks rise and arrive faster, and baseflow support during dry periods weakens. DOC, POC and nutrient export increase, often causing water colour and treatment problems downstream. The peatland usually becomes a consistent net CO_2 source.

Stage 3 – Severe degradation, subsidence, fire and regime shift (>20 years; often only partly reversible)

Pressures: deep, long-standing drainage, peat extraction, decades of intensive agriculture on peat, repeated burning, and sometimes external groundwater drawdown (e.g. from plantations or abstractions).

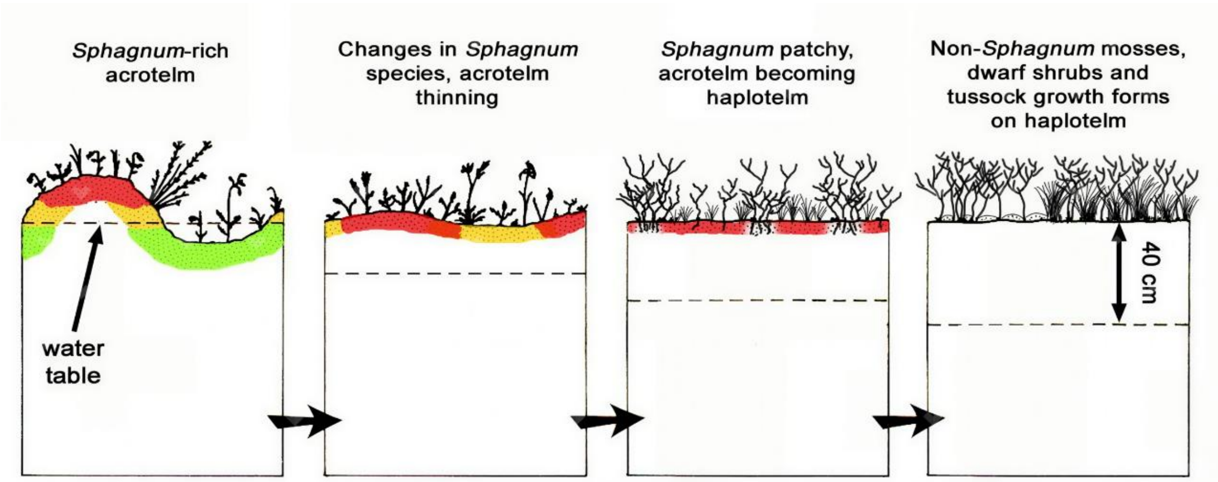
Hydrology: Large parts of the peat profile remain unsaturated for extended periods; water tables may fall >50–100 cm below the surface in summer; dense ditch and canal networks efficiently export water and can disconnect peat from former groundwater supplies.

Soils and peat mass: Upper peat is strongly decomposed and often moorsh-like; deeper peat is progressively oxidized or physically removed. Pronounced subsidence lowers surfaces relative to surrounding land and water bodies; in extreme cases peatlands subside below sea level or adjacent rivers, creating dependence on pumping and raising salinisation risk.

Vegetation: Peat-forming vegetation is largely absent; land cover is dominated by non-peatland communities (intensive grassland, cropland, plantations) or highly degraded secondary communities with extensive bare peat, erosion gullies and collapsed microtopography (Fig. 10).

Function: The peatland’s role as a water-regulation “sponge” is largely lost; systems behave more like rapid through-flow channels, amplifying both floods and droughts. Drained peatlands in this stage act as long-term CO_2 sources, with large cumulative emissions and sustained DOC, POC and nutrient export. Fire susceptibility is very high, and repeated fires and erosion further entrench the degraded state.

Figure 10. Conceptual illustration of vegetation and microtopographic change in a bog following artificial drainage, showing progressive thinning and loss of the *Sphagnum*-rich acrotelm, increasing dominance of non-*Sphagnum* mosses, dwarf shrubs and tussock-forming species, and the transition from peat-forming surface conditions toward a degraded haplotelm.



Source: Lindsay et al. (2014a).

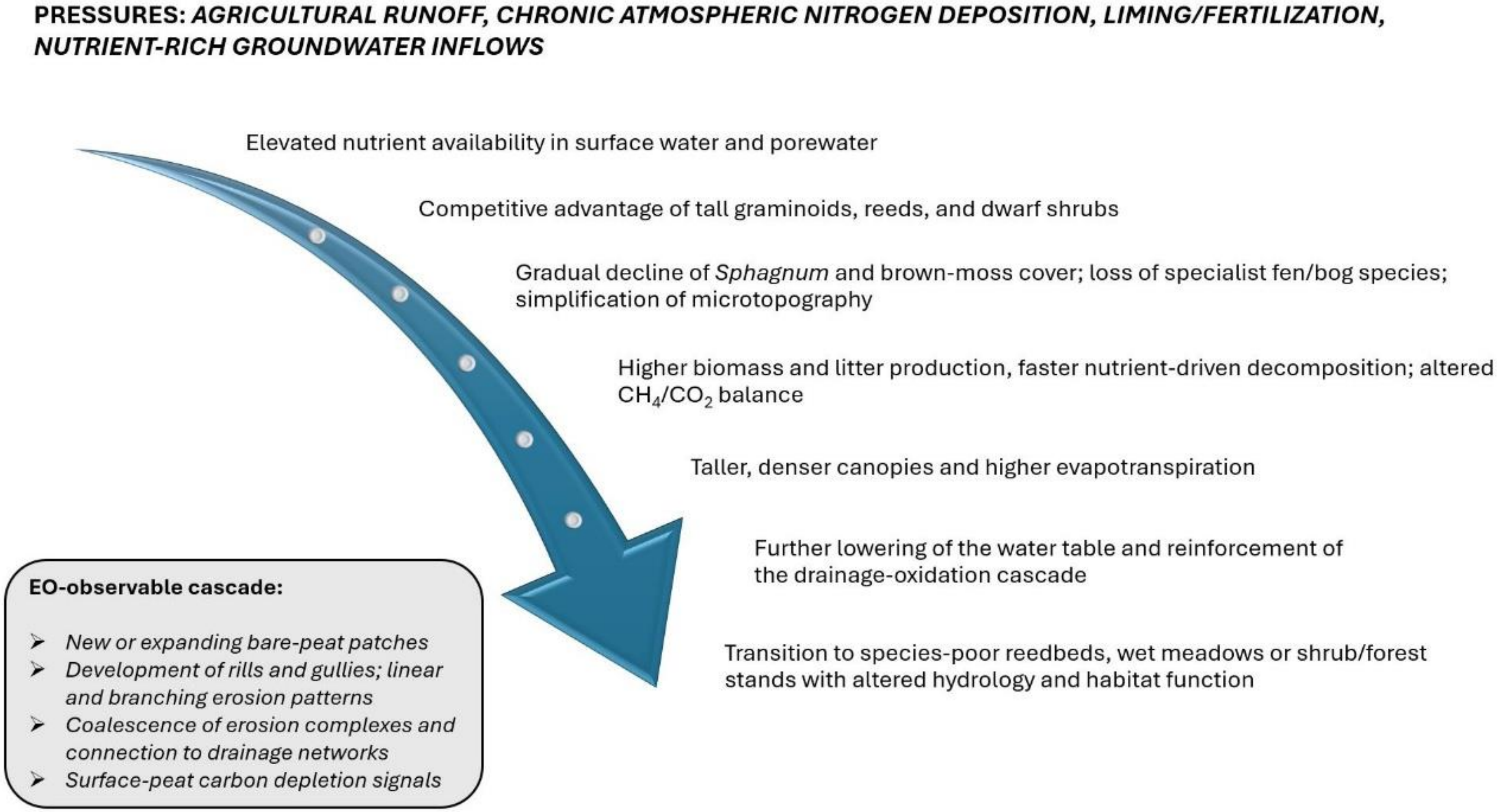
2.5.2 Degradation pathway: Nutrient enrichment and atmospheric deposition → vegetation and microbial shift → structural change

Nutrient enrichment acts as a second major degradation pathway in peat-forming wetlands, often superimposed on hydrological drainage (UNEP, 2022). Inputs from agricultural runoff, chronic atmospheric nitrogen deposition, liming and fertilisation, or nutrient-rich groundwater elevate nutrient availability in surface waters and peat porewater. This gives tall graminoids, reeds and dwarf shrubs a strong competitive advantage over peat-forming *Sphagnum* and brown mosses (Harmens et al., 2014; Vesala et al., 2021; UNEP, 2022).

Over time, this drives a gradual decline of moss carpets, simplification of hummock–hollow microtopography and loss of specialist fen and bog species, while biomass and litter inputs increase. In parallel, microbial communities and enzyme activity typically shift towards faster, more nutrient-driven decomposition, altering pathways of carbon and nutrient cycling and contributing to changes in the CH_4/CO_2 balance and reduced long-term carbon accumulation (Gunnarsson & Rydin, 2000; Heijmans et al., 2002; Malmer et al., 2003; Tolunay et al., 2024). As canopies become taller, denser and more productive, evapotranspiration increases, further lowering the water table and reinforcing the drainage–oxidation cascade. In advanced stages, peatlands transition to reedbeds, wet meadows or shrub/forest stands with strongly modified hydrology and habitat function (Lindsay et al., 2014a).

Along degradation gradients, changes in nitrogen cycling can be detected via $\delta^{15}N$ in soils and vegetation. In many cases $\delta^{15}N$ becomes more enriched (higher) where systems are more 'open' and leaky under dried, nutrient-rich conditions, reflecting altered sources and fractionating loss pathways. In very degraded soils, $\delta^{15}N$ can shift again depending on nutrient depletion and changing dominant processes, so direction should be interpreted in context (Fig. 11).

Figure 11. Nutrient enrichment and atmospheric deposition cascade in peat-forming wetlands. Chronic nutrient inputs shift competitive balance towards tall graminoids, reeds and shrubs, causing loss of *Sphagnum* and brown mosses, simplification of microtopography, higher biomass and nutrient-driven decomposition, enhanced evapotranspiration and ultimately transition to species-poor, structurally altered vegetation states that reinforce hydrological drying and carbon loss, with characteristic EO-detectable signals.



Source: Own elaboration.

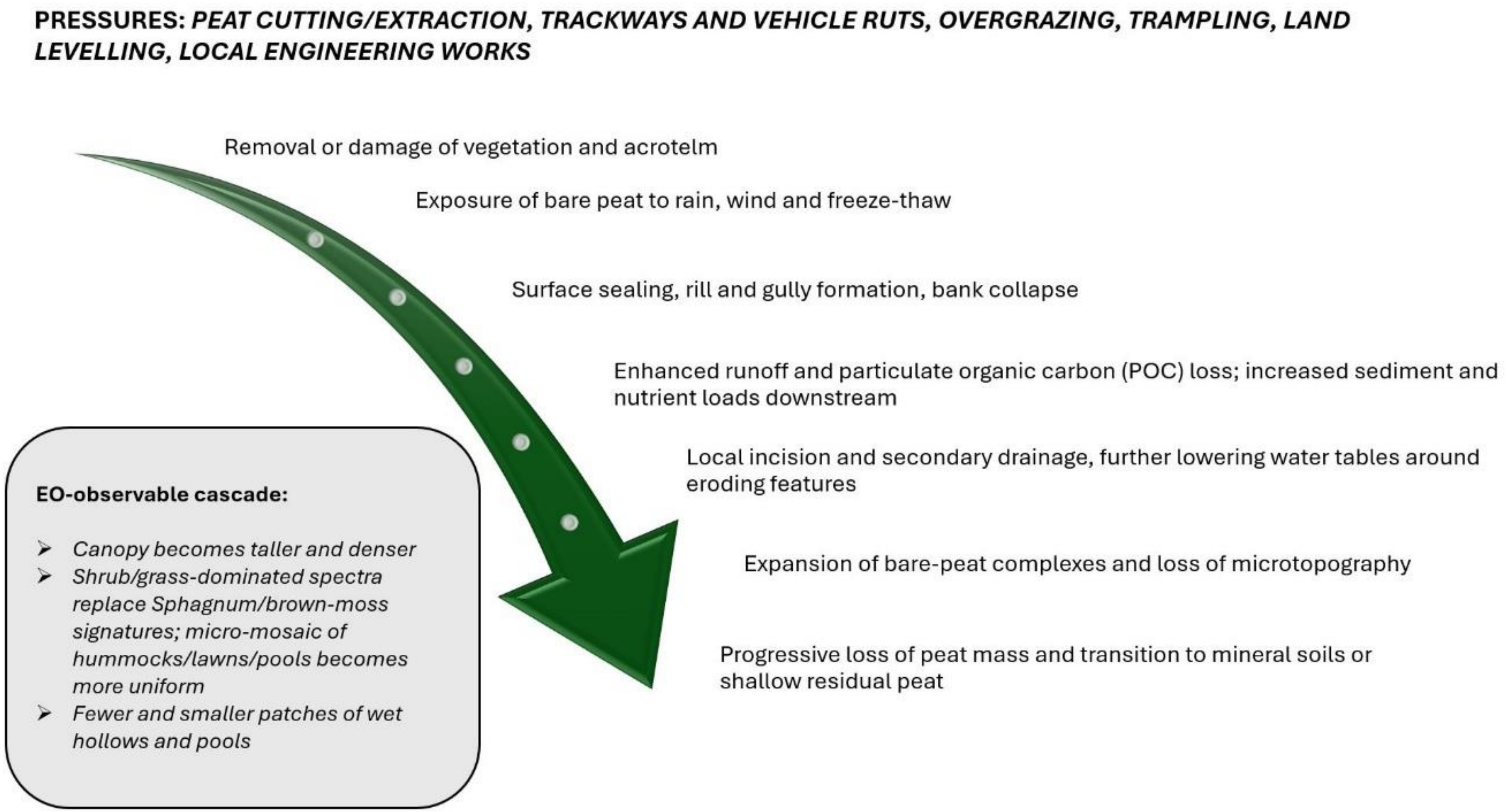
2.5.3 Degradation pathway: Physical disturbance → bare peat → erosion and secondary drainage

Local physical disturbance, e.g. from peat cutting and small-scale extraction, vehicle ruts and trackways, overgrazing and trampling, land-levelling or other engineering works, strips or damages the protective acrotelm layer and breaks the surface structure (Lindsay et al., 2014a; 2014b). Once vegetation and acrotelm are removed, dark, highly erodible peat is exposed to rain, sun, overland flow, wind, and freeze–thaw cycles.

This exposure triggers surface sealing and the rapid development of rills, gullies and bank collapses that concentrate runoff, export POC, sediment and associated nutrients, and incise into the peat mass. As gully networks deepen and connect to existing ditches and streams, they create secondary drainage pathways that lower local water tables around eroding features, expand bare-peat complexes and erase fine microtopography (Ward et al., 2007; Lindsay et al., 2014a; 2014b).

Over time, these erosion complexes can coalesce and, especially where drainage is also present, drive a progressive loss of peat volume and carbon, ultimately leaving shallow residual peat or exposing mineral soils, with strongly altered hydrology, reduced habitat quality and heightened vulnerability to fire (Ward et al., 2007; Maltby & Barker, 2009; Cummins et al., 2011; Meļņiks et al., 2023) (Fig. 12).

Figure 12. Physical disturbance, bare peat, erosion and secondary drainage cascade. Local pressures such as peat cutting, vehicle ruts, overgrazing, trampling and land levelling remove vegetation and the acrotelm, exposing bare peat to rain, wind and freeze–thaw. This promotes rill and gully formation, enhanced runoff and particulate carbon loss, and local incision that lowers water tables around eroding features. As bare-peat complexes expand and microtopography is lost, peat oxidation and erosion accelerate, driving a long-term transition towards shallow residual peat or mineral soils.



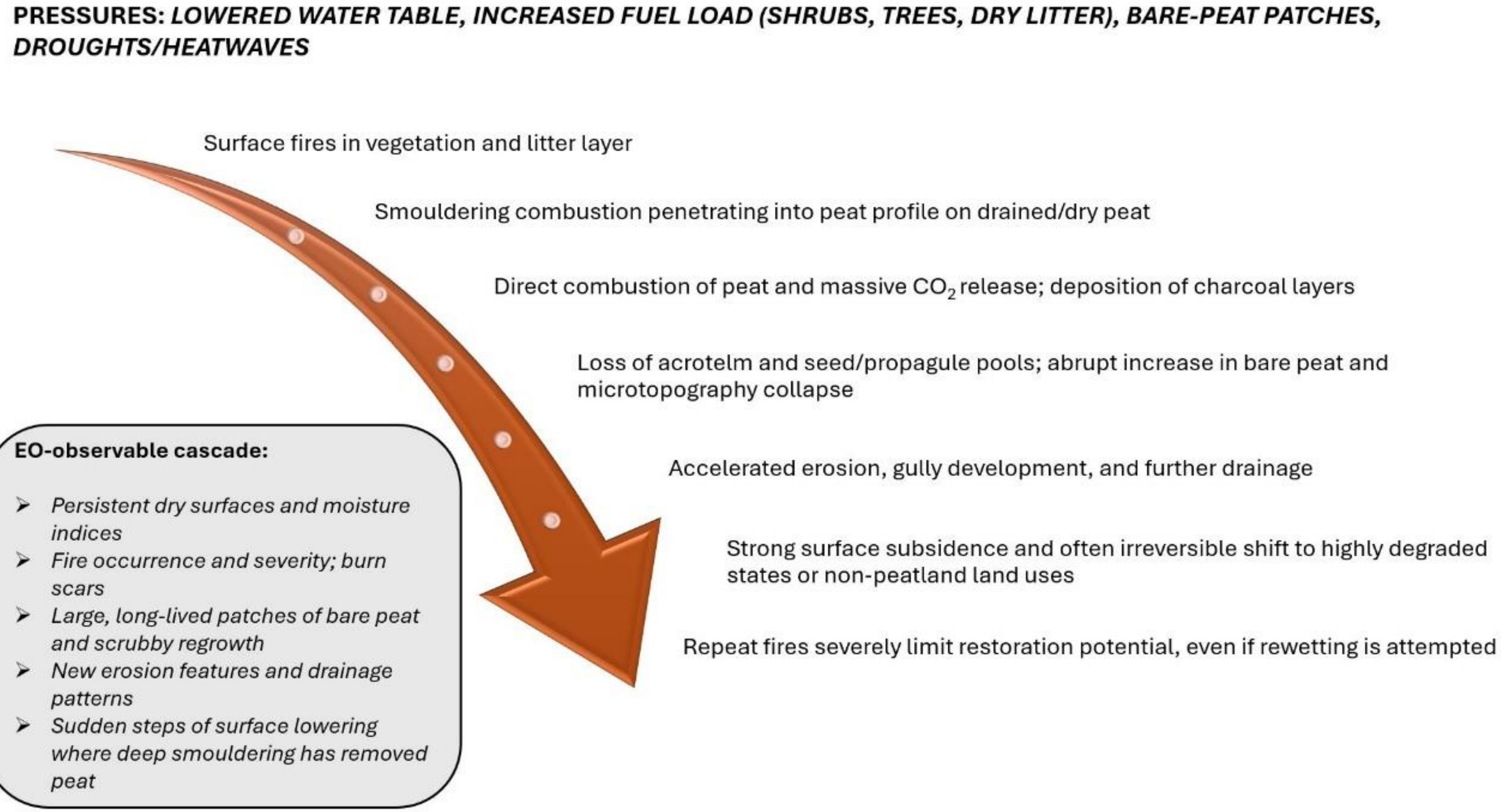
Source: Own elaboration.

2.5.4 Degradation pathway: Drying → fuel build-up → fire → rapid carbon loss

In many peatlands, fire acts as a secondary but powerful amplifier of existing hydrological and nutrient pressures (Ramchunder et al., 2009; Turetsky et al., 2014; Brown et al., 2015). Prolonged low water tables, expansion of shrubs and trees, accumulation of dry litter and bare-peat patches create a flammable surface layer. Under droughts, heatwaves or deliberate burning, surface fires ignite this fuel and, on drained peat, can transition to smouldering combustion that penetrates deeply into the peat profile (Davies et al., 2013). This leads to direct combustion of peat, large and abrupt CO_2 emissions, deposition of charcoal layers, loss of the acrotelm and propagule banks, and rapid expansion of bare peat with collapsed microtopography (Turetsky et al., 2014). The resulting surfaces are highly erosion-prone, promote new gullies and secondary drainage, and often subside markedly, locking the system into a severely degraded state where restoration is difficult, especially if fires recur (Fig. 13).

Fire interacts strongly with drought–rewetting biogeochemical cascades: severe drought can lower water tables, increase oxygen penetration and activate abiotic and biotic decomposition pathways; subsequent rewetting does not simply restore pre-drought conditions, but can maintain elevated CO_2 and DOC losses as nutrient-rich, labile carbon pools are mineralized. Fires superimposed on such dried, oxidized peat can therefore represent a non-linear step change in degradation, rapidly moving systems from moderate (Stage 2) to severe (Stage 3) states.

Figure 13. Fire-driven degradation cascade. Prolonged drying and fuel build-up allow surface and smoldering fires to consume peat, causing abrupt CO_2 release, bare-peat expansion, erosion, subsidence, and often an irreversible shift to highly degraded or non-peatland states.



Source: Own elaboration.

2.5.5 Interactions between degradation pathways

These cascades rarely occur in isolation. Instead, they reinforce one another over time:

- *Drainage × nutrient enrichment*: Lowered water tables and increased nutrient inputs favor tall, productive vegetation, which further increases evapotranspiration and internal drying, reinforcing the hydrological degradation pathway.
- *Drainage × physical disturbance*: Ditches, trackways, and bare-peat scars jointly promote rill and gully formation, deepening local water-table drawdown and propagating erosion through the peat body.
- *Fire on already degraded peat*: Fires usually occur in systems that have already dried and accumulated fuel (Stage 2 or Stage 3). When they occur, they greatly accelerate carbon loss, bare-peat expansion and subsidence, and can push peatlands beyond reversible thresholds even if rewetting is later attempted. Repeated fires can entrench alternative, low-carbon states (e.g. scrub or grassland on shallow peat).

Climate extremes as amplifiers: Droughts, heatwaves and intense storms superimpose on these pathways. Drought increases susceptibility to fire and oxidation; extreme rainfall events then mobilize large pulses of DOC, POC, acidity and metals from degraded peat to downstream waters.

Recognizing these linked pathways helps to identify (i) early warning signals (e.g. decreasing wetness indices, first signs of shrub encroachment, small bare-peat patches, loss of pools); (ii) indicators of negative changes and urgent need for action (e.g. extensive bare peat, dense drainage networks, strong subsidence trends, large burn scars, conversion to non-peatland land cover); and (iii) long-term cumulative change (e.g. long-term subsidence, stable non-peatland vegetation, chronic loss of wet microtopography).

2.6 Key variables and indicators for detecting degradation responses

Ecological indicators serve as quantitative representations of ecosystem states and responses, summarizing complex interacting mechanisms into interpretable evidence for management. However, indicators can be misleading if treated as universally transferable or if aggregated in ways that obscure causal pathways. This risk is particularly pronounced for peat-forming habitats, where fine-scale heterogeneity and often poor or inconsistent delineation can constrain the reliability of spatial indicators (UNEP, 2022). Consequently, expert evaluation cautions against the use of generic indicators not calibrated for peatlands (Nicholson Thomas et al., 2025). Instead, indicator selection should be guided by three principles: indicators must have mechanistic relevance to peatland functioning (primarily the hydrological regulation of peat formation), be sensitive to the main degradation pathways (drainage, land-use conversion, fire, and eutrophication), and remain operationally feasible, ideally combining EO for spatial tracking with field-based measurements where direct process verification is essential (FAO, 2020; UNEP, 2022).

Variables presented in this section can be classified into two tiers: (1) Essential variables – mandatory core indicators that must be measured in any assessment to characterize habitat condition (primarily water table depth, *Sphagnum* cover, peat subsidence, drainage density); and (2) Optional/Complementary variables – indicators providing additional diagnostic context, most valuable when essential variables indicate degradation or when resources permit expanded monitoring (e.g. water chemistry, microtopographic relief, bulk density, GHG fluxes). The symbols used to distinguish indicator priority, monitoring approach and data source in Tables 3, 4, and 8-11 are defined in Table 2. The weighting methodology for aggregating variables into composite condition scores requires type-specific calibration and should be adapted to the particular Group 71 or Group 72 habitat types being assessed, as described in section 2.6.2.

Table 2. Key of symbols used in the tables of variables and indicators for peatland (Tables 3, 4) and coastal lagoons (Tables 8-11) monitoring and diagnostics.

Symbol	Description
E	Essential (basic key indicator for ecological status diagnosis and risk assessment)
R	Relevant (can be necessary for accurate diagnostic or understanding processes)
O	Optional (additional information or important depending on the system and the studied problem)
	*in situ* (field sampling campaigns from boats, diving or from land, using instruments, gears or sampling tools)
	Requiring laboratory analyses of samples
	Cartographic tools
	EO-direct (data captured by satellites or remote sensors)
 P	EO-proxy (use of satellite data to infer a variable that is not directly measurable by the sensor)
	EO-direct (data captured in situ by buoys)
	Modelling or calculations based on measured data

Source: Own elaboration.

2.6.1 Ecological rationale behind selected indicator groups

Peatlands, being fundamentally water-controlled ecosystems, hydrology is treated as the primary axis for assessing condition and recovery. Water table depth is prioritized as the key variable for monitoring degradation risk and restoration effectiveness, as drainage remains the dominant driver of peatland carbon loss (FAO, 2020; UNEP, 2022). Importantly, water table dynamics are also modulated by climatic variables – particularly temperature (which governs evapotranspiration demand), precipitation, and insolation – so water table indicators must be interpreted in relation to regional and seasonal climate context rather than as absolute values. In regions experiencing increasing drought frequency, climate-driven water table decline can compound or mimic the effects of drainage, making it important to monitor both drivers simultaneously. Where direct measurement is limited, feasible proxies - such as canal depth or freeboard - are recommended. In parallel, remote sensing provides scalable indicator layers, including drainage patterns, vegetation inundation extent, and surface moisture content, which characterize hydrological conditions across large areas (UNEP, 2022; Ghezelayagh et al., 2025). Complementing hydrological data, indicators of physical peat condition are selected to capture the mechanisms of drainage-driven oxidation. Rather than generic soil properties, monitoring guidance explicitly prioritizes peat surface subsidence (and associated peat depth/extent changes). Peat surface subsidence refers to the progressive lowering of the peat surface relative to a fixed reference datum (e.g. a benchmark or adjacent mineral ground), caused by peat compression, shrinkage, and oxidative loss of peat mass following drainage. Subsidence serves as a direct proxy for sustained carbon loss and is frequently used to signal restoration need as well as to track success when rewetting reduces loss rates (FAO, 2020; Anshari et al., 2021; UNEP, 2022; Ghezelayagh et al., 2025).

Vegetation indicators integrate hydrological and nutrient conditions over time and often provide the most visible evidence of degradation and recovery. For peat-forming wetlands, selection focuses on metrics tightly linked to peat formation, especially *Sphagnum* cover, and compositional shifts, such as the ratio of wet- to dry-tolerant taxa - measured in terms of both percentage cover and species richness (van Breemen, 1995; Draga et al., 2025). In bog contexts, surface-wetness features associated with microtopography are also highlighted as relevant to functioning (Milton et al., 2005). Crucially, these biological metrics must be interpreted in a peatland-type-specific way (e.g. distinguishing bog versus fen baselines) rather than assuming universal thresholds (Maanavilja et al., 2023; Draga et al., 2025).

Finally, this report recognizes that no single indicator is sufficient to capture peatland condition or recovery on its own. A suite of complementary indicators is required to preserve interpretability across spatial heterogeneity and time lags. Therefore, condition indicators (hydrology, peat condition, vegetation) must be paired with information on key pressures, including drainage extent, land-use intensity, nutrient pollution, and fire risk (FAO, 2020; UNEP, 2022; Maanavilja et al., 2023; Nicholson- Thomas et al., 2025).

2.6.2 Use of indicators in the assessment workflow

The indicators listed in Table 3 are applied within a tiered assessment workflow designed to balance ecological relevance, operational feasibility, and EO scalability. Essential indicators constitute the minimum dataset required to classify peatland habitat condition, capturing core ecosystem state (hydrology, peat condition, vegetation structure) and dominant direct pressures (e.g. drainage, fire, extraction). These indicators are prioritized for routine monitoring and are evaluated using quantitative metrics derived from EO-direct or EO-proxy data, calibrated and validated where necessary using *in situ* measurements. Optional indicators provide complementary diagnostic and contextual information, supporting interpretation of degradation pathways, pressure attribution, and confidence in condition assessments, but are not required for initial status classification. For each site or spatial unit, indicator values are compared against reference conditions and indicative degradation thresholds (Table 3), interpreted in a peatland-type-specific context (e.g. bogs versus fens). Indicators are not aggregated into a single composite score; instead, they are evaluated as a structured indicator set to preserve mechanistic interpretability and to avoid masking causal relationships. This approach allows flexible application across data availability gradients, supports repeated assessments and trend analysis, and facilitates translation of ecological knowledge into EO-enabled monitoring systems while retaining expert oversight where uncertainty or context sensitivity is high.

Table 3. Indicators for peatland habitat condition assessment, including metric definitions and data observability. Indicators are grouped by ecological domain and classified as essential (E), relevant (R), or optional (O)* based on expert judgement. For each indicator, operational metric definitions and data sources are specified, distinguishing between EO-direct, EO-proxy, *in situ*, model/ancillary, and hybrid approaches to support repeatable, EO-enabled monitoring and assessment (see Table 2 for an explanation of the symbols).

Indicator and relevance to ecological status	Objective	Parameter description	Measurement
Hydrology			
Water table (WT) depth E	Maintain waterlogged conditions for peat formation and prevent oxidation	— Annual mean WT depth below ground surface (cm) over the hydrological year (preferably from continuous piezometer/logger data)	P
Water table duration near surface E	Avoid prolonged WT drawdown	— Number of days per hydrological year with WT within a near-surface band (e.g. 0–20 cm below surface) — Number of days with WT deeper than a stress threshold (e.g. >20 cm)	P
Water table fluctuation amplitude R	Limit large seasonal WT fluctuations	— Annual WT amplitude (max-min, cm)	P
Water table drawdown R	Limit very low summer water levels (i.e. prolonged dry phases)	— Summer minimum WT (cm); and % time WT is in a defined 'dry zone' (>20cm below surface)	P
Density and connectivity of drainage ditches R	Minimize artificial drainage and concentration of flows	— Total ditch/straightened-channel length per unit peatland/catchment area (e.g. m/ha or km/km^2) plus a connectivity descriptor (e.g. proportion connected to streams/outlet)	
Occasional surface flooding / flashy runoff R	Maintain high storage capacity and slow runoff response	— Frequency of rapid ponding/inundation events (events/yr) — Event extent (max inundated area; % of wetland) following rainfall — Optionally flood duration (days)	
Hydroperiod (duration of flooding / water saturation per hydrological year) E	Characterize how long a wetland is inundated or water-saturated each year	— For each pixel/wetland: number of days (or months) per hydrological year classified as inundated/saturated; summarize as mean/median and variability across the site	

Indicator and relevance to ecological status	Objective	Parameter description	Measurement
Surface soil moisture anomaly R	Detect unusually dry or wet conditions compared to a long-term baseline	— Surface soil moisture anomaly relative to a long-term reference (e.g. z-score, percentile or standardized anomaly), reported seasonally/annually — Optionally count of anomalously dry/wet periods	
Change in outflow water quality O	Detect deterioration of downstream water quality potentially linked to peat degradation/drainage and export or carbon/nutrients	— Outflow concentrations (and where possible loads) of DOC, TN/TP (or NO3/PO4), turbidity and/or colour (mg/L; Nephelometric Turbidity Units - NTU; Platinum-Cobalt Scale - Pt/Co), summarized as annual mean and event peaks	
Vegetation			
***Sphagnum*/peat-forming moss cover** E	Maintain dominance of peat-forming mosses and functioning acrotelm	— Percent cover (%) of *Sphagnum* and other peat-forming mosses in target open-bog/fen areas — Optionally patch area (m^2/ha) and temporal trend	 P
Presence of extremely specialized and poorly competitive peatland species (flora and fauna) R	Maintain rare, protected, and poorly competitive specialist species	— Occurrence/abundance of selected peatland specialist taxa (presence/absence, occupancy, counts/cover) — Richness of specialist species	
Vascular plant encroachment (shrubs, trees, reeds) E	Limit expansion of tall, dry-tolerant or eutrophic taxa	— Encroachment extent as % cover of shrubs/trees/reeds — Vegetation height (m) — Optionally woody canopy cover (%) and rate of expansion (ha/yr).	
Community-level vegetation composition E	Avoid shift towards drier, more productive vegetation types	— Relative area (%) of key vegetation classes along wet–dry gradient (e.g. wet lawns/pools vs dry heath/grassland), derived from classification — Optionally compositional indices	 P
Invasive alien species (IAS) and strongly eutrophic vegetation R	Prevent establishment and dominance of IAS	— Presence and % cover/area of IAS or strongly eutrophic stands (e.g. dense reedbeds), including trend in areal extent	 P

Indicator and relevance to ecological status	Objective	Parameter description	Measurement
Canopy height and roughness O	Distinguish low moss/graminoid surface from tall shrub/forest canopies	— Canopy height metrics (mean/median/95th percentile, m) and surface/canopy roughness indices (e.g. SD of height or SAR texture), referenced to ground/digital terrain model	
Morphology and soils			
Visible erosion and extensive bare peat E	Minimize bare peat, which signals erosion, oxidation, fire risk	— Percent of peatland surface classified as bare peat/erosion scars (%) — Optionally erosion-feature density (#/ha) and scar expansion rate (m^2/yr)	
Peat bulk density and degree of decomposition O	Detect peat compaction, oxidation, structural degradation	— Peat bulk density (g cm^{-3}) and humification/decomposition class (e.g. Von Post) from core samples, typically in the upper peat profile (e.g. 0–30 cm)	 
Gully formation on slopes; expansion of gully system R	Limit gully network expansion and associated peat loss	— Gully length and density (m/ha or km/km^2) — Gully area (m^2/ha) — Where repeated surveys exist: headcut retreat rate (m/yr)	
Moorsh soil development / strong peat degradation R	Identify transition from peat to degraded moorsh soils	— Presence/extent of moorsh horizons and oxidized/degraded peat (e.g. thickness of degraded layer, cm; bulk density increase; loss of peat structure/porosity)	
Soil subsidence rate E	Detect vertical peat surface lowering (likely due to drainage / oxidation)	— Vertical surface displacement rate (mm–cm/yr) relative to a stable reference surface — Optionally seasonal subsidence–uplift amplitude	
Disturbance and land use			
Fire frequency and burn scars on peatland surfaces E	Detect fire-affected peat and quantify recurrence and severity; avoid repeated fires that remove peat and propagules	— Annual burned area (% of peatland or ha/yr) — Burn-scar mapping per year — Fire return interval (years) — Optionally burn severity index	

Indicator and relevance to ecological status	Objective	Parameter description	Measurement
Agricultural use intensity (mowing, grazing, arable conversion) R	Limit agricultural pressure on peat soils	— Mowing frequency (cuts/yr) and timing — Grazing presence/intensity (binary or livestock units/ha/season where available) — % area converted to arable/managed grassland	 P
Fertilizer and seeding on peat soils (pressure indicator) O	Avoid nutrient enrichment of peatlands	— Fertilizer application rate and frequency (e.g. kg N/P/ha/yr; applications/yr) and reseeding/seed-mixture type — Optionally proxy via vegetation response + soil/water nutrients	 
Drainage system maintenance and regulation R	Identify actively managed drainage that perpetuates degradation	— Extent/frequency of ditch cleaning/deepening (e.g. length maintained per year) — Presence/condition of water-level control structures (weirs, dams) — Change in ditch cross-section where measurable	 P
Grazing, trampling and infrastructure (tracks, fences, vehicles) R	Minimize trampling-induced erosion and compaction	— Track/path density (m/ha) — Area of bare/trampled strips (%) — Mapped infrastructure features (count/length/area) — Optionally change over time	 P
Peat cutting and extraction E	Detect strong direct disturbance and loss of peat	— Area of cut-over/actively extracted peat (% or ha) — Rate of expansion (ha/yr) — Optionally proximity to extraction edges	
Dams and reservoirs affecting wetlands O	Capture hydrological alteration by upstream/downstream dams	— Number/density of dams and reservoirs in relevant basins (e.g. no./100 km^2) and their distance/position relative to wetlands — Optionally regulated storage/flow alteration index	
Landscape condition			
Habitat fragmentation and connectivity between subsites E	Capture how wetland patches are connected in space and via water flows; maintain	— Patch and connectivity metrics from habitat maps (e.g. number of patches, mean patch size (ha), edge density (m/ha), nearest-neighbour	

Indicator and relevance to ecological status	Objective	Parameter description	Measurement
	connectivity among peatland basins and associated habitats	distance (m), connectivity index/corridor presence)	
Transition / lagg zones extent and integrity R	Maintain transitional habitats between peat and mineral soils	— Width/area of lagg (wet perimeter strip) and other ecotones (m; ha) — Integrity metrics (e.g. % perimeter with intact wet buffer; vegetation/hydrology continuity class)	 P
Wetland abundance R	Quantify how 'wetland-rich' a landscape is	— Wetland area proportion of landscape (%) — Wetland density (no. per unit area), including temporal change in extent (ha/yr or %/decade)	
Carbon and greenhouse gases			
Net ecosystem CO_2 flux; overall GHG balance linked to drainage and management R	Maintain or restore peatland as a long-term net CO_2 sink, while acknowledging CH_4 emissions from natural mires	— Annual NEE of CO_2 (g C m^{-2} yr^{-1}) and CH_4/N_2O fluxes to estimate net GHG balance (e.g. g CO_2-eq m^{-2} yr^{-1})	 
All indicators (meta)			
Trend and anomaly of indicator value (Maes et al., 2020) R	Detect gradual and sudden changes in wetland condition or pressures	— For each indicator: trend as % change per decade relative to a baseline year; anomalies as standardized departures (e.g. z-score/percentile) vs a 20–30 yr reference period	

* Essential indicators (E) refer to a minimum set to classify habitat condition (core state and main direct pressures, as assessed by the Expert Panel). Relevant indicators (R) can be necessary for accurate diagnostic or understanding processes. Optional indicators (O) provide complementary diagnostic or contextual information and may become essential in specific peatland types or management contexts – e.g. agricultural use intensity (if most sites are managed peatlands), gully formation on slopes (if sloping/blanket bog settings are present, where gullying drives condition), dams and reservoirs affecting wetlands (if regulation is a dominant hydrological control).

Source: Own elaboration.

To facilitate translation of ecological indicators into operational monitoring systems (particularly by EO specialists), the following synthesis identifies which key variables for peat-forming habitats can be monitored via satellite or airborne remote sensing and which require *in situ* field measurement, along with the primary ecological rationale for each.

— **Primary indicators (EO-accessible)**: These are the most important drivers of peatland condition and are detectable at landscape-scale via satellite.

- Drainage density and ditch extent – mapped with fine-resolution optical imagery, aerial imagery, Light Detection and Ranging (LiDAR) Digital Terrain Model (DTM) products, and sometimes Specific Absorption Rate (SAR)-assisted approaches; fundamental because drainage is a major driver of peatland degradation
- Surface water extent and inundation frequency – detectable via SAR (e.g. Sentinel-1) and optical indices; partial proxy for near-surface hydrological conditions and flooding regime. Surface moisture / wetness index – detectable via SAR backscatter and microwave sensors; useful proxy for shallow water table dynamics or surface hydrological condition where direct measurements (i.e. field data) are absent
- Peat surface subsidence – detectable via interferometric SAR (InSAR) and high-resolution DTM change detection; strong EO indicator of ongoing degradation and carbon loss risk
- Vegetation cover type and composition (including *Sphagnum* extent, woody encroachment, bare peat) – detectable via multispectral and hyperspectral imagery; integrates hydrological and nutrient conditions and directly reflects peat-forming capacity
- Land use / land cover change (afforestation, drainage, burning scars) – detectable via change detection on optical imagery, and in some cases SAR; captures anthropogenic pressures

— **Optional/complementary indicators (*in situ* field measurement required):** These variables cannot be measured reliably by EO but are essential for process verification and threshold interpretation.

- Water table depth (direct dip-well measurement) – a key hydrological indicator, required to calibrate EO wetness and moisture proxies and validate drainage assessments
- Peat depth and bulk density – usually require direct coring; needed to quantify carbon stock and assess long-term peat loss
- Water chemistry (pH, electrical conductivity, DOC, N, P, nitrates, phosphates, sulfates) – requires water sampling; important for distinguishing fen vs. bog conditions, nutrient enrichment, and atmospheric deposition impacts
- *Sphagnum* cover and species composition – although broad-scale *Sphagnum* extent can be estimated from hyperspectral imagery, fine-scale species-level composition typically requires botanical survey
- Microtopographic relief (hummock-hollow ratio) – requires field survey or very high-resolution elevation data, typically from LiDAR/Unmanned Aerial Vehicle (UAV)-derived Digital Elevation Models (DEMs); reflects peatland surface structure and ecohydrological integrity

- An effective monitoring system for peat-forming habitats at European scale should prioritize the EO-accessible primary indicators for spatial coverage and trend detection, using *in situ* measurements at representative sites to calibrate EO products and validate ecological interpretations. Where resources are limited, initial monitoring effort should focus on drainage density and surface water extent (EO-based) paired with dip-well water table measurements (*in situ*), as these together capture the dominant degradation pathway.

2.6.3 Interpretation of indicator thresholds

The thresholds and reference values presented in Table 4 represent a synthesis of published empirical evidence and expert consensus. For variables with a robust evidence base, specific quantitative boundaries are drawn directly from literature; for instance, critical water table depths for degradation onset (typically > 20cm below surface) and thresholds for vegetation are supported by established studies (e.g. BfN, 2017; Wołejko et al., 2019; Albert-Saiz et al., 2025). Conversely, for indicators where standardized metrics are less established, thresholds are expert-derived and qualitative, relying on relative deviations from a baseline rather than absolute values.

Table 4. Reference ranges and degraded thresholds for peatland habitat condition indicators (where available), with supporting notes and literature sources. For each indicator listed in Table 3, good ecological status is characterised by a reference range reflecting near-natural hydrological, vegetation, morphological, and disturbance conditions, while poor status is defined by degraded thresholds beyond which significant functional or structural impairment is expected. Thresholds are indicative and should be interpreted in the context of peatland type, geographic region, and available baseline data. Indicators are grouped by ecological domain and classified as essential (E), relevant (R) or optional (O) following the scheme described in Table 3 (see Table 2 for symbol key).

Indicator and relevance to ecological status	**Reference range** (good status)	**Degraded threshold** (poor status)	**Notes / interpretation**	**Source**
Hydrology				
Water table (WT) depth E	<10-20 cm below surface	>20 cm below surface	Mean WT >20 cm for extended periods indicates degradation	Wołejko et al., 2019; Albert-Saiz et al., 2025
Water table duration near surface E	WT <20 cm below surface for most of the year	WT >20 cm below surface for most of the year		FAO, 2020
Water table fluctuation amplitude R	Stable WT (<10-20 cm depth) with low seasonal variation	Large seasonal fluctuations		FAO, 2020
Water table drawdown R	Shallow summer minima	Larger drops in water levels during dry periods compared to natural sites		

Indicator and relevance to ecological status	**Reference range** (good status)	**Degraded threshold** (poor status)	**Notes / interpretation**	**Source**
Density and connectivity of drainage ditches R	>30 m from nearest ditch; no dense ditch network visible	Large areas within 30 m of ditches; dense connected ditch network	Indicates artificial drainage and accelerated water export	NatureScot, 2016
Occasional surface flooding / flashy runoff R	Limited flooding confined to natural hollows, laggs, floodplains	Flashy behaviour with frequent surface flooding after moderate rainfall; ponding where water used to infiltrate	Reflects reduced storage and infiltration capacity due to compaction and drainage	
Hydroperiod (duration of flooding / water saturation per hydrological year) E	High saturation and/or wet pools present year-round	Longer dry periods; only ephemeral pools	Direction of change is wetland-type specific, as longer hydroperiod may be better or worse depending on ecosystem type (Maes et al., 2020)	BfN, 2017
Surface soil moisture (SSM) anomaly R	Conditions similar to 20–30 yr reference mean	Large positive or negative anomalies	Strong anomalies indicate abnormally wet/dry conditions	
Change in outflow water quality O	Outflow consistent with baseline conditions	Increased DOC, nutrients, turbidity or colour at outflow	Reflects drainage-driven export of C and nutrients	
Vegetation				
***Sphagnum*/ peat-forming moss cover** E	**Bogs:** >85% *Sphagnum* (other peatland bryophytes >9%); **Fens:** >50% moss cover (of which 70% brown mosses)	**Bogs:** <20% peat-formers; **Fens:** <20% moss cover, lack of brown mosses, dominance of *Sphagnum* in calcareous fens	Loss indicates functional breakdown of peat formation	González & Rochefort, 2019; Wołejko et al., 2019
Presence of extremely specialized and poorly competitive peatland species (flora and fauna) R	Specialists present and widespread in suitable microhabitats; ≥9 characteristic species present or total cover >50%	Specialists absent from most of peat body; confined to refugia or lost completely; fewer than 4 species of <20% cover		BfN, 2017; Wołejko et al., 2019

Indicator and relevance to ecological status	**Reference range** (good status)	**Degraded threshold** (poor status)	**Notes / interpretation**	**Source**
Vascular plant encroachment (shrubs, trees, reeds) E	Open peat-forming vegetation with only scattered shrubs or trees	Extensive (>15%) shrub thickets, tree cover or dense reedbeds replacing peat-forming vegetation	Indicates drying and nutrient enrichment	Wołejko et al., 2019
Community-level vegetation composition E	Hydrology-dependent peat-forming communities dominate; productive grasses and herbs limited to small areas	Predominance of drier, more productive vegetation (e.g. birch and pine, *Calluna vulgaris*, *Molinia careulea*) types over much of peat body	Reflects long-term hydrological change	Bonnett et al., 2009
Invasive alien species (IAS) and strongly eutrophic vegetation R	0% IAS present	>5% IAS cover	Often locally important; pressure-driven	Wołejko et al., 2019
Canopy height and roughness O	Low, rough moss/graminoid microtopography	Tall, homogeneous shrub/tree canopy	Structural shift from open peatland	Bonnett et al., 2009
Morphology and soils				
Visible erosion and extensive bare peat E	Minimal bare peat (outside natural erosion features)	Extensive bare-peat complexes; actively eroding faces	Strong indicator of oxidation and fire risk	Reynolds et al., 2025
Peat bulk density and degree of decomposition O	Low bulk density; and weakly decomposed acrotelm over fibrous catotelm	High bulk density; strongly decomposed or moorsh-like upper peat layers		
Gully formation on slopes; expansion of gully system R	Minimal active erosion	High gully density; rapidly expanding gully networks, and high peat loss (e.g. tens of mm yr^{-1} or more)	Indicates severe peat loss	

Indicator and relevance to ecological status	**Reference range** (good status)	**Degraded threshold** (poor status)	**Notes / interpretation**	**Source**
Moorsh soil development / strong peat degradation R	No or very limited moorsh horizons	Widespread moorsh soils; peat morphology largely lost		
Soil subsidence rate E	Very low subsidence	Sustained high subsidence (>1–3 cm yr^{-1}) over decades	Proxy for long-term carbon loss	FAO, 2020
Disturbance and land use				
Fire frequency and burn scars on peatland surfaces E	Fires very rare; nonexistent or isolated burn scars (low burned area, no reburn)	Recurrent fires within short intervals (years to decades); increasing reburn area and/or sustained burn-area fraction; >10% area burned		Silva-Sánchez et al., 2019
Agricultural use intensity (mowing, grazing, arable conversion) R	Extensive, low-intensity use only; no arable crops on peat	Intensive production, frequent mowing (1-2 times yr^{-1}), heavy grazing or arable conversion		
Fertilizer and seeding on peat soils (pressure indicator) O	No fertilisation or only very occasional, low-input management compatible with habitat objectives	Regular fertilisation on peat (typical of intensive grassland/cropland) and/or heavy nutrient loading from surrounding land	Best treated as pressure indicator	
Drainage system maintenance and regulation R	Drains grown over, no longer functional, or large-scale rewetting	Actively maintained drainage systems; deep ditches (2-3 m) kept open; intensive regulation/irrigation; fast water export; more seasonal flows downstream		BfN, 2017
Grazing, trampling and infrastructure (tracks, fences, vehicles) R	Low density of (localized) tracks and paths; limited local impacts	Extensive network of trampling tracks and compacted surfaces; often linked to erosion features		

Indicator and relevance to ecological status	**Reference range** (good status)	**Degraded threshold** (poor status)	**Notes / interpretation**	**Source**
Peat cutting and extraction E	Traces of historical extraction affecting up to 5% of the area, without current activity	Sporadic small-scale current extraction or historical larger-scale extraction; large-scale current extraction	Direct peat loss	Wołejko et al., 2019
Dams and reservoirs affecting wetlands O	River basins without, or with few, dams affecting natural flow to/from wetlands	High density or strategic placement of dams	Context-dependent pressure indicator (alteration of flow regiments, sediment transport, and nutrient distribution)	
Landscape condition				
Habitat fragmentation and connectivity between subsites E	Well-connected peatland complexes with functional corridors	Isolated habitat fragments; loss of connectivity between basins and associated habitats	*Landscape connectivity:* Patch metrics (size, shape, proximity, isolation); presence of corridor habitats (peat-filled valleys, transitional fens, wet forests). *Hydrological connectivity:* exchange of water, nutrients, sediments, and organisms between wetlands and surrounding hydrological networks	
Transition / lagg zones extent and integrity R	Continuous or widespread lagg/transition zones with characteristic zonation	Lagg zones absent or degraded	Extent and condition of lagg zones (very wet strip around the edge of a bog or peatland) and other ecotones	
Wetland abundance R	Near-natural distribution	Strong declining trend	Landscape-scale indicator	
Carbon and greenhouse gases				

Indicator and relevance to ecological status	**Reference range** (good status)	**Degraded threshold** (poor status)	**Notes / interpretation**	**Source**
Net ecosystem CO2 flux; overall GHG balance linked to drainage and management R	Slightly negative NEE (CO_2 sink) typical of intact/active mires	Persistent positive NEE (CO_2 source) under drained or intensively managed peatlands; strong N_2O emissions from fertilized peat; altered CH_4 dynamics		Nugent et al., 2018
All indicators (meta)				
Trend and anomaly of indicator value (Maes et al., 2020) R	Stable relative to baseline	≥5% negative change per decade		Maes et al., 2020

Source: Own elaboration.

It is important to recognize that these thresholds are inherently context-dependent. As a result, the generic thresholds provided here should be treated as indicative guiding ranges rather than rigid universal standards. Significant further research and site-/context-specific calibration are required to define precise quantitative thresholds for all proposed indicators, ensuring they account for local typological heterogeneity and climatic baselines. In particular, threshold ranges for water chemistry indicators (pH, electrical conductivity, DOC, N, P) must be differentiated by habitat type within Group 7 — both by the major division between Group 71 (acid bogs) and Group 72 (calcareous fens, petrifying springs), and within each group by the degree of peat formation. For example, acceptable pH ranges and nutrient concentrations in a '7210* Calcareous fen' differ substantially from those in a '7110* Active raised bog', and applying single universal thresholds would be ecologically inappropriate. Similarly, threshold ranges for water table depth should be differentiated between ombrotrophic bogs (where the water table relative to the peat surface is the critical parameter) and minerotrophic fens (where water table depth must be interpreted relative to local groundwater head and surface-water connectivity). Further calibration efforts are strongly encouraged to establish type-specific threshold values within the framework provided here.

3 Coastal lagoons

Key message

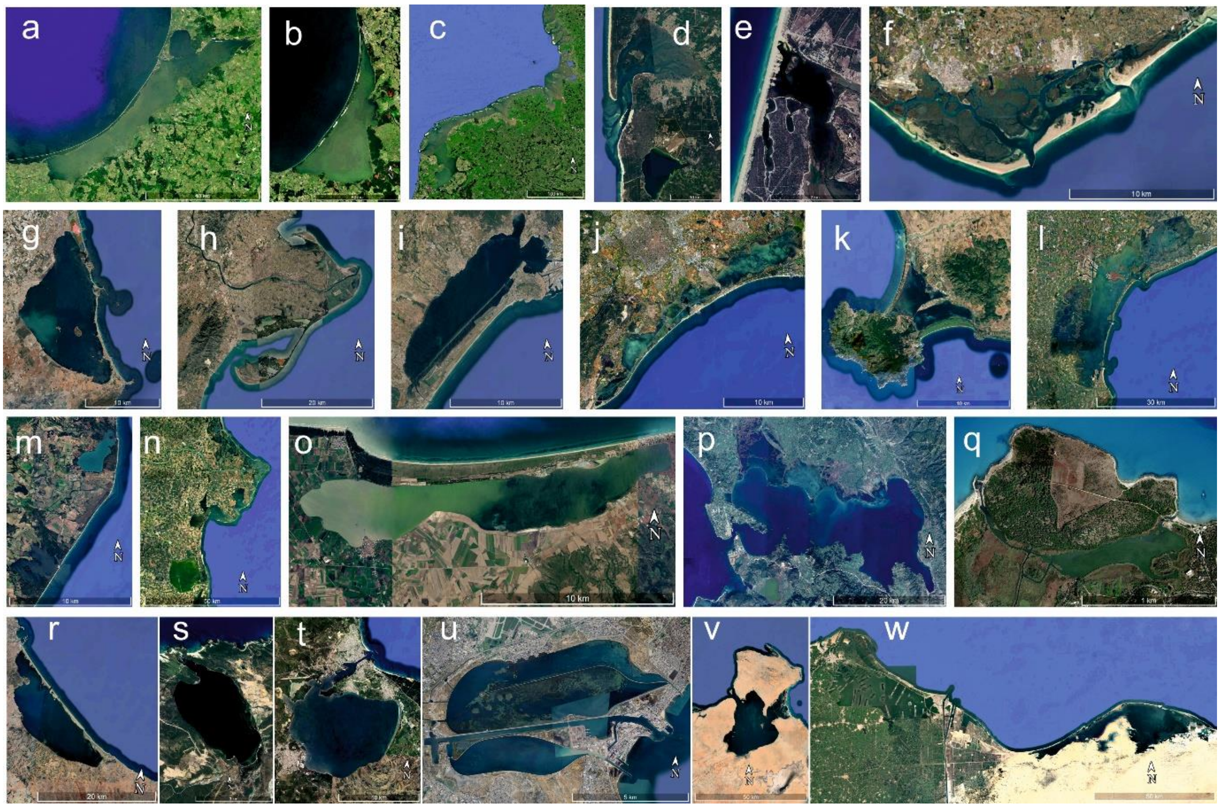
Coastal lagoons differ from estuaries and other coastal wetland habitats. No two coastal lagoons are alike, either in their hydrographic conditions or in the faunal composition of their communities. However, despite their geomorphological, hydrographic, and biological variability, there are key elements of their functioning that make it work in a similar way and generate the same goods and ecological services to human societies. Controlling the scales of variability at the local level (habitat types, vertical (depth) and horizontal (confinement) gradients, as well as temporal variability (seasonality, interannual, and ecological succession) is essential to differentiate anthropogenic impacts and climate change from the natural variability of the ecosystem. The identification of key variables and the interpretation of indicators must be placed in the context of this heterogeneity.

3.1 Coastal lagoons: heterogeneity and complexity of transitional ecosystems between terrestrial and marine domains

Coastal lagoons have been defined as an area of salt or brackish water separated from the adjacent sea by a low-lying sand or shingle barrier (Barnes, 1980; Annex 1). Indeed, they are transitional aquatic ecosystems, in the boundary between continental and marine domains, sharing functional traits with estuaries and other land–sea interface systems. They are characterized by marine influence combined with partial isolation from the open sea by barriers such as land spits or reefs, with water and organism exchange occurring through one or more restricted inlets. Lagoons are typically shallow, with mean depths rarely exceeding 2-6 m (Pérez-Ruzafa et al., 2011a), and exhibit pronounced physical–chemical gradients over short spatial scales (UNESCO, 1981; Tagliapietra et al., 2009; Pérez-Ruzafa et al., 2011b; 2024). Despite differences in origin and evolution, their biological functioning is regulated by common features, notably shallowness and restricted connectivity with the open sea (Pérez-Ruzafa et al., 2020a).

Coastal lagoons can be found in all European seas, from the Baltic to the North Sea and the Mediterranean. In the Mediterranean alone there are around 400 coastal lagoons, covering more than 6400 km (Cataudella et al., 2015) (Fig. 14). Globally, a wide range of terms is used to describe these environments, including coastal lagoons, coastal lakes, semi-enclosed bays, estuaries and multiple regional denominations like rías, lagunas, "small sea" (Mar Chica, Mar Menor, Mar Piccolo), lagoas, lagunes, étangs, ponds, albuferas, brednings, caletas, cienagas, marismas, esteros, marsh, marais, marios, stagni, sacca, limans, limnothalassas, zalews, semienclosed coastal systems (SECS), or the more recent terms intermittently closed and open lakes and lagoons (ICOLLs), or regions of restricted exchange (RRE) (Tett et al., 2003; Tagliapietra et al., 2009; Cataudella et al., 2015; Pérez-Ruzafa et al., 2019b).

Figure 14. Some representative coastal lagoons in European and Mediterranean seas. Baltic sea - a: Vistula lagoon (Poland-Russia); b: Curonian lagoon (Lithuania-Russia); North sea - c: Wadden sea (Netherlands); Northeast Atlantic - d: Arcachón (France); e: Sto. André (Portugal); f: Ria Formosa (Portugal); Mediterranean - g: Mar Menor (Spain); h: Delta del Ebro lagoons (Spain); i: Thau (France); j: Étangs palavasiens (France); k: Orbetello (Italy); l: Venice (Italy); m: Diana and Urbino (Corsica, France); n: Delta Po lagoons (Italy); o: Lesina (Italy); p: Ambrakikos system (Greece); q: Antinioni (Greece); r: Mar Chica (Morocco); s: Mellah (Algeria); t: Bizerte (Tunisia); u: Lac Tunise (Tunisia); v: Boughrara (Tunisia); w: Manzala - Bardawil (Egypt). Each lagoon is represented at a different scale.



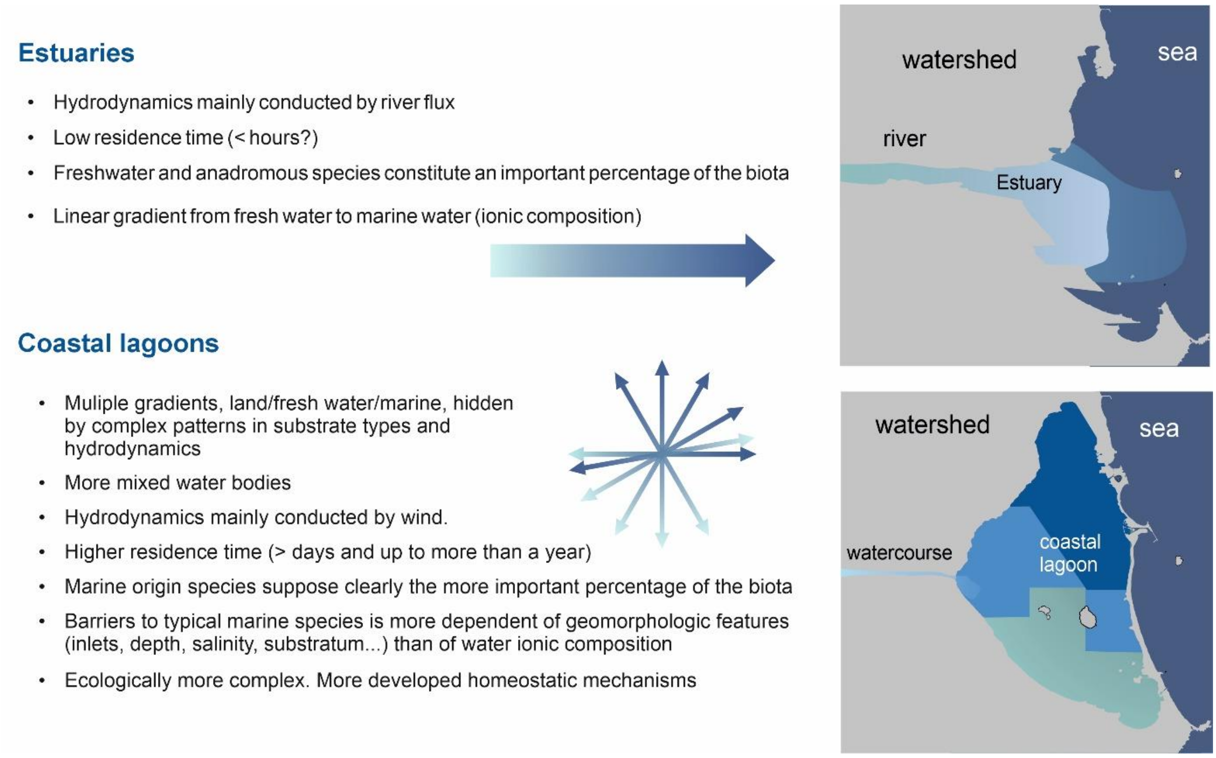
Source: *Google Earth*.

Under the European Union's Water Framework Directive (WFD; 2000/60/EC), among the numerous types of water bodies classified, a distinction is made between transitional waters (including mainly estuaries and water bodies on the coast with freshwater influence) and coastal waters. However, with the applied criteria, the classification of coastal lagoons remains ambiguous (Pérez-Ruzafa et al., 2011a).

The definitions of estuary and coastal lagoon partially overlap. Estuaries have been defined as "a semi-enclosed coastal body of water, which has a free connection with the open sea, and within which seawater is appreciably diluted with freshwater derived from land drainage" or as "an inlet of the sea that reaches the upper limit of the tide in a river valley" (McLusky, 1999). Thus, while estuaries are directly associated with river mouths and form part of fluvial systems, coastal lagoons may be connected to estuarine complexes but can also occur independently.

In this context, both transitional waters and coastal lagoons are land-sea transitional ecosystems characterized by strong physico-chemical gradients, high environmental variability, and assemblages dominated by estuarine and migrant marine species. However, they differ fundamentally in freshwater influence and spatial organisation (Fig. 15). Estuaries exhibit a clear longitudinal freshwater-marine gradient, with a one-dimensional structuring of salinity, ionic composition, and biological communities. In contrast, many coastal lagoons, particularly in the Mediterranean, receive little or no freshwater input and therefore do not strictly conform to the WFD definition of transitional waters (Pérez-Ruzafa et al., 2011a) and show more complex spatial patterns and gradients.

Figure 15. Main differences between estuaries and coastal lagoons ecosystems in structure and functioning.



Source: Modified from Pérez-Ruzafa et al., 2011a.

Coastal lagoons display highly variable and often extreme salinity conditions. Their ecological structure is governed by multidimensional interactions among salinity, confinement, hydrodynamics, depth, the type of substrate, and nutrient inputs. Confinement, expressed as restricted marine colonisation of larvae or individuals (Pérez-Ruzafa & Marcos, 1992; 1993), is a key driver of lagoon assemblages and generates horizontal zonation patterns, independent of salinity, like those described by Guelorget & Perthuisot (1983) and Guelorget et al. (1983).

In conclusion, although estuaries and coastal lagoons share high productivity and similar species migratory guilds, they differ mainly in the role of freshwater input and in the dimensional complexity of environmental gradients. These differences support the recognition of coastal lagoons as a distinct functional category rather than as simple transitional waters under current regulatory frameworks.

3.1.1 Geomorphological and hydrographic features characterizing coastal lagoons

3.1.1.1 Size and morphology

Lagoons occupy the transition between land and sea and are found in diverse environmental settings. Coastal lagoons are here considered as “inland water bodies separated from the ocean by a barrier, connected to the ocean by one or more restricted inlets which remain open at least intermittently, and have water depths which seldom exceed a few meters" (Adlam, 2014). About 32,000 lagoons are reported along 13% of the world's coastline (Carter & Woodroffe, 1994), with a coastline contribution estimated at 17.6% for North America, 12.2% for South America, 5.3% for Europe, 17.9% for Africa, 13.8% for Asia, and 11.4% for Australia (Barnes, 1980; Table 5). Coastal lagoons are areas of salt or brackish water separated from the adjacent sea by a low-lying sand or shingle barrier (see examples in Fig. 16). These dynamic habitats can develop into other coastal environments, such as semi-enclosed marine bays, freshwater lakes, or estuaries.

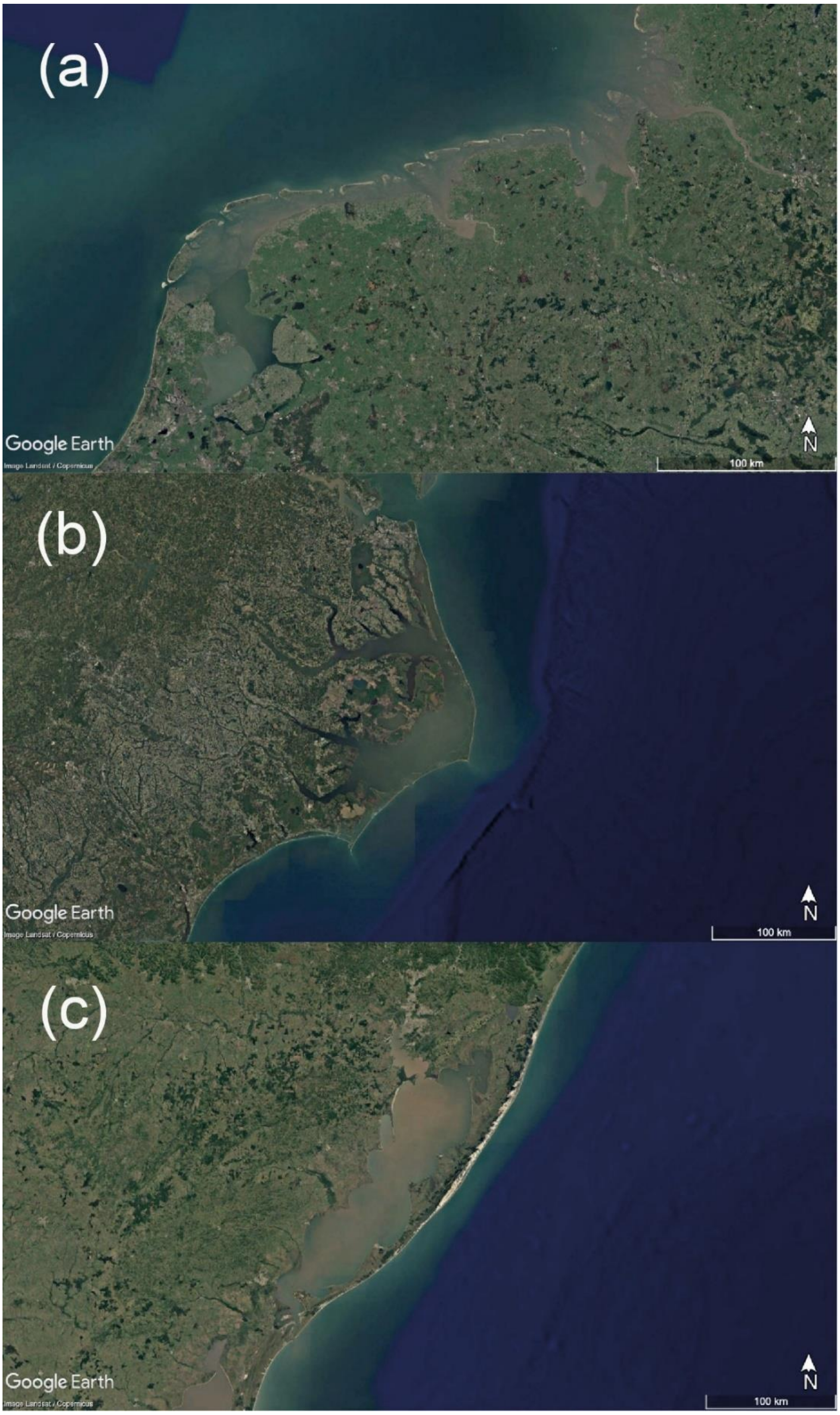
Table 5. Distribution of barrier/lagoon coasts by continent.

Continent	Barrier/lagoon coast (km)	% of continent's coastline	% of world's lagoonal coast
N. America	10765	17.6	33.6
Asia	7126	13.8	22.2
Africa	5984	17.9	18.7
S. America	3302	12.2	10.3
Europe	2693	5.3	8.4
Australia	2168	11.4	6.8
Total	32038		

Source: Barnes (1980).

Lagoons are most common in microtidal areas, though they can also be found in mesotidal and occasionally in macrotidal environments (Kjerfve, 1994). As a result, they display a wide range of physical and chemical characteristics (Carter & Woodroffe, 1994). Within this framework of bay or lake-like systems, semi-isolated lagoons bounded by sand or shingle barriers vary greatly in size, from small pools just tens of meters in length, to large inland seas such as Lagoa dos Patos in southern Brazil. Their size and abundance, along with their significance for fisheries and other human interests, make lagoons an important and valuable habitat for people (Barnes, 1980). Evolutionary processes of lagoons vary according to their environmental settings. Their morphodynamic evolution accordingly adapts to the specific conditions of their environment (Carter & Woodroffe, 1994).

Figure 16. Lagoon systems enclosed by offshore barrier-island chains. (a) German East Frisian Island – Wadden seas (Germany); (b) Pamlico Sound (USA); (c) and a lagoon enclosed within the land - Lagoa dos Patos (Rio Grande, Brazil).



Source: *Google Earth.*

The entrance to a coastal lagoon may be bordered by a spit or paired spits, as seen in the estuarine lagoons at Poole, Christchurch, and Pagham on the South Coast of England. Alternatively, there may be several entrances separating barrier islands, such as in the Dutch, German, and Danish Wadden Sea. Some lagoon entrances are residual gaps that have persisted between barrier islands or spits, where the lagoon was never fully separated from the sea. Others result from breaching caused by storm waves or floodwaters spilling out of the lagoon. Many lagoon entrances are artificially created, having been excavated and stabilized, usually with breakwaters, to facilitate navigation, regulate fisheries, or to accelerate the drainage of floodwaters to the sea. In Italy, for example, the three entrances of the Venice Lagoon have been artificially stabilized with breakwaters up to two kilometers long, with intervening barrier islands emerging as large seawalls (Bird, 2008; Fig. 17). In Portugal some entrances of the Ria Formosa Lagoon have being kept natural and they do migrate according to the natural longshore drift (Vila-Concejo et al., 2004; Fig. 17).

Figure 17. View of entrances: upper panel, the lagoon of Venice, showing one of the stabilized inlets; lower panel the Ria Formosa, displaying a natural, non-stabilized inlet (Ancão Inlet).

Source: Google Earth.

The length of barriers depends on shoreline and subtidal topography, but very long barriers can form, as is the case of Padre Island, which encloses the Laguna Madre in Texas and stretches approximately 200 km. Sand is the most common material for these barriers, although shingle can also be present; shingle, being denser and heavier, is less easily moved and often contributes to offshore barriers formed by storm waves (Barnes, 1980).

Lagoons often function as restricted estuaries, featuring tidal oscillations and the mixing of saltwater from the sea with freshwater from inflowing rivers (Bird, 2008). Some lagoons require some degree of marine or at least brackish water, some others are completely isolated from the sea (rarely), while others can be hypersaline and freshwater influence can be a factor of stress (Pérez-Ruzafa et al., 2011a). Although their entrances are often narrow compared to the total coastwise extent, barriers or natural landforms typically close off the lagoon, maintaining a limited connection with the open ocean (Bird, 2008). Typically, they have one or more channels that facilitate water exchange with the larger adjacent waterbody. In some small lagoons, water exchange may occur primarily through percolation across the confining barrier. If a lagoon becomes completely isolated, its water usually becomes fresh, leading the lagoon to evolve into a freshwater coastal lake or pond. Consequently, lagoons are considered semi-isolated systems, though opinions may differ on the exact level of separation needed to classify a body of water as a lagoon. Generally, a typical lagoon features a small entrance channel relative to the overall size of the lagoon and the length of the barriers - sometimes to the point where no discrete channel exists at all (Barnes, 1980). In practice, the region that is closer to the entrance channel, and therefore dominated by water from outside the barrier, is often marine. However, this is not always the case when there are rivers that are flowing directly into the lagoon.

The varying influence of freshwater inflows compared to marine influences, and the balance between rainfall and evaporation, determine a high degree of salinity variability among coastal lagoons. This is why salinity has been used since 1958 to classify the lagoons and other brackish or estuarine waters according to the Venice lagoon system. Accordingly, a lagoon with salinities below 5 ppt is oligohaline; mesohaline waters have salinities between 5 and 18 ppt, and polyhaline waters between 18 and 30 ppt. Lagoons with salinities above 30 ppt but below that of seawater are termed mixoeuhaline (de Wit, 2011), while lagoons with salinities higher than the adjacent sea are considered hyperhaline.

It is important to note that salinity within individual lagoons can also exhibit extreme variability (Pérez-Ruzafa et al., 2019b). For example, values in Anatoliki Klisova Lagoon (Greece) range from 0.5 to 42 (Katselis et al., 2003), while Mundel Lagoon (Sri Lanka) shows a broader range from 9 to 109 (Silva et al., 2013). Numerous studies document transitions of lagoons between hypersaline, hyposaline, and freshwater states over time (Havinga, 1959; Pérez-Ruzafa et al., 2005a; Ramos Miranda et al., 2005; Dailidienė & Davulienė, 2008; García-Seoane et al., 2016). Because salinity strongly regulates species colonisation, establishment, and community structure, it constitutes a key parameter in lagoon monitoring designs (Pérez-Ruzafa et al., 2019b) and sustainable management plans. Climate change projections further emphasize its importance, as increased precipitation variability and enhanced connectivity with the open sea due to sea-level rise are expected to intensify spatial and temporal salinity fluctuations in coastal lagoons (Anthony et al., 2009; Angus, 2017).

For its part, hydrodynamics are conditioned by the winds (limited by the fetch, since they are semi-closed systems) and by the currents through the communication channels with the sea. The current velocity in a lagoon depends on the wind, mainly in non-tidal areas, and the tidal prism, i.e. the volume of water that enters and leaves the lagoon as the tide rises and falls. As the tidal volume increases, the mean cross-sectional area of the lagoon's entrance also increases (Pethick, 1992). Currents are generated through the entrances in several ways: one is by outflow from rivers, especially after heavy rains and floods, which raise the lagoon's water level and cause water to flow out through the entrance. Another source of currents is wind action, with onshore winds driving water out to sea. Strong currents tend to maintain or shape the dimensions of a lagoon's entrance, with the cross-sectional area varying in relation to the volume of water passing through, being wider or deeper during episodes of floodwater discharge. The deposition and dimensions of lagoon entrances often change frequently in response to these processes, with migrating entrances punctuating the outer barrier. On the Atlantic coast of the United States, for example, the Outer Banks of North Carolina have experienced a long and complex history of breaching, enlargement, reduction, migration, and closing of tidal inlets. The entrance to Pamlico Sound, in particular, has moved back and forth along the coast over time (Bird, 2008). Cyclic migration of lagoon entrances can also occur on many barriers. This phenomenon has been observed in the Ria Formosa barrier system (Vila-Concejo et al., 2004).

All this high variability in the stages of development, geomorphological differences, the marine influence versus that of continental waters, or the climatic conditions that affect the hydrology and hydrodynamics of water masses have made it difficult to have a unified system for classifying coastal lagoons. While lagoons can be classified based on factors like salinity, substratum type, or mode of formation, the most practical distinction usually depends on their degree of isolation from the sea, the diversity of sub-habitats they encompass, and their size (Kjerfve, 1994; Pérez-Ruzafa et al., 2019b). These classifications distinguish lagoons bounded by offshore barrier islands from those enclosed by alongshore bars and spits. In nature, the former are closer to enclosed bays, and the difference between lagoons such as the Wadden Sea lagoons in the North Sea and those in the Gulf of Mexico is mainly a matter of degree. Conversely, more inland systems, isolated by longshore barriers, tend to resemble lakes and ponds into which they may ultimately evolve (Barnes, 1980). There are many smaller lagoons along river mouths, some of which are of significant interest despite their small size. An example of such a lagoon is Oyster Pond on Cape Cod in the United States (Bird, 2008). The lagoons, their barrier islands, coastal spits, and peripheral wetlands create ecotones that generate highly diverse habitats, supporting a high level of biodiversity (de Wit, 2011).

Coastal lagoons enclose morphologies presenting different spatial scales, ranging from bed forms (~ cm to m) such as ripples and dunes, to channel–shoal patterns, wetlands, and full basins (~ 1–100 km; Carter & Woodroffe, 1994, Hibma et al., 2004), which have complex evolutions that do not follow a specific formula or respond linearly to forcing factors (Ranasinghe et al., 2012, Carrasco et al., 2016). Wetland landscapes are characterised by distinct ecogeomorphic units, such as tidal flats, and low- to high-marshes (Zhou et al., 2022), which are commonly referenced in relation to mean sea-level, mean high tide, or highest high tide (e.g. Bertness et al., 2002; Lima et al., 2025). Alterations in topographic features and/or tidal exchange have long-term consequences for plant growth, and the volumes and pathways of sediment transport across the wetland platform, determining its ultimate shape and slope (Fitzgerald & Hughes, 2019).

Some European lagoons have no significant freshwater inputs, and are thus considered to be sheltered, coastal waters in the context of the WFD, whereas they are considered to be coastal lagoons and therefore transitional in character by the scientific community. This is one of the complications for the implementation of the WFD that arises from the characterisation of coastal lagoon systems (Pérez-Ruzafa & Marcos, 2008). In the Atlantic, significant tidal exchanges often dominate the hydrography, with freshwater inputs playing a relatively important role. In contrast, the Mediterranean experiences large meteorological variability in precipitation, leading to fluctuating freshwater runoff from drought to flood conditions. This creates highly dynamic conditions, making it particularly challenging to characterize transitional waters in this region (Newton et al., 2014).

3.1.1.2 Formation, development and fate

The lagoons existing today are relatively young ecosystems, most of which were formed during the Holocene transgression, particularly between 5,000 and 1,000 years BP. Barnes (1980) states that a typical lifespan for a coastal lagoon is around 1,000 years, but many lagoons have longer durations. For example, the Curonian Lagoon on the Baltic Sea, the largest coastal lagoon in Europe at 1,540 km^2, has existed for about 5,000 years. Some Mediterranean lagoons date back to Roman times, while others were formed during the medieval period (de Wit, 2011).

The formation of a coastal lagoon is initially shaped by the inlet or opening and the surrounding barriers. Its evolution is influenced by geological history, sea-level changes, and land submergence, which determine inlet positions and lagoon morphology (Pérez-Ruzafa et al., 2019b). Once formed, they are continually modified by erosion and other natural processes (Bird, 2008). Coastal lagoons, especially the smaller ones, are rapidly changing, highly dynamic systems. Their biology can only be understood within the context of their formation, evolution, and subsequent decline.

Where spits or offshore barriers develop more or less parallel to the original shoreline, elongate lagoons will be formed with their long axes parallel to the coastline. Offshore bars and barriers are produced by wave action on shallow, gently shelving sand coasts. Wave action may be of two types: constructive and destructive. When constructive waves enter shallow water, they begin to break, and the shoreward motion and swash move particles of sediment towards the coast, and when waves begin to break some distance offshore a bar is formed there by the deposited sediment. Sediment is then accumulated in offshore deposits in lines parallel to the coast which accumulate and merge in the breaking zone (Barnes, 1980).

During the last 15,000 years, the sea level rose by around 100 meters. This has had the effect not only of drowning low-lying areas but also of permitting the sweeping action of constructive waves to gradually roll large quantities of material up the gentle slopes so that the barriers contain more material than they would have done had sea-level been constant during this period (Barnes, 1980). Once enclosed, the movement of sediment may continue to dominate the history of the lagoon and determine its precise shape. In elongated lagoons, for example, the largest fetch will be along the lagoon's long axis and end wind-induced waves and currents will oscillate along its length (Barnes, 1980).

The bed of a lagoon is usually composed of soft muds, primarily due to the deposition of fine sediments in the relatively sheltered conditions. These sediments are mainly supplied by inflowing rivers. As a result, particle sizes tend to be particularly small in the freshwater and brackish zones and gradually increase toward the entrance, where more vigorous water movement keeps silts in suspension. Due to their shallowness, wind action can reach the lagoon bed, and fine sediments are easily resuspended, often causing the water to become turbid during periods of high wind. The rate of sedimentation, which is enhanced by submerged and emergent vegetation, results from a balance between the input of materials from the river system and the processes of resuspension and outflow. Usually, there is a net accumulation of sediment. Storms can also deposit fans of sediment eroded from the barriers into the main body of the lagoon (Barnes, 1980).

Fluvial sediment supply varies worldwide but is highest in regions where chemical weathering occurs alongside sufficient rainfall to transport eroded debris. In some cases, this process can be hindered by dense vegetation, which reduces erosion and sediment transport (Carter & Woodroffe, 1994). Where sediment supply is high, there is an increased likelihood of lagoon infilling, resulting in tidal prism reduction, progradation, and eventual outlet closure. Sites with greater tidal ranges tend to have stronger currents, which promote the development of well-defined tidal channels, whereas lagoons with smaller tidal ranges are more influenced by waves and tend to have less pronounced tidal channels (Carter & Woodroffe, 1994). Tidal range also impacts the volume of flood-tidal deposition by affecting tidal space and the length of barrier islands (Carter & Woodroffe, 1994).

Lagoon orientation affects wave fetch and sediment redistribution along the shoreline. Lagoons in narrow valleys perpendicular to dominant winds experience less wave-induced sediment movement, while those elongated parallel to the winds are more affected. Their orientation is mainly determined by local geology, often occupying joints or depressions. Lagoons aligned perpendicular to the coast have stronger fluvial influence, especially when confined between steep margins. Sediment burial reduces volume less in typical coastal lagoons than in those oriented parallel to the coast (Carter & Woodroffe, 1994).

Once formed, lagoons do not generally persist for long periods in geological terms; most last for considerably less than 1000 years, with lifespan tending to increase with size. Several processes can lead to the destruction of lagoons. Many, possibly even the majority formed since the postglacial rise in sea level, have now become freshwater lakes, as barriers eventually prevent seawater entry except during storm surges, hurricanes, or other events causing exceptionally high water levels. Most lagoons are extremely shallow and are relatively easily transformed into swamps or marshes, ultimately leading to plant colonisation and encroachment. This process is less significant in highly saline lagoons (Barnes, 1980). One of the most important environmental features of lagoons, which in turn influences many other aspects, is their shallow depth. Due to their mode of formation, maximum depths exceeding 10 meters are rarely found; many lagoons are much shallower. Similar to estuaries, lagoons are typically floored by soft sediments, but unlike estuaries, large areas of intertidal flats generally do not occur because of the stability of the water level. However, lagoons that shrink in size during a dry season may expose seasonally some stretches of sand and mud that are somewhat equivalent to intertidal flats (Barnes, 1980).

3.1.2 Hydrographic and ecological processes that determine the functioning and high biological productivity of coastal lagoons

The functioning of coastal lagoons is fundamentally determined by geomorphology, basin topography, substrate composition, and hydrodynamic regime (Duck & Figueiredo da Silva, 2012). The presence of intense environmental gradients with the watershed and the sea favors the flow of energy, materials, and organisms. At the same time, their shallow depth favors both the arrival of light and the effect of currents and waves on the bottom, intensifying the gradients. All of this allows for high biological productivity, both benthic and planktonic, which places coastal lagoons among the most productive marine ecosystems with the highest fishing yields (Kennish, 2016; Pérez-Ruzafa et al., 2024), with maximum values approaching or even exceeding those of the main upwelling systems.

Although salinity has traditionally been viewed as the principal driver of lagoon biodiversity gradients and characterizing lagoon assemblages ; McLusky, 1999), it represents only one among several interacting factors. Differences from open-sea salinity, lagoon size, seawater exchange, wave exposure, habitat distribution, and trophic status together provide a more accurate explanation of community patterns (Franco et al., 2006a, b; Pérez-Ruzafa et al., 2007a). Importantly, salinity and nutrient regimes are especially sensitive to anthropogenic modifications.

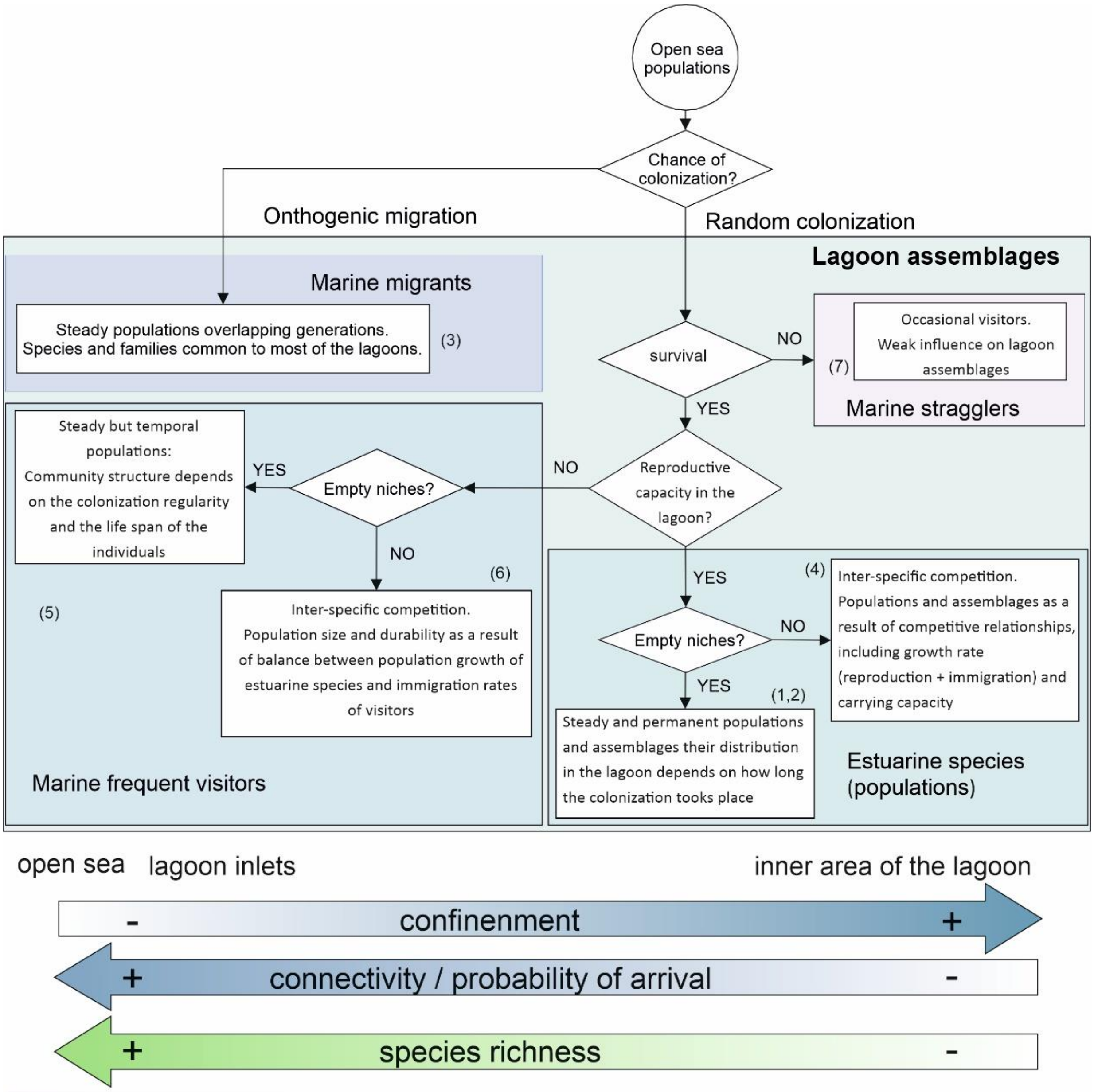
Frequent environmental variability combined with elevated productivity typifies early ecosystem successional stages (Odum, 1969). Under the fluctuating environmental conditions that characterize coastal lagoons, benthic communities of many coastal lagoons tend to display low diversity, dominance by a few opportunistic *r*-strategists, and bottom-up regulatory dynamics (Margalef, 1969; Kjerfve, 1994). Resultant community states exhibit low biomass-to-abundance ratios, high production-to-biomass (P/B) ratios, and trophic networks dominated by detritivores and nutrient-limited algal producers (Wilkinson et al., 1995). Although coastal lagoons are often conceptualized as uniform euryhaline or highly tolerant biocoenoses (due to the wide variability of environmental conditions such as salinity or trophic state) (Pérès & Picard, 1964; Gascón et al., 2008), lagoon benthos are, in practice, far more heterogeneous than expected. Vertical zonation, substrate diversity, and confinement gradients generate distinct assemblages from lagoon inlets to internal basins (Guelorget & Perthuisot, 1983; Pérez-Ruzafa et al., 2008, 2011b). Examples from the Mar Menor show that macrophyte diversity is strongly structured by irradiance, hydrodynamics, substrate, and the magnitude of environmental fluctuations (García-Sánchez et al., 2012; Pérez-Ruzafa & Marcos, 2015).

High primary productivity, together with significant organic and nutrient inputs from adjacent wetlands, sustains rich resident and migratory faunal communities (Kennish, 2016). While some stressed lagoons may show phytoplankton-dominated, bottom-up control similar to anthropogenically eutrophicated systems, pristine lagoons can be strongly top-down regulated, with benthic micro- and macro-phytobenthos as key primary producers.

Numerous marine species that reproduce in the open sea migrate to coastal lagoons as juveniles or larvae and take advantage of the lagoon biological productivity to develop the part of their biological cycle that requires the greatest energy input for growth and sexual maturation. They then return to the sea as adults to reproduce. These species are typical in most coastal lagoons and form stable communities together with the species native to each coastal lagoon, which reproduce there, maintaining stable populations. As a result, lagoon communities are spatially and temporally heterogeneous.

In this context, among structural and hydrographic factors that determine lagoon assemblages' composition and structure, the degree of isolation and confinement (Guelorget & Perthuisot, 1983; Kjerfve, 1994) strongly influences colonisation dynamics and thus community assemblages (Pérez-Ruzafa & Marcos, 1993). A revised confinement model conceptualises lagoon assemblages as outcomes of colonisation probability modulated by species-specific establishment capacity and competitive interactions, shaped by local hydrodynamics, substrate, organic content, salinity, and environmental predictability (Pérez-Ruzafa & Marcos, 1993) (Fig. 18). Restricted connectivity and stochastic colonisation promote the persistence of rare taxa and prevent community homogenisation, thereby enhancing structural complexity and producing highly variable spatial and temporal assemblage patterns (Pérez-Ruzafa, 2015). Observed annual turnover may exceed 40% in ichthyoplankton (Quispe, 2014) and benthic macrofauna (Sala-Mirete et al., 2025).

Figure 18. Conceptual model of the faunal components of coastal lagoon biological communities based on confinement gradients and species colonisation probabilities according to their ecological strategies and interactions.



Marine stragglers	(7)
Frequent visitors whose presence and distribution depend on colonization rates, competition and life span	(6)
Frequent visitors whose presence and distribution depend on colonization rates and life span	(5)
Fluctuating lagoon populations. Horizontal gradient strongly depends on competition and colonization rates	(4)
Stable lagoon populations (recent colonisers. Horizontal gradient is temporary and disappear after some years	(2)
Marine migrants	(3)
Stable lagoon populations (old colonisers)	(1)

Source: Modified from Pérez-Ruzafa & Marcos (1992) and Pérez-Ruzafa et al. (2019a).

Although there are no taxa strictly exclusive to lagoons, many marine species that inhabit coastal lagoons maintain populations that are specially adapted to their respective environmental conditions (Pérez-Ruzafa et al., 2008). The effect of environmental conditions in each lagoon leads to differences in the optimal ranges of salinity or temperature between lagoon populations and those in the open sea. The restricted colonisation imposed by inlets often produces unique haplotypes or rare allelic compositions due to selection and bottleneck amplification (Vergara-Chen et al., 2010; Milana et al., 2012). Since lagoon environments are more extreme than adjacent marine systems, they can function as reservoirs of genotypes preadapted to future climate stressors (Pérez-Ruzafa et al., 2019b). The combined effects of high productivity, environmental heterogeneity, habitat diversity, and episodic recruitment of migratory fauna yield high overall biodiversity (European Environment Agency, 2010; De Wit, 2011).

Inter-lagoon comparisons reveal equally pronounced variability. Morphometric and hydrological attributes – average depth, degree of opening, coastal complexity, inlet geometry, shallow-area extent – as well as salinity and chlorophyll-*a* concentrations, explain differences in floral and faunal composition, species richness, and fishery yields (Pérez-Ruzafa et al., 2007a). Species co-occurrence in different lagoons is typically low: among 179 fish species surveyed across 40 lagoons, only 98 occurred in more than two (Pérez-Ruzafa et al., 2007a), and of 621 macrophyte species recorded in 73 lagoons across the Atlanto-Mediterranean region, only 7.3% occurred in >10 lagoons (Pérez-Ruzafa et al., 2011a). Only a few macrophytes (e.g. *Chaetomorpha linum*, *Ulva intestinalis*, *Zostera noltei*, *Ruppia* spp.) and fish species (e.g. *Dicentrarchus labrax*, *Anguilla anguilla*, Mugilidae, *Sparus aurata*, *Solea solea*, *Atherina boyeri*) appear in >50% of systems (Pérez-Ruzafa et al., 2011a).

3.1.3 Spatial and temporal scales of variability in coastal lagoons

The environmental and geomorphological heterogeneity of coastal lagoons, as well as the randomness of species colonisation processes, result in great biological variability – genetic, specific, physiological, and community – both spatially and temporally, within the same lagoon and between different lagoons. These conditions require the search for impact indicators based on species composition and dominance. However, although part of this variability is due to stochastic processes, the dynamics of the system remain limited by environmental attractors that restrict viable conditions. Populations adjust their demographic and reproductive strategies to daily, seasonal, and interannual fluctuations in factors such as nutrients, temperature, light, and food availability. This allows probabilistic predictions of system states based on trophic strategies or structural indices.

At the community and ecosystem levels, these adaptive responses generate complex networks of interactions that vary continuously in space and time. Spatial variability responds to environmental heterogeneity, while temporal changes define system dynamics and are a consequence of seasonality, climate change, evolving human pressures, and stochastic colonisation processes. Both spatial heterogeneity and temporal dynamics are expressed across multiple hierarchical spatiotemporal scales, from millimeters to biogeographic regions and from seconds to geological time (Table 6, Fig. 19).

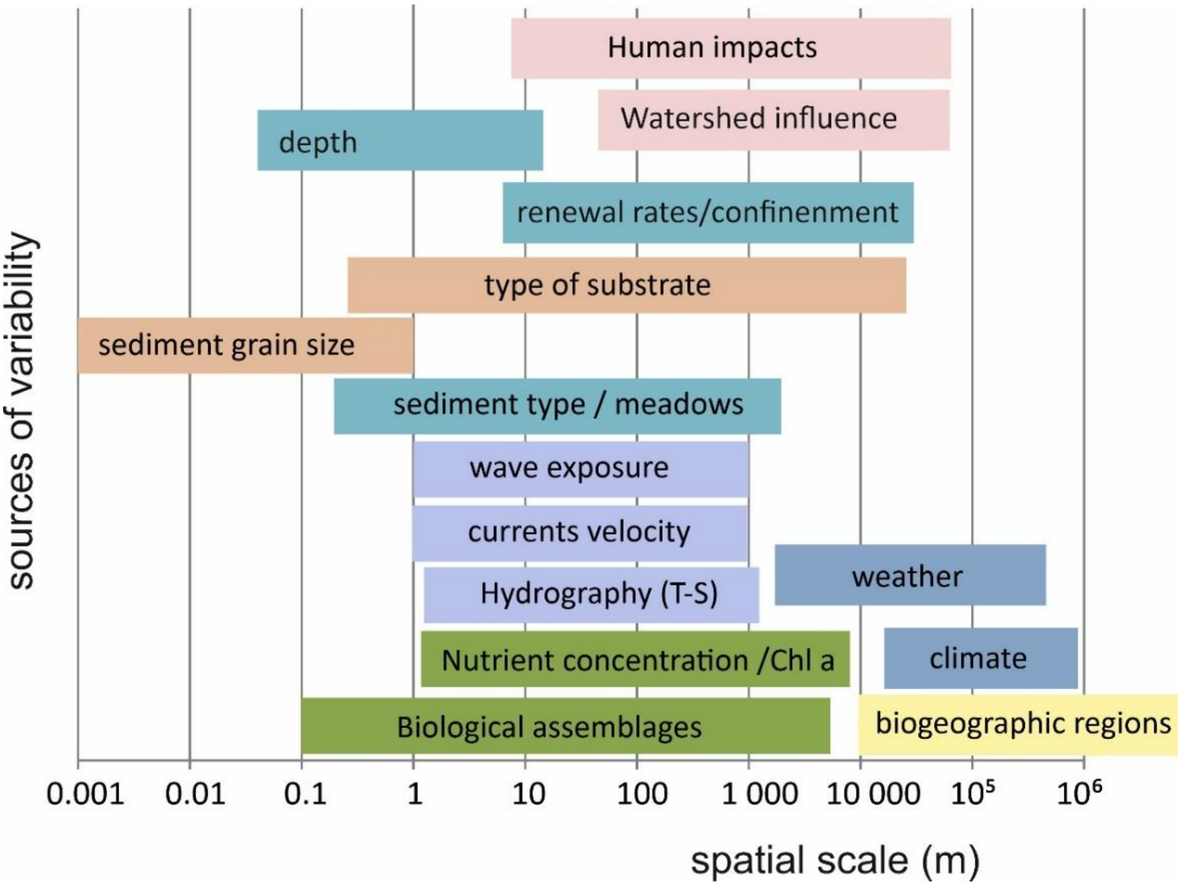
Table 6. Sources and spatio-temporal scales of variability of physico-chemical and biological parameters in coastal lagoons and main driving forces operating at each scale.

Spatio-temporal scales	Main driving forces	Parameter
Biogeographic regions (10^2-10^3 km)	Climate	Temperature Solar radiation
Lagoons (10^1-10^2 km)	Geomorphologic features (size, coastal development, openness, sea/freshwater influence), water balance, tidal range. Watershed extension and characteristics. Human activities.	Salinity Nutrients Chlorophyll Species composition
Basins (10^1 km)	Confinement: colonisation rates (faunal assemblages) / Environmental stability (macrophytes) Human activities in the basin Inlets distribution Watercourses and rivers mouth distribution	Hydrographic characteristics Trophic status Benthic assemblages
Zones (10^0-10^1 km)	Sea/fresh water influence; Seasonal weather Currents intensity	Temperature Salinity Chlorophyll Ichthyoplankton Fish diversity Fishing yields
Sectors (10^{-1}-10^0 km)	Sea/fresh water influence; Seasonal weather; Wind and circulatory patterns	Salinity Nutrients Chlorophyll Ichthyoplankton
Locality (<10^2 m)	Substrate nature and habitat complexity Wave exposure	Hydrodynamic Benthic assemblages
Vertical zonation (<10^0 m)	Vertical gradients in light and hydrodynamic	Benthic assemblages
Decades (10^1-10^2)	Ecological succession; Climate change; Human uses: Coastal works; inlets modifications, land claiming, river divergence / Changes in agricultural practices in the watershed, urban and industrial development, waste inputs	Temperature Salinity Renewal rates Nutrients Chlorophyll Ichthyoplankton Benthic assemblages
Years (10^0)	Interannual variation in weather and water balance Random connectivity	All parameters

Spatio-temporal scales	Main driving forces	Parameter
Seasons (10^{-1} -10^{0})	Seasonal weather/biological cycles and life stories	All parameters
Months (10-2 - 10-1)	Seasonal weather/biological cycles and life stories	All parameters
Forthnight (<10^{-2})	Wind and circulatory patterns	All water column parameters

Source: Modified from Pérez-Ruzafa et al. (2007b).

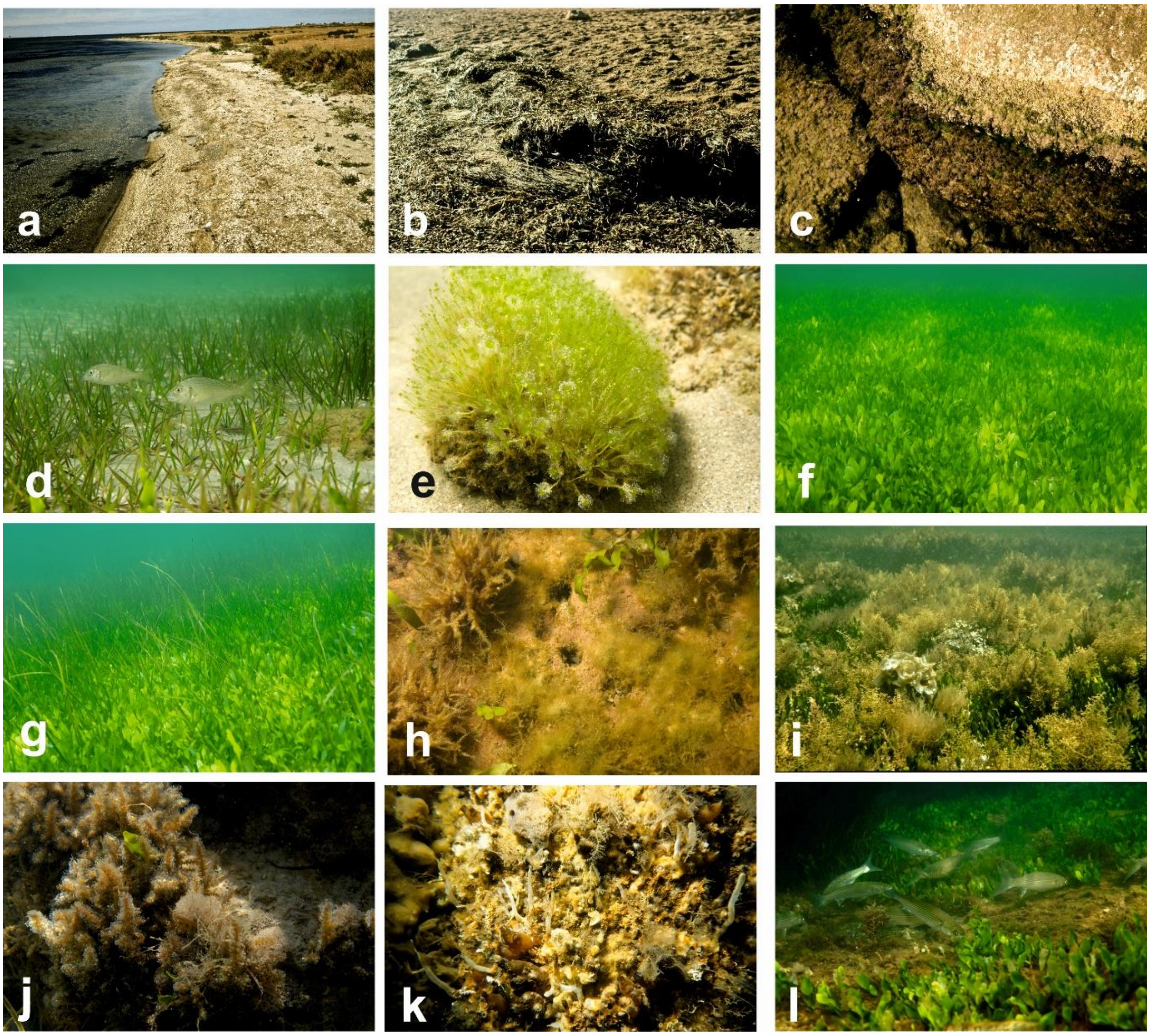
Figure 19. Spatial scales of the main sources of variability in coastal lagoons' climatic, geomorphological, hydrographic and biological characteristics. These sources of variability must be controlled in sampling designs to be able to differentiate natural changes from human or climate change impacts.



Source: Own elaboration.

In coastal lagoons, where terrestrial-lagoon-marine and bathymetric gradients intersect, environmental variability and community renewal are particularly pronounced. Gradients in humidity, salinity, temperature, oxygen, irradiation, and hydrodynamics drive canonical patterns of vertical zoning from the supralittoral to the midlittoral and infralittoral zones, as described in classical biogeographic frameworks (Peres & Picard, 1964; Peres, 1982) (Fig. 20). Horizontal and coastal perimeter variability is further conditioned by the geological and geomorphological context, human activities, and coastal occupation, resulting in a heterogeneous mosaic of habitats differentiated by sediment and substrate type, wave exposure, hydrodynamics, terrestrial inputs, water quality, and nutrient and contaminant regimes. The resulting seascape is therefore a spatially and temporally complex set of habitats with varying degrees of structural and functional complexity, whose natural variability must be distinguished from anthropogenic impacts when conducting environmental diagnoses and impact assessments (Pérez-Ruzafa et al., 2007b; 2011c).

Figure 20. Some of the benthic communities that can be found in coastal lagoons depend on depth gradients and substrate types. The figure uses, as an example, those found in the Mar Menor coastal lagoon, where the heterogeneity of habitats and the transparency of its waters allow for particularly high diversity of communities. Supra- and mid-littoral: (a) accumulations of *Cerastoderma* shells and other; (b) *Cymodocea nodosa* and algae washed ashore; (c) midlittoral communities on rock; Infralittoral: (d) seagrass meadows (*C. nodosa*) on sand; (e) facies of *Acetabularia calyculus* in fine calibrated infralittoral sand; (f) macroalgal beds (*Caulerpa prolifera* meadow) on mud; (g) mixed *C. prolifera* - *C. nodosa* meadow on mud; (h) infralittoral compacted terrigenous red clay; (i) shallow infralittoral rock in a well-illuminated exposed environment, with *Cystoseira* spp.; (j) shallow well-illuminated infralittoral rock without *Cystoseira* spp. and with Rhodomelaceae; (k) sheltered and shaded shallow infralittoral rock; (l) group of gray mullets feeding on an infralittoral rock community colonized by *C. prolifera*.



Source: © Angel Pérez-Ruzafa.

3.1.4 Seasonal cycles, interannual variability, ecological succession

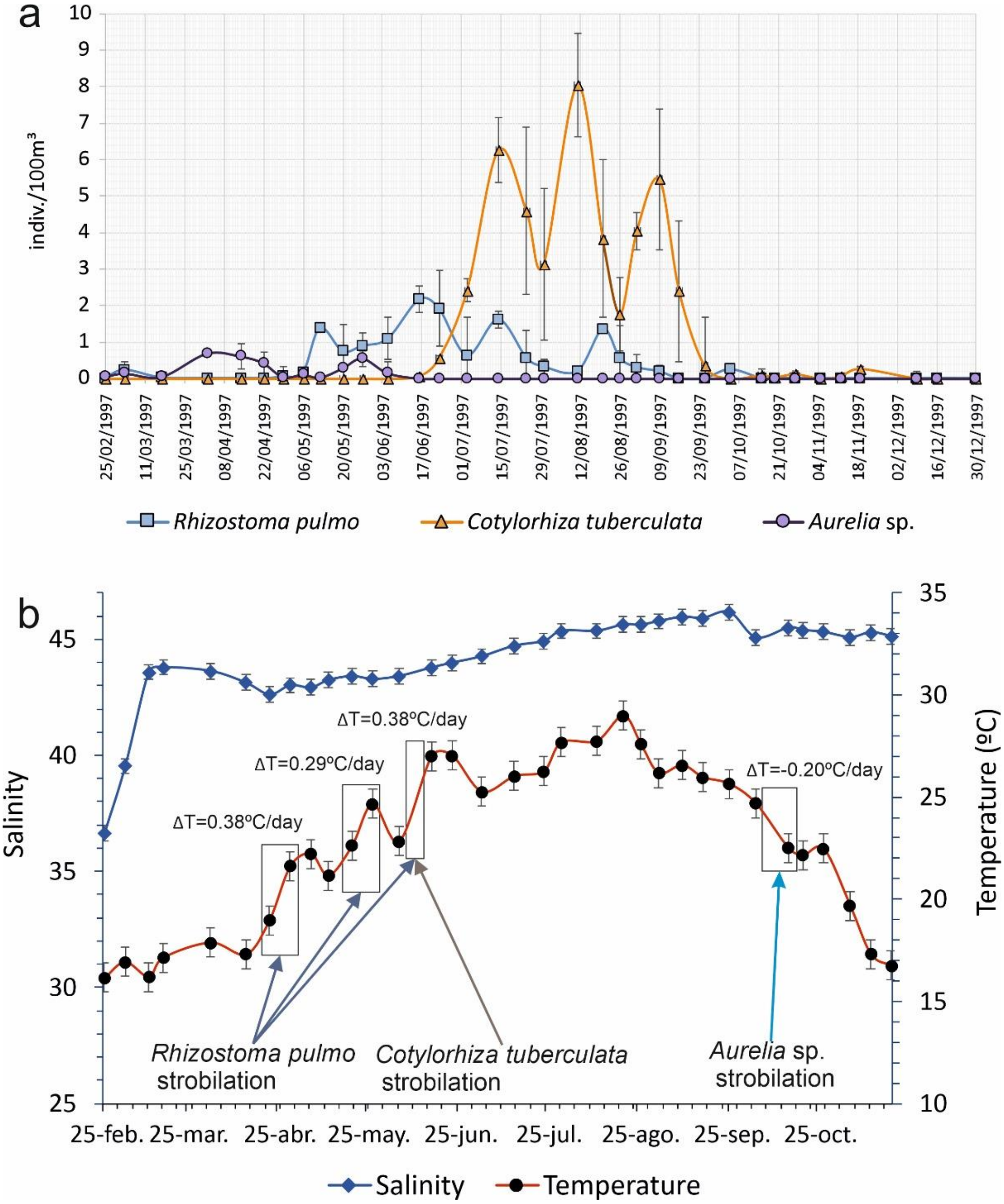
One of the reasons why coastal lagoons and wetlands are considered naturally stressed ecosystems is that several physical and chemical water characteristics are under a wide range of change at all temporal scales (Pérez-Ruzafa et al., 2018a). This high variability of environmental conditions is amplified by the small volume of the water body. This variability, combined with the stochasticity of the colonisation processes that contribute to shaping lagoon populations, makes it difficult to trace a definite temporal direction in ecological succession and even in inter-annual changes, which remains a task pending further research in these ecosystems. In the same way, temporal patterns on the coastal wetland communities maintaining permanent or semi-permanent water layers are also particularly complex to generalise, since different wetland types are under different main abiotic drivers. Hydrology is a main factor, but, as it has been stated in the previous section, a high number of other factors are co-working, such as salinity and nutrients, among others (Quintana et al., 1998a; Pérez-Ruzafa & Marcos, 2015).

In the case of small Mediterranean coastal lagoons and wetlands without permanent sea connection, communities are strongly determined by hydrological connectivity, which drives community assembly structure and composition because of the alternation of flooding-confinement periods (Guélorget & Perthuisot, 1983; Beklioglu et al., 2007). Flooding periods are driven by rainfall, sea storms, and groundwater inputs, whereas confinement periods can result in a high decrease of the water level and even desiccation in temporary ponds (Quintana et al., 1998a; Menció et al., 2017). These fluctuations influence both abiotic conditions and hydrological connectivity among ponds (Comín, 1982; Equisuany et al., 2026). Moreover, these alternations between flooding and confinement determine community composition, especially in phyto- and zooplankton (Comín, 1982; Quintana et al., 1998b). Thus, patterns of seasonal changes in coastal wetland zooplankton have been described, but thye are less related to seasonality than to episodes of flooding and subsequent confinement. It has been observed that the composition of the community changes, and six phases have been described (from flooding to confinement), according to six different species compositions (Quintana et al., 1998b; Brucet et al., 2005; Badosa et al., 2006). However, differences in this pattern are observed between temporary and permanent coastal ponds, due to the effect of biotic interactions such as fish predation (Brucet et al., 2003; Badosa et al., 2007). Sudden community composition changes between phases (from copepod dominance to rotifer dominance) have been reported, with rapid changes in oxygen and temperature being the main drivers (Quintana et al., 2021). Although benthos composition also changes according to flooding-confinement, the pattern is quite different, since when flooding occurs community composition becomes similar among all coastal ponds, whereas in confinement situations, temporary and permanent waterbodies host different communities (Gascón et al., 2007).

In larger coastal lagoons temporal rhythms may be marked by seasonal climatic and hydrological cycles that determine rainfall patterns, evapotranspiration, light radiation, and temperature. These will affect salinity, the inflow of nutrients and materials from runoff, photosynthetic capacity, and the reproductive cycles of species. In addition, they also indirectly determine cycles in human activities, such as farming and tourism, which also tend to have seasonal fluctuations, but these are much more variable depending on the geographical area, the nature of the activity, and the socio-economic context.

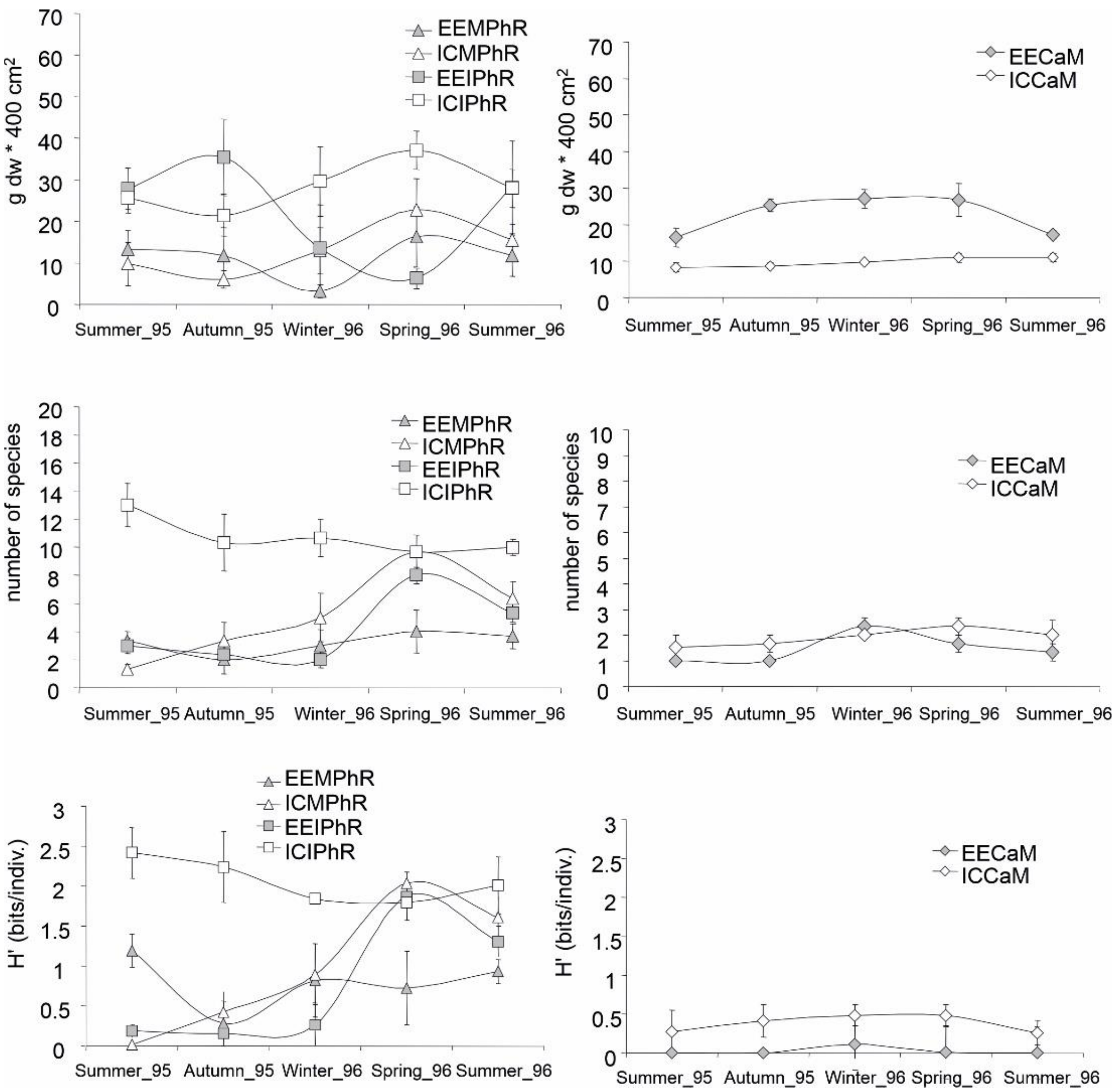
However, even though there are more or less defined seasonal patterns, these may vary depending on the taxonomic groups, lagoon communities, and assemblage descriptors (abundance, biomass, species richness, etc.). Figures 21-23 show some of these patterns in the Mar Menor case study.

Figure 21. a: Seasonal cycle of adults of the three main species of jellyfish in the Mar Menor. b: Seasonality of the strobilation periods of the three species of jellyfish marked by periods of sudden increases or decreases in temperature, depending on the species.



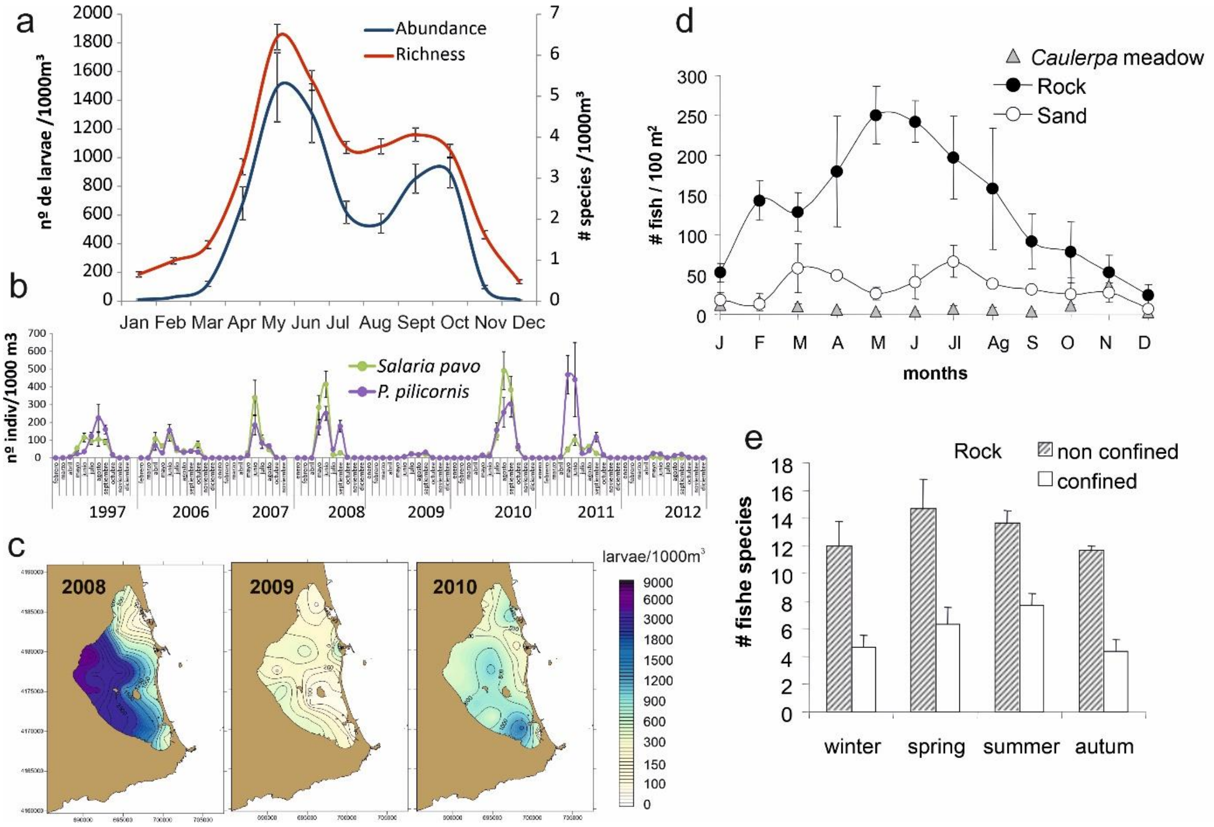
Source: Fernández-Alías et al. (2020).

Figure 22. Seasonal dynamics of different macrophyte assemblages descriptors - total biomass (up), species richness (middle), and Shannon H' diversity (down) - in different communities in two localities (EE: El Estacio; IC: El Ciervo Island) in the Mar Menor lagoon. MPhR: midlittoral photophilous assemblage on rock; IPhR: infralittoral photophilous assemblage on rock; CaM: *Caulerpa prolifera* meadow.



Source: Pérez-Ruzafa et al. (2007b).

Figure 23. Fish assemblages' spatial and temporal variability in the Mar Menor lagoon. a: Ichthyoplankton seasonal dynamics (abundance and species richness); b: Seasonal dynamics and interannual variability of the larvae of two species of blennies (*Salaria pavo* and *Parablennius pilicornis*); c: spatial and interannual variability in ichthyoplankton abundance; d: Seasonal dynamics of benthic fish assemblages' abundance in three different habitats; and e: benthic fish species richness in two basin areas with different confinement.



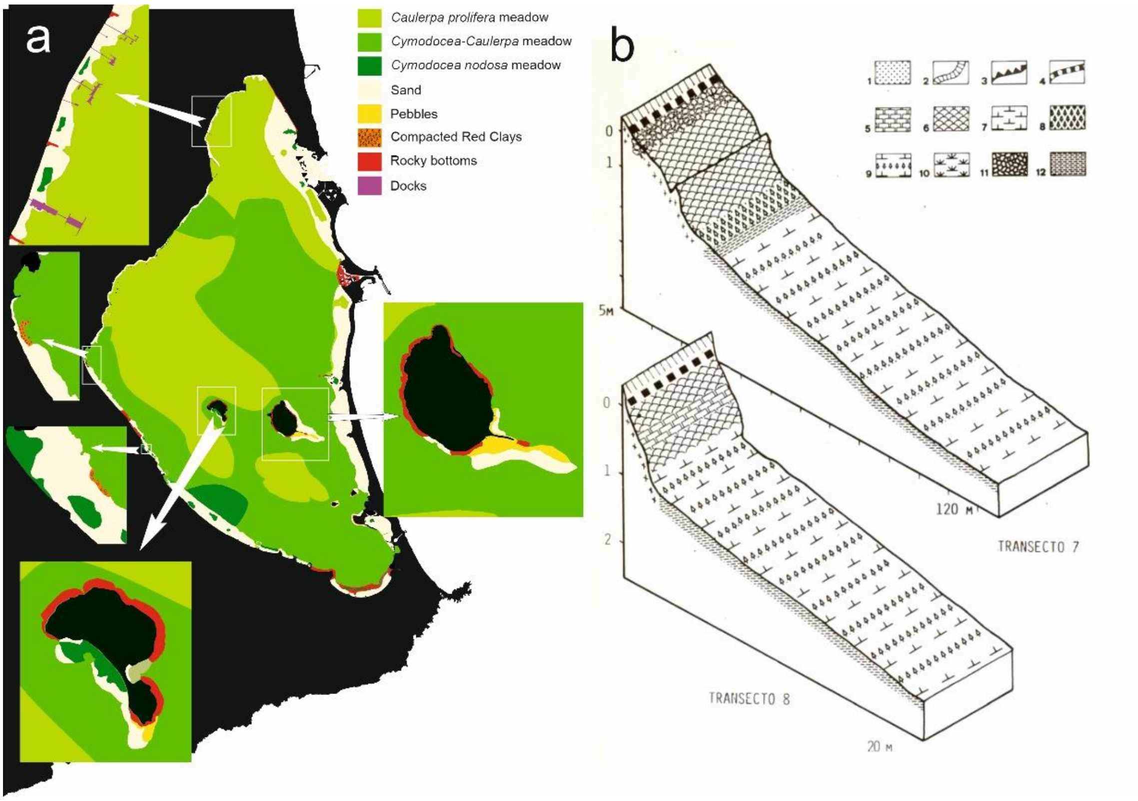
Source: Quispe (2014); Pérez-Ruzafa et al. (2007b).

3.1.5 Spatial intra- and inter-lagoon variability: vertical and horizontal gradients, habitats, geographical and biogeographical variability

In coastal lagoons, spatial variability responds to three main patterns: a vertical gradient marked by tides or waves, in the case of microtidal lagoons, and the intensity of environmental oscillations; a horizontal gradient marked basically by confinement and the probability of colonisation from the open sea; and both vertical and horizontal patchiness due to substrate distribution and water column quality (Fig. 24). These are overlaid by other gradients associated with changes in salinity or the influx of nutrients due to watercourses that flow temporarily or permanently into the lagoons, or environmental impacts due to coastal works and other activities.

For its part, as mentioned above when discussing biological populations, the variability between lagoons is due to, on the one hand, biogeographical reasons, but above all to the geomorphological, hydrographic and climatic differences between lagoons. Here too, the randomness introduced by the restricted connectivity of coastal lagoons leads to great variability between lagoons as a result of their evolutionary history, with migratory species being the only ones common to most lagoons.

Figure 24. Horizontal (a) and vertical (b) distribution patterns of benthic communities in the coastal lagoon of Mar Menor (taken from Pérez-Ruzafa et al., 2020b; 1988, respectively). b: Distribution of communities along the different transects carried out on Perdiguera Island. Key to symbols: 1: community of emerged sands; 2: community of supralittoral rock; 3 and 4: communities of upper and lower mesolittoral rock; 5: photophilic community of upper infralittoral rock in calm conditions with *Cystoseira*; 6: photophilic community of upper infralittoral rock in calm conditions without dominance of fucals; 7: *Cymodocea nodosa* meadow; 8: *Caulerpa prolifera* meadow; 9: mixed *Cymodocea-Caulerpa*; 10: *Ruppia cirrhosa* meadow; 11: loose infralittoral gravel; 12: community of muddy sands in calm conditions (see Fig. 20 for a picture of the main communities).



Source: Pérez-Ruzafa et al. (1988; 2020a).

3.2 Ecological and economic importance of coastal lagoons: good and services provided by coastal lagoons

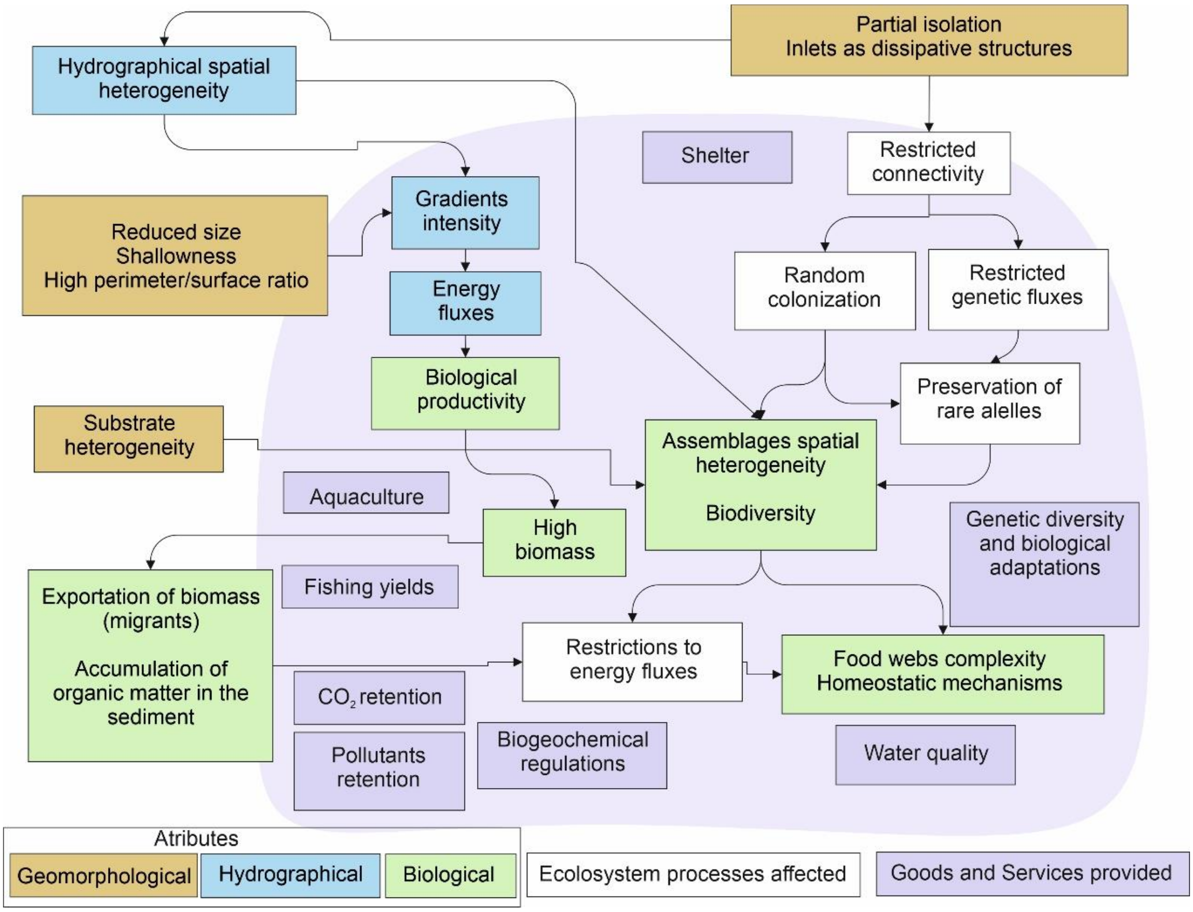
Coastal lagoons are transitional ecosystems between terrestrial and marine environments, characterized by varying degrees of isolation from the sea (Kjerfve; 1994) and pronounced physical, chemical, and biological gradients (UNESCO, 1981). Recent research has challenged long-standing assumptions regarding the ecological simplicity and uniformity of coastal lagoons (Pérez-Ruzafa et al., 2011b), and these systems are now understood to exhibit considerable ecological complexity and spatial heterogeneity, which are closely associated with their exceptionally high biological productivity and their notable resistance and resilience to both natural and anthropogenic pressures (Pérez-Ruzafa et al., 2019b).

Coastal lagoons harbour an important biodiversity (EEA, 2010a; De Wit, 2011; Pérez-Ruzafa et al., 2011b), sheltering a collection of habitat types for many species, and offering conditions for life cycle maintenance, while functioning as refugia, nursery areas, and feeding grounds for many marine organisms, or migratory and autochthonous birds (Sabetta et al., 2007; Pérez-Ruzafa et al., 2019b).

At the same time, the lagoon environment exerts a strong environmental selection on lagoon biological populations that leads to localized adaptations and often unique genetic profiles (Pérez-Ruzafa et al., 2008; Vergara-Chen et al., 2010, b; Milana et al., 2012). This makes coastal lagoons reservoirs of genetic biodiversity and potentially resilient to future environmental change.

As a consequence of their biodiversity values, productivity, and their geomorphological configurations, coastal lagoons play an important role in provisioning, regulating and supporting a wide range of ecological services and social benefits (Anthony et al., 2009; Pérez-Ruzafa et al., 2011c; 2019a; 2020a). Their influence on local economies is strong, but they are also important on a global scale because, despite the small individual size of most lagoons, they occupy 13% of the world's coastline (Barnes, 1980).

Figure 25. Causal relationships between the attributes of coastal lagoons (geomorphological, hydrographical, and biological), the ecosystem processes affected by these attributes, and the main ecosystem services and social goods provided.



Source: Modified from Pérez-Ruzafa et al. (2019a; 2020b).

Fig. 25 illustrates some of the intricate causal relationships between the primary attributes (geomorphological, hydrographic and biological) of coastal lagoons, the processes influenced by these attributes, and the associated ecosystem goods and services that they provide. These attributes can be affected by natural lagoon dynamics (e.g. sedimentation, redistribution processes, and species population dynamics), climate change consequences (e.g. hydrographic characteristics, sea level, water regimes, and invasive species), or human activities and their consequences (e.g. direct discharges of water, nutrients, pollutant and sediments or species introduction; activities in the catchment area; coastal works; and nautical and fishing activities). Thoroughly understanding the processes involved and monitoring the key variables that describe the different ecosystem compartments is necessary for adequately monitoring its ecological integrity and maintaining its goods and services, as well as assessing the effect of human activities and management measures.

In addition, coastal lagoons exert a significant biological and hydrographical influence on the adjacent coastal sea. Lagoon assemblages have an intense exchange of organisms and biomass with adjacent marine and fluvial environments. Despite the low probability of passive larval colonisation from the open sea (Pérez-Ruzafa et al., 2018b), continuous renewal of species composition (Quispe, 2014) ensures sustained connectivity and genetic flow between lagoon and coastal populations (González-Wangüemert et al., 2006; 2009; Vergara-Chen et al., 2010; González-Wangüemert & Pérez-Ruzafa, 2012; Hernández-García et al., 2015; Pérez-Ruzafa et al., 2019c; Fernández-Alías et al., 2022). This restricted yet persistent exchange acts as a selective filter, promoting local adaptation and the amplification of rare alleles within lagoon environments (Pérez-Ruzafa et al., 2011c; 2018b). Consequently, lagoons operate as both recipients and significant sources of genetic and biomass export towards the adjacent sea, thereby contributing to the resilience and adaptive potential of coastal marine populations.

In addition to their biological role, despite their negligible volumetric contribution to the Adriatic Sea (0.002%), they significantly modify regional hydrodynamics, enhancing tidal range by approximately 5% and current intensity by up to 10% (Ferrarin et al., 2017). These findings show that lagoons function as biogeochemical and physical regulators of coastal marine environments, thereby reinforcing their strategic importance in the structure and functioning of coastal socio-ecological systems (Pérez-Ruzafa et al., 2019b).

3.3 Pressures and threats to the ecological integrity of coastal lagoons

3.3.1 Human activities and management actions affecting coastal lagoons functioning

Coastal lagoons have been subject to long-standing human uses and anthropogenic pressures that have significantly degraded them and altered their natural dynamics. Mining activities dating back to the Bronze Age have contributed to the accumulation of heavy metals in lagoon sediments, as documented in systems such as the Mar Menor (Simonneau, 1973; Pérez-Ruzafa et al., 2023). Since the Middle Ages, agriculture and deforestation have increased sedimentation rates and driven major landscape transformations. During the 19th and early 20th centuries, many lagoons were drained or hydrologically altered for control of diseases or agricultural use, while more recent pressures include urban development, tourist activities, infrastructure construction, and hydromorphological modifications (Pérez-Ruzafa & Marcos, 2005; Doody, 2007; Pérez-Ruzafa et al., 2011c; Monfort et al., 2014; García-Oliva et al., 2018). Contemporary threats also include the introduction of non-native species, particularly through aquaculture, which can disrupt local biodiversity and ecosystem functioning (Cataudella et al., 2015). Some lagoons also host centers of industrial activity (such as Venice), and important port activities (such as the Curonian Lagoon). Globally, this translates in general degradation due to anthropogenic pressures such as nutrient over-enrichment resulting in eutrophication (Zaldívar et al., 2008), habitat destruction (De Wit, 2011), and contaminants inputs (Munaron et al., 2012) (Table 7).

Table 7. Main degradation causes and drivers in coastal lagoons.

Degradation Pathway	Main Causes
Colmatation (sediment infilling)	Increased sediment supply from watershed erosion (agriculture, urbanisation), reduced tidal flushing, engineering works that trap sediments
Erosion	Reduced sediment supply from watersheds (damming, river regulation), sea-level rise, increased wave energy, inlet stabilisation works that disrupt sediment budgets
Geomorphological alterations	Coastal works (dredging, inlet modification, embankments, reclamation, dredging and sand dumping for beach creation, and the construction of sand retention dikes or port facilities), subsidence (natural or anthropogenic), sea-level rise
Hydrological alterations affecting connectivity, currents velocity, wave exposure	Inlet modification (dredging, gates, narrowing), freshwater diversion, damming of tributaries, reclamation, coastal works
Changes in salinity (Increase or decrease of salinity).	Altered hydrological connectivity (inlet modifications), freshwater diversion, reduced or increased river inflow, sea-level rise, rising water table, climatic shifts (precipitation/evaporation changes)
Changes in temperature	Altered hydrological connectivity (inlet modifications), climate change
Pollution (nutrients and contaminants)	Agricultural runoff (fertilizers), urban wastewater discharges, industrial effluents, aquaculture operations, mining activities
Water quality degradation	Eutrophication from nutrient enrichment, hypoxia/anoxia, macroalgal blooms, phytoplankton blooms, dystrophic crises
Fishing yields changes	Altered hydrological and biological connectivity (inlet modifications), eutrophication, habitat loss (seagrass meadows), overfishing, ecosystem shifts from benthic to pelagic dominance.
Loss of homeostatic capability and loss of resilience	Chronic nutrient loading overwhelming natural buffering mechanisms (benthic filter feeders, seagrass uptake, sediment microbial processes), reduced biodiversity, homogenisation of habitats, hydrography and biological communities; coastal works; extreme alterations (due to excessive isolation or excessive exchange) in connectivity with the open sea

Degradation Pathway	Main Causes
Sediment degradation	Organic matter enrichment (from eutrophication), altered biogeochemical cycling (sulfide accumulation, heavy metal contamination), anoxia/hypoxia, siltation caused by coastal works and runoff
Loss of habitats	Eutrophication (replacement of seagrasses by macroalgae/phytoplankton), coastal works (reclamation, dredging), geomorphological alterations, hydrological modifications

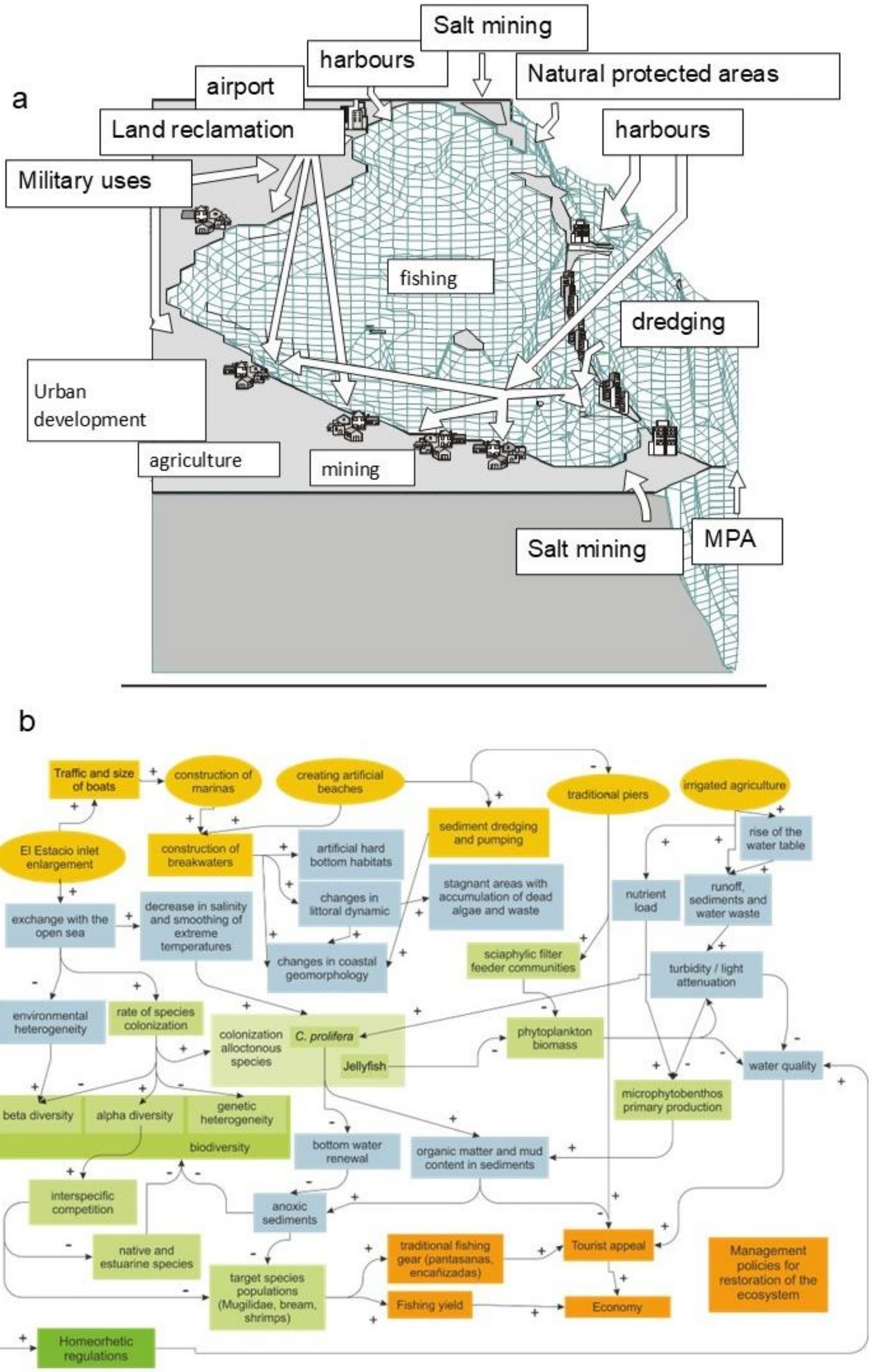
Source: Own elaboration.

However, not all of these activities pose the same level of threat to coastal lagoons, and some may even have beneficial effects. Thus, whilst fishing does not usually cause major impacts because the populations exploited come from the open sea and it is difficult to reach over-exploitation (Pérez-Ruzafa & Marcos, 2012), agriculture is one of the activities that exerts the greatest pressure on the ecosystem, mainly contributing nutrients, but it can also promote soil erosion and the entry of sediments and pesticides. At the same level in many lagoons is aquaculture, which not only contributes nutrients, but also alien and invasive species. Tourism, urban development, and associated coastal works depend largely on the socio-economic situation of the region in which they are located and can be a major source of impacts, for example in the form of (i) discharges of both nutrients (mainly P) or emerging pollutants, (ii) hydrocarbon spills, (iii) dredging and (iv) other geomorphological alterations induced by land reclamation, or the construction of dikes, marinas and harbors. For its part, salt mining can lead to an increase in habitat and biological diversity and act as an attractor for nesting, feeding and resting for a wide variety of migratory waterbird species.

Furthermore, many lagoons have different conservation statuses, particularly related to habitats of interest (Sites of Community Importance - LICs, Specially Protected Areas of Mediterranean Importance - ZEPIMs, etc.), sensitive species (such as *Pinna nobilis* in the Mediterranean) and waterbird populations (RAMSAR sites).

The vast majority of lagoons have many of these uses, pressures and protection measures. Each of these activities usually involves different management measures, with very different effects on the lagoon ecosystem, and both positive and antagonistic synergies. For example, marinas, coastal works on beaches, breakwaters and navigation channels built to promote tourism often have very negative effects on hydrodynamics, sediment and water quality, and the ecological integrity of the ecosystem, which in turn detracts from the area's tourist appeal. Similarly, dredging or closing some of the channels connecting to the sea is sometimes proposed as a fisheries management measure in some lagoons, or as a way to increase renewal rates to minimize the effects of eutrophication or other sources of pollution. However, these measures can have negative effects. Increased exchanges with the open sea beyond certain limits can lead to decreased fishing yields (Pérez-Ruzafa & Marcos, 2012; Pérez-Ruzafa et al., 2024), homogenization of biological communities with the consequent loss of self-regulation and resilience mechanisms (Pérez-Ruzafa, 2015), and changes in environmental conditions that favour the introduction of invasive species and diseases.

Figure 26. a: Human uses in the Mar Menor lagoon; b: main cause-effect relationships between human activities and management actions (yellow-orange boxes), hydrographic (blue boxes) and biological processes (green boxes).



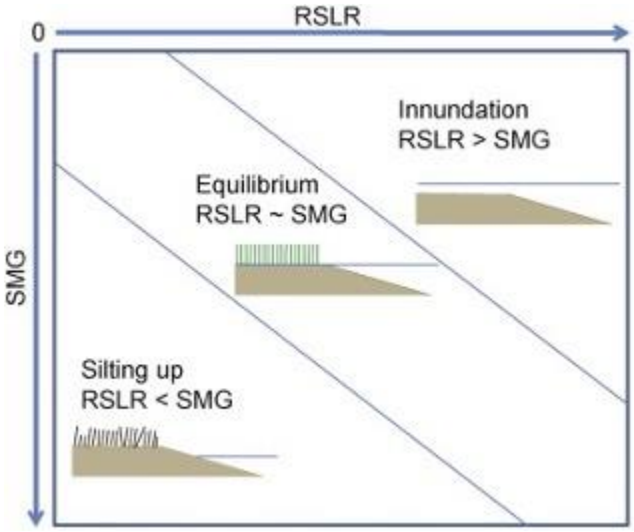
Source: Modified from Pérez-Ruzafa et al. (2005b).

3.3.2 Cause-effect relationships between pressures, responses and degradation pathways on the functioning of coastal lagoons: identification of key indicators for ecological assessment

From a geoscientific perspective, coastal lagoons are inherently ephemeral formations that exist as part of a continuum of coastal environments in different stages of formation and development, which, in turn, can be easily altered by humans. Thus, their natural hydromorphological evolution rarely unfolds without hindrance, primarily due to human activities, either directly through engineering works to maintain or establish navigable channels, or indirectly through activities within their watersheds (Duck & Figueiredo da Silva, 2012; Pérez-Ruzafa et al., 2019b). Numerous distinct processes contribute to the evolution of coastal lagoons; in some cases, a single process dominates, while in others, a variety of processes interact, each playing different roles. The relative importance of each process significantly influences the evolutionary trajectory of a given lagoon. The major processes include changes in the barrier headings and in the back barrier and can be found in detail in Barnes (1980), Carter & Woodroffe (1994), and Bird (2008), amongst other reviews.

Coastal lagoons are typically formed during periods of transgression when sea levels rise. However, the current rate of sea-level rise is much faster than what has been observed during previous geological periods (de Wit, 2011). The rate of sea-level rise or fall is critically important in the evolution of lagoon coastlines and the preservation of depositional facies. During rapid sea-level rise, barriers are often destroyed through breaching or overstepping, as they fail to adjust quickly to changes in sea level and sediment supply. In such cases, antecedent topography plays a minor role, and rapid erosional fronts develop. Conversely, with slower sea-level rise, barriers can adjust more effectively to the pre-existing topography, allowing the coastal cell structure to evolve gradually (Carrasco et al., 2016). Assuming a sufficient sediment supply, salt marsh growth will initially increase and then stabilize, at a rate matching sea-level rise (Fig. 27, central panel; Morris et al., 2002; Carrasco et al., 2016), with the marsh platform remaining just below the highest tide levels depending on local tidal conditions. If the growth rate is too slow due to low silt input, the lagoon's intertidal zone becomes dominated by inundation, preventing effective marsh development; excessive inundation can stress or kill vegetation, leading to peat collapse, erosion, and potential marsh drowning (Fig. 27, central panel; Hartig et al., 2002). Conversely, if growth surpasses sea-level rise because of high sediment input, the sediment-saturated marsh can shift horizontally, provided there is enough space for expansion (Fig. 27, lower panel; Carrasco et al., 2016). In this process, features such as protruding spits or barriers can extend further into the sea, reshaping the shoreline and lagoon morphology over time. As sea-level rises, the interior shore of a lagoon generally moves inland. However, man-made structures can impede this inland migration, leading to the submersion and incorporation of salt marshes and wetlands into the lagoon without forming new wetlands (Carrasco et al., 2021). Natural features like cliffs can also restrict lagoon movement in certain landscapes. This process, known as coastal squeeze, reduces the overall surface area of coastal landscapes and lagoons, as they become trapped between advancing shorelines and fixed inland boundaries (de Wit, 2011).

Figure 27. Conceptual scheme showing salt marsh response to relative sea-level rise (RSLR) and salt marsh growth (SMG).



Source: Carrasco et al. (2016).

Macro-scale drivers include sea-level changes, climate, and tectonic stability. Local drivers encompass fluvial sediment supply, longshore drift, salinity, surrounding geology, coast morphology, lagoon orientation, wave energy, and tidal range. Fluvial sediment yields to lagoons may be modified and accelerated by the reduction of vegetation cover, or the onset of soil erosion due to farming practices or mining activities in the watershed. On the other hand, there could be reduction of sediment yield to a coastal lagoon where a dam constructed on an inflowing river is intercepting sediment in the reservoir or by successful soil conservation works in the hinterland (Bird, 2008).

Shallow coastal lagoons are particularly vulnerable to climate change, with predicted increases in temperature of about 3°C and UV radiation of around 20% by 2100. These changes could significantly impact the pelagic food web (e.g. Vidussi et al., 2011) and alter species distribution patterns. For example, the seagrass *Zostera marina*, whose southern limit is in the Mediterranean Sea, might disappear from Mediterranean lagoons, potentially causing dramatic impacts on biodiversity, as it is a key ecosystem engineer. Additionally, maintaining high connectivity among lagoons is crucial to enable species migration and adaptation to a changing climate. In contrast, isolated lagoons are at greater risk of losing species and are less likely to be invaded by new species adapted to future climatic conditions (de Wit, 2011).

Other pressures, like pollution from sewage, agriculture, and industry, lead to a chain of effects on coastal lagoons, including nutrient enrichment, salinity shifts, and hypoxia. These state changes then drive degradation pathways, such as eutrophication, which causes algal blooms, and the loss of habitats and biodiversity. Responses involve both natural adjustments, like species shifts, and human-led management efforts to mitigate degradation and its impacts on the ecosystem and its services. Human activities have often led to the overexploitation of natural resources in coastal lagoons. This has generally caused a decline in biodiversity, a trend that has become particularly pronounced and dramatic since the 1950s (de Wit, 2011).

3.4 Monitoring coastal lagoons for ecological assessment

Key message
Tackling environmental problems not only requires the detection of harmful agents or the drivers that induce changes in ecosystems and their effects, but also the knowledge of their action mechanisms and the processes involved to design solutions, recover the damaged systems, and, above all, prevent any deterioration before it occurs. Adequate sampling designs and monitoring protocols are essential to performing adequate assessments and coping with natural variability.

3.4.1 Experimental sampling designs for impact assessment

Although robust experimental approaches for assessing the environmental impact of human activities have been available for decades (e.g. Underwood & Peterson, 1988), they are still not widely used in practice. There is a persistent mismatch between the methodological requirements needed to reliably detect ecosystem changes and attribute environmental impacts to human activities, and the criteria and requirements imposed by legislation and practices in management. Consequently, many impact assessments are methodologically deficient and often incomplete or biased, lacking basic biological quantification (Margalef, 1985). Reliable impact detection requires the use of appropriate controls and adequate replication, to distinguish true effects from natural variability and measurement error, ensuring the randomness of replicates to avoid pseudoreplication *sensu* Hulbert (1984). This implies having a good prior knowledge of the spatio-temporal scales of variability that affect the ecological processes involved and the variables that need to be monitored and quantified in order to make the diagnoses (Underwood & Chapman, 1996; Underwood, 1997). Effective monitoring programs in coastal lagoons must therefore define relevant variables and standardized measurement methods, as well as establish baseline conditions and ensure sufficient spatial and temporal replication, based on prior characterisation of natural system variability (Pérez-Ruzafa & Marcos, 2015). Basic techniques for sampling and analysing water, sediments or biological samples can be found in numerous publications and manuals (see, for example, Strickland & Parsons, 1972; Holme & Mcintyre, 1984; Parsons et al., 1984; Tyler et al., 2016). A review of Earth observation methods applied to coastal lagoons has recently been published (Lagoons for Life, 2025). In any case, rapid advances in analytical and observation techniques, covering both traditional parameters and emerging pollutants, make it advisable to periodically review the specialist literature in the various fields of study.

Therefore, it is essential that environmental impact assessments or diagnoses of the ecological integrity of lagoon ecosystems are based on experimental designs incorporating control locations or reference ecological statuses (Box 1). These designs must consider the variability and idiosyncrasies of these ecosystems to ensure adequate replication that incorporates the sources of variability that cannot be controlled as a fixed factor in the experimental design. This is important both within and between lagoon comparisons.

Box 1. Key considerations to correctly determine the ecological status of coastal lagoons.

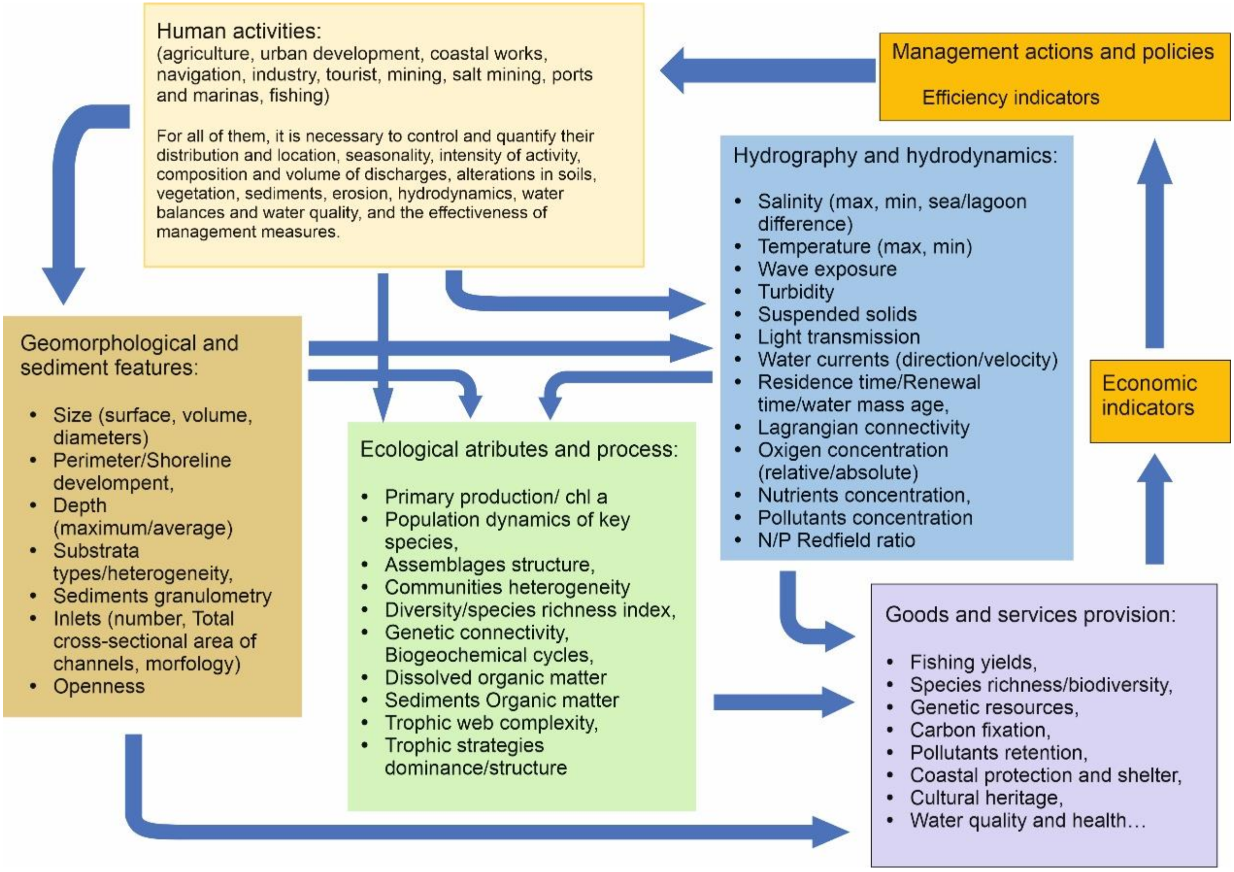
- Identify the scales of natural spatial heterogeneity (substrate types, influence of the sea and natural inflows from land, vertical zoning)
- Identify patterns and scales of natural temporal variation (daily, seasonal, interannual cycles, stage of ecological succession)
- Identify adequate controls and reference conditions, not affected by the treatment or analyzed impact
- Select sampling stations and times considering the sources of variability, fixing them as a factor and /or incorporating the scales of variability as random replicates
- Select the appropriate and relevant variables or indicators for the description of the processes and the diagnosis of the ecological state
- Selected variables must be sensitive to subtle yet ecologically meaningful changes in ecosystem integrity and functioning, while maintaining robust and consistent responses across spatial and temporal scales
- Differentiate between natural changes in species composition (which can be random, up to 40-50% interannual) and changes in species composition or population dynamics associated with anthropogenic impacts or ecosystem deterioration, or the presence of invasive species and new colonizers affecting ecological integrity
- Take into account that emblematic species requiring protection at a given time may have been new colonizers in previous eras due to climatic or anthropogenic changes (e.g. the seagrass *Cymodocea nodosa* or the mollusk *Pinna nobilis* in the Mar Menor lagoon).
- Consider indicators holistically and in a multivariate manner, while maintaining their individuality, contextualizing the temporal dynamics, range of variation, fluctuation patterns of each variable, and the interactions, synergies and antagonisms between them, in the context of the state of development of the ecosystem's ecological succession stage and its historical evolution

3.4.2 Key variables and indicators for detecting responses in coastal lagoons functioning

Whilst experimental sampling designs are essential for accurately assessing the ecological status of an ecosystem, evaluating the impact of human activities and measuring the effectiveness of management and restoration measures, it is equally important to select the appropriate variables that can serve as the best indicators of both the drivers and pressures acting on an ecosystem and the various states in which it may find itself and its responses to stress (Fig. 28).

There are numerous variables that determine and have a key role in the functioning and the hydrographical and biological characteristics of coastal lagoons (Fig. 28). These can be geomorphological (surface area, depth, complexity of the coastal perimeter and its relationship with the surface area, types and diversity of substrates, river mouths, connections with the sea), climatic (rainfall patterns, evapotranspiration, light radiation, wind patterns), hydrographic (salinity, temperature pH, dissolved oxygen, turbidity and stratification of the water column, currents, residence times, waves, tides), trophic (dissolved or particulate organic matter, nutrient inputs), or biological and many of them are essential for characterising the state of the ecosystem or diagnosing the impacts of human activities (Tables 8-11).

Figure 28. Main variables or indicators to consider for the diagnosis and management of coastal lagoons in a simplified model that relates human activities, geomorphological, hydrographic, and ecological characteristics and processes, goods and services provided, economic benefits, and management measures.



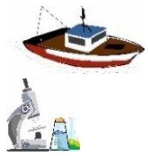
Source: Modified from Pérez-Ruzafa et al. (2020a).

Table 8. Morphometric and geomorphological variables and indices to characterize and categorize coastal lagoons that affect their ecological functioning. Variables marked with have been found relevant in some studies to explain the functioning of lagoons like the response to eutrophication, fish production (Joyeux & Ward, 1998; Pérez-Ruzafa et al., 2007a; 2024), assemblages structure and biodiversity (Sala-Mirete et al., 2025) (see Table 2 for symbol key).

Morphometric parameters	Abbreviations/ Formula	Parameter description	Measurement
Lagoon surface R	SLAG	Lagoon surface (km^2)	
Lagoon volume R	VLAG	Lagoon volume (m^3)	
Minimum diameter O	DMIN	Minimum diameter of the lagoon (m)	
Perpendicular distance O	DPER	Perpendicular distance to the open sea coast (m)	
Parallel distance O	DPAR	Parallel distance to the open sea coast (m)	

Morphometric parameters	Abbreviations/ Formula	Parameter description	Measurement
Perimeter O	PERI	Lagoon perimeter (km)	
Depth (maximum/ average) O	PMAX	Maximum depth in the lagoon (m)	
Mean depth R	PMED	Mean depth of the lagoon (m)	
Inlets characteristics O	CMAR	Number of inlets or channels	
Inlet width O	ACOM 2,3,4	$Inlet_{1, 2, 3, \ldots}$ width (m)	
Inlet length O	LCOM 2,3,4	$Inlet_{1, 2, 3, \ldots}$ length (m)	
Inlet surface O	SCOM 2,3,4	Inlet surface (m)	
Inlet cross section area R	ATRC (1,2,3,4)	Cross section area of $channel_{1, 2, 3, \ldots}$ (m^2)	
Inlet Mean depth O	PMEDC(1,2,3,4)	Mean depth of $channel_{1, 2, 3, \ldots}$ (m)	
Total inlets width O	ACOMT	Total width of inlets (m)	
Total inlets length R	LCOMT	Total length of inlets or channels (m)	
Total inlets surface O	SCOMT	Total surface of inlets or channels (m^2)	
Total cross section area O	ATRCT	Total cross section area of channels (m^2)	
Substratum types / heterogeneity E		% types of substrata: Hard bottoms (natural rock, artificial structures of concrete/wood/stone), sand, mud	 
Sediment Parameters			
Sediments granulometry E		Grain Size Distribution Results expressed as % gravel (>2mm), sand (2mm-50 μm), silt (coarse: 20-50μm, fine: 2-20μm), and clay (<2μm). (Duchaufour, 1975; Buchanan, 1984)	 

Morphometric parameters	Abbreviations/ Formula	Parameter description	Measurement
Organic Carbon / Organic Matter E	OC / OM	Amount of carbon derived from living organisms present in the sediment. Estimated by oxidation with dichromate (Walkley-Black method) and reported as a percentage (%) (Buchanan, 1984)	
Total Carbonates O	$CaCO_3$	Inorganic carbon content, primarily as calcium carbonate ($CaCO_3$) from shells and skeletal fragments. Reported as a percentage (%) by weight. Calcimeter Method	
Nitrate (in sediment) O		Concentration of nitrate in the pore water of the sediment, extracted as a saturated paste and analyzed by Ion Chromatography (IC; mg/kg or mg/L in extract). Analysis of saturated paste extract by IC. Richards Saturated Paste Method	
Total Phosphorus O		Digestion with sulfuric acid and potassium persulfate. Colorimetric measurement of absorbance at 885 nm (Parsons et al., 1984)	
Pesticides O		Concentration of pesticide residues adsorbed to sediment particles. Analysed after ultrasonic extraction by Gas Chromatography – Electron Capture Detector/Nitrogen-Phosphorus Detector (GC-ECD/NPD) and Gas Chromatography – Mass Spectrometry (GC-MS; μg/kg). Ultrasonic extraction and analysis by GC-ECD/NPD and GC-MS (same compounds as in water)	
Heavy Metals (Zn, Cd, Cu, Pb) O		Concentration of metallic elements (e.g. Zn, Cd, Cu, Pb) in the solid sediment matrix, measured after acid digestion by Atomic Absorption Spectroscopy (AAS; mg/kg). Microwave digestion with HF/HNO_3. Analysis by Flame (FAAS) or Electrothermal (ETAAS) Atomic Absorption Spectrometry	

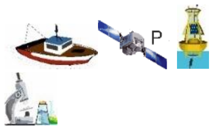
Morphometric parameters	Abbreviations/ Formula	Parameter description	Measurement
Polycyclic aromatic hydrocarbon (PAHs) O	PAH	Concentration of PAHs adsorbed to sediment particles. Multiple extraction with ether, analysis by High-Performance Liquid Chromatography with Diode-Array Detection (HPLC/DAD; same compounds as in water)	
Morphometric indices (Chubarenko et al., 2005)			
Restriction ratio 1 (PRES) O	$p_r = d/b$, $p_r = \sum d_i/b$	Restriction ratio is the ratio between the total width of the lagoon entrances (di) and the parallel shore direction (b)	
Restriction ratio 2 (PRES2) O	$p_r = \sum d_i/P$	Restriction ratio is the ratio between the total width of the lagoon entrances (di) and the lagoon perimeter (P)	
Orientation or shape (POR) O	$p_{or}=b^2/S_{lag}=S_{lag}/a^2$	The lagoon has orthogonal dimensions of the same order if por ≈ 1. It is more elongated in the parallel (b) or perpendicular (a) to shore directions if por ≥ 1 or por ≤ 1, respectively. Slag is the area of the lagoon surface	
Extreme depth (PDEP) O	$P_{deep} = h_{max}/h_{av}$	Provides information on the deepest part of the lagoon and how it compares to the mean depth	
Openness (PSEA) R	$P_{sea} = \sum S_i^{in}/S_{lag}$	Characterises the potential influence of the sea on lagoon general hydrology because flow velocities through the entrances are not included. Siin is the cross section area of i-th lagoon entrance for i=1, n entrances, and Slag is the area of the lagoon surface	
Shore development (PSH) R	$P_{shore} = l\,(4\pi A)^{-0.5}$	It is the ratio of the length of shore line, the lagoon perimeter (l), to the circumference of the circle whose area (A) is equivalent to that of the lagoon	

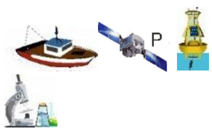
Source: Own elaboration.

Table 9. Hydrographical and physico-chemical parameters to characterize and categorize environmental conditions of coastal lagoons that affect its ecological functioning. Variables marked with * have been found relevant in most studies to explain the functioning of lagoons (response to eutrophication, fish production, etc.) (see Table 2 for symbol key).

Hydrographical parameters	**Measurement**	**Description**
Salinity (max, min, sea/lagoon difference) E		Salinity of the water in the lagoon
SMIN, SMAX E		Minimum and Maximum salinity in the water of the lagoon
DS R		Salinity range or difference between maximum and minimum salinity in the lagoon
Salinity difference with the sea* DSM E		Maximum absolute difference between the salinity values in the lagoon and those in open sea in the zone near the studied lagoon.
Temperature (max, min) TMXA, TMNA E		Maximum and Minimum temperature of the water in the lagoon (ºC)
Turbidity E		Measure of the cloudiness or haziness of water caused by suspended particles. Measured in Nephelometric Turbidity Units (NTUs).
Suspended Solids (TSS) E		Total dry weight of particles suspended in a water sample, retained on a pre-weighed filter (usually glass fiber). Measured in milligrams per liter (mg/L) (Strickland & Parsons, 1972)
Water Transparency / Secchi Depth E		Maximum depth at which a standard Secchi disk (white/black) remains visible from the surface. An indicator of light penetration and water clarity, in meters (m)
Light transmission / photosynthetically active radiation (PAR) / extinction coefficient / compensation depth* E		Different measures of light absorption in the water column, particularly photosynthetically active radiation (PAR), which determines the photosynthetic capacity of primary producers. The compensation depth is the depth at which the rate of oxygen production by photosynthetic organisms is exactly equal to the rate of oxygen consumption through the respiration of those same organisms. It determines the viability of microphytobenthos and macrophyte beds on the bottom.
Dissolved Oxygen* DO E		Concentration of molecular oxygen (O_2) dissolved in water. Measured in milligrams per liter (mg/L) or as percent saturation (% sat.)

Hydrographical parameters	Measurement	Description
Biochemical Oxygen Demand (BOD_5) O		Amount of dissolved oxygen consumed by microorganisms over 5 days in the dark at 20°C. An indicator of biodegradable organic matter. Measured in mg O_2/L. American Public Health Association (APHA) Standard Methods
pH R		Measure of the acidity or alkalinity of water on a logarithmic scale from 0 (acidic) to 14 (basic)
Hydrodynamic		
Wave exposure O		Wave Exposure = $\sum$ (mean wind velocity$_{22.5^\circ}$ * wind frequency$_{22.5^\circ}$ * effective fetch$_{22.5^\circ}$) where the effective fetch (F_e) is the direct fetch (F), or distance in kilometres along which the wind blows from each direction, corrected by fetches in directions of less than 45º using the equation, $F_e = [F_{(\alpha)} * \cos\alpha + F_{(\alpha + 22.5)} * \cos(\alpha + 22.5) + F_{(\alpha - 22.5)} * \cos(\alpha - 22.5)] / [\cos\alpha + 2 * \cos(\alpha + 22.5)]$ where α is equal to 0 for each F_e calculated. (Keddy, 1984; Pérez-Ruzafa et al., 2008)
Wave characteristics O		Upper limits for average wave height & period in shallow water: H_avr = 0.24 * (W_a² / g) T_avr = 2.63 * (W_a / g) where W_a is wind speed, g is gravity. These are depth-limited estimates
Water currents (direction/velocity) R		The speed and direction of water flow (3D numerical models)
Residence Time / Flushing Time / Renewal Time R		The time required for a measurable volume of water in a lagoon to be replaced. A measure of the system's self-cleaning capability. *Integral flushing time*: time for the total lagoon volume to be replaced. *Local flushing time*: the same concept applied to any compartment or grid cell
Concentration decay (for initial concentration C_0) O		$C(t)=C_0*\exp(-t/\tau\_flush)$

Hydrographical parameters	Measurement	Description
Retention time E		The average time molecules of incoming waters remain in the system
Lagrangian connectivity* R		Coupling a Lagrangian particle model to a hydrodynamic model and tracking particles released from each grid point.
Pollutants concentration		
Oils and Greases O		Hydrocarbon-based substances (mineral or organic) present in water. Quantified by infrared spectroscopy after extraction with an organic solvent (mg/L)
Detergents (Anionic) O		Surface-active agents (surfactants) that produce foam. Quantified by the Methylene Blue Active Substances (MBAS) method (mg/L). Standard Methods (5540C)
Phenols O		Organic compounds that can impart taste/odor and are often toxic. Spectrophotometric determination with 4-Aminoantipyrine and chloroform extraction (mg/L). Standard Methods (5530)
Heavy Metals O		Total concentration of metallic elements with high density (e.g. Zn, Cd, Cu, Pb) in an unfiltered water sample, measured by Electrothermal Atomic Absorption Spectroscopy (ETAAS; μg/L).Standard Method (ETAAS)
Pesticides (Organochlorine / Organophosphate) O		Specific synthetic compounds used for pest control. Analysed by Gas Chromatography – Electron Capture Detector/Nitrogen-Phosphorus Detector (GC-ECD/NPD) and confirmed by Gas Chromatography – Mass Spectrometry (GC-MS; μg/L).ISO 17025 Method
Polycyclic Aromatic Hydrocarbons (PAHs) O		A group of persistent organic pollutants with multiple aromatic rings, often from combustion. Analyzed by High-Performance Liquid Chromatography (HPLC) with UV/Fluorescence detection (μg/L). EPA Method 525.1
Total Coliforms O		Group of bacteria used as an indicator of fecal contamination and general water sanitary quality. Reported as Colony Forming Units per 100 ml (CFU/100ml). Membrane Filtration Method
Trophic parameters		

Hydrographical parameters	Measurement	Description
Nutrients concentration E		Grashoff et al., 1983; Parsons et al., 1984; SEAL Analytical, 2008
Nitrate* (N-NO_3^-) E		Oxidized form of inorganic nitrogen. A primary nutrient that can drive eutrophication (µmol/L or mg/L)
Maximum concentration of Nitrate NMAX E		Maximum concentration of nitrate in the lagoon waters (µg-at/l)
Minimum concentration of Nitrate NMIN E		Minimum concentration of nitrate in the lagoon waters (µg-at/l)
Nitrite (N-NO_2^-) R		Intermediate oxidized form of inorganic nitrogen, usually found in low concentrations
Ammonium* (N-NH_4^+) R		Reduced form of inorganic nitrogen, readily available for phytoplankton uptake. Measured in µmol/L or mg/L
Phosphate* (P-PO_4^{3-}) E		Inorganic form of phosphorus, a key limiting nutrient for primary production. Measured in µmol/L or mg/L
Maximum concentration of phosphate PO4MAX E		Maximum concentration of phosphate in the lagoon water (µg-at/l)s
Minimum concentration of phosphate* PO4MIN E		Minimum concentration of phosphate in the lagoon water (µg-at/l)s
Total P/Particulate P E		Relationship between TP(Dissolved reactive P (DRP) + Dissolved unreactive P + Particulate P) and PP (Particulate inorganic P (minerals) + Particulate organic P (detritus, algae)) (Labry et al., 2013; Serrano et al., 2017)
Silicate (Si-SiO_4^{4-}) R		Inorganic form of silicon, essential for the growth of diatoms (µmol/L or mg/L)
Chlorophyll-*a** Chla E		Estimated concentration of the primary photosynthetic pigment in phytoplankton, used as a proxy for algal biomass. Measured in µg/L or Relative Fluorescence Units (RFU)
Maximum concentration of chlorophyll* E		Maximum concentration of chlorophyll in the lagoon (mg/m^3)

Hydrographical parameters	**Measurement**	**Description**
Minimum concentration of chlorophyll* E		Minimum concentration of chlorophyll in the lagoon (mg/m^3)
Redfield ratio* N/P E		Proportion of N and P in water vs. primary producers' composition
Trophic Index (TRIX) R		Eutrophication indicator. Integrates chlorophyll-a, oxygen, total N and P (Wollenweider et al., 1998)
Dissolved Organic Matter (fDOM) R		Fraction of organic matter in water that is dissolved and fluorescent, indicative of terrestrial runoff or biological activity. Measured in Quinine Sulfate Units (QSU) or Relative Fluorescent Units (RFU)

Source: Own elaboration.

Table 10. Biological Communities, ecological attributes and process parameters to characterize ecological functioning and integrity. Variables marked with* have been found relevant in most studies to explain the functioning of lagoons (response to eutrophication, fish production, etc.) (see Table 2 for symbol key).

Biological communities, ecological attributes and processes	**Formula / Measurement**	**Description**
Primary producers		
Phytoplankton composition and abundance R		Taxonomic identification and cell counting of microscopic photosynthetic organisms (algae, cyanobacteria) in the water column. Reported as cells/L or relative abundance. EU Water Framework Directive (2000/60/EC).
Phytoplankton Biomass* E		Phytoplankton standing crop usually estimated via chlorophyll-*a* concentration (µg/L).
Phytoplankton Blooms R		Frequency (events/time) and intensity (e.g. cell density, Chl-*a* concentration) of proliferations of phytoplankton.
Primary production R		The amount of carbon biomass that primary producers create in a given length of time (GPP). Net primary production (NPP) is the rate at which all the autotrophs in an ecosystem produce net useful chemical energy after discounting the consumed energy by respiration.
Community structure*		

Biological communities, ecological attributes and processes	Formula / Measurement	Description
Benthos (Macroalgae) Structure R		Species composition, percent cover (%), and biomass (Orfanidis et al., 2001; 2003)
Macrophytes Ecological Indices R		Indices based on the distribution and abundance of key macroalgal species to evaluate the ecological status of coastal waters (e.g. Cartography of Littoral and Upper-Sublittoral Rocky-Shore Communities - CARLIT, Ecological Evaluation Index – EEI; Orfanidis et al., 2001; 2003)
Ecological Evaluation Index (EEI) R		Based on macrophyte coverage. Distinguishes two functional-form groups (Ecological State Groups – ESGs): — ESG I: Perennial, late-successional species (e.g. *Cystoseira* spp., *Chondrophycus tenerrimus*). — ESG II: Annual, opportunistic species (e.g. *Ulva* spp., filamentous algae). The index value is derived from the relative coverage of these groups (Orfanidis et al., 2001; 2003; 2008)
Expert Macrophyte Quality Index (E-MaQI) R		Based on the presence/absence of indicator species, each assigned a score: 0 = Tolerant taxa (to pollution) 1 = Indifferent taxa 2 = Sensitive taxa The index value is calculated from these scores. (Sfriso et al., 2009; Sfriso & Facca, 2011)
BENTHOS Index R		Based on the relationship between rocky shore community composition and environmental/chemical variables indicating high anthropogenic impact. Used to establish Ecological Quality Ratios (EQR; Pinedo et al., 2007; MED-GIG, 2007)
Fish Community Structure R		Species composition and abundance (number of individuals) of fish populations, assessed via visual censuses or trawling. Several indices have been proposed in the context of the EU WFD (2000/60/EC)
Benthic Biotic Indices R		Numerical indices calculated from macrofaunal species lists and their ecological assignments (sensitive/tolerant) to assess ecological quality (e.g. AZTI's Marine Biotic Index – AMBI, Borja et al., 2000; BENTIX, Simboura & Zenetos, 2002)
Diversity* E		Indicator of the structure of a community based in its species richness and abundance. Several indices have been proposed. Can be applied to the different

Biological communities, ecological attributes and processes	Formula / Measurement	Description
		taxonomic groups and assemblages or the community (Magurran, 2004)
Species Richness (S)* E		Simple count of the number of different macrophyte species present at a sampling station (Magurran, 2004)
Margalef index R	d = (S - 1)/ln(N), where S=species, N=individuals.	Margalef, 1958; Magurran, 2004
Shannon-Wiener Diversity Index (H')* E	$H' = -\Sigma(pi * \log_2 pi)$ where pi is the proportion of individuals or coverage of species i	Based on information theory. Measures species diversity, incorporating both richness and evenness. When $\log_2$ are used the units are bits/indiv (Magurran, 2004)
Population dynamics of key species		
Communities' heterogeneity O		Patterns of alpha, beta and gamma diversity of the different communities in a lagoon (Whittaker, 1960; Sala-Mirete et al., 2025)
Genetic connectivity O		Gene fluxes (Fst) between populations in different localities inside and outside the lagoon.
Trophic web complexity O	Production-to-biomass (P/B) ratio; Connectance Index; System Omnivory Index; Shannon diversity index	Estimators of the structure and functioning of food webs based on trophic models such as ECOPATH (Ulanowicz, 1986; Pérez-Ruzafa et al., 2020b)
Trophic strategies dominance / structure R		Relative dominance or diversity of different trophic strategies (Herbivorous/Deposit feeders/Suspension feeders/Omnivorous/Predators/Scavengers) in the community or ecosystem (Sala-Mirete et al., 2025)
Biogeochemical cycles and balances O		Oxygen, Carbon, Nutrients, or contaminant fluxes between ecosystem compartments (particularly sediment and water column)

Source: Own elaboration.

Table 11. Possible indicators to evaluate ecosystem services provision by coastal lagoons (see Table 2 for symbol key).

Goods and services provision		Description and measurement
Fishing yields	R	Commercial catches (kg)
Total catch	R	Total catches (kg)
Fishing effort	R	Effort made by the fishery. There are different ways of measuring effort: number of fishermen, number of boats, length of nets, number of gear types, etc. Comparisons to assess efficiency or trends must be made using the same units.
Catch per Unit Effort (CPUE)	E	Standardisation of catch based on effort
Catch per Unit Effort and lagoon surface (CPUEA)	E	Standardisation of catch based on effort and lagoon surface area expressed in CPUE/km^2
Species richness/biodiversity	E	Using the diversity indicators in Table 10 and Table A1 (Annex 2)
Water quality and health	E	Using water quality indicators in Table 9
Carbon fixation	E	Based on concentration ratios, analysing photosynthetic production and the concentration of organic and inorganic carbon in different compartments (dissolved in water, precipitated in sediments, skeletons and shells of organisms, in the form of living and dead biomass).
Pollutants retention	E	Based on bioconcentration ratios and factors, comparing pollutant inputs with those present in the water column, sediments, and organisms.
Coastal protection and shelter	R	Based on current and wave intensity using indicators from Table 9
Others Salt mining Health care and thalassotherapy Nautical sports, races and competitions Tourism activities Cultural heritage Landscape	O	These should be developed using biological or socio-economic indicators (i.e. number of users, users demand, total economic benefits of activity, willingness to pay, etc.)
Genetic resources	O	Through genetic indicators (genetic diversity, frequency of rare alleles or those exclusive to lagoon environments, gene expression of traits linked to resistance to environmental stress and climate change)

Source: Own elaboration.

All these variables (Fig. 28, Tables 8-11) are strongly interrelated, often acting as multistressors, showing complex interactions between them. They can present synergies or antagonistic effects, and depend on the latitude, climatic or biogeographic region in which each lagoon is located. In addition, the response times and lags of the cause-effect interactions that occur may differ depending on the location and the compartments of the system analyzed (Pérez-Ruzafa et al., 2019b; Lorente-González et al., 2026).

As an example of interaction between pressure variables, increases in temperature and salinity may enhance stratification in poorly mixed lagoons and reduce oxygen solubility, increasing the risk of hypoxia or anoxia (Pérez-Ruzafa et al., 2019b; Lorente-González et al., 2026). As another example, redox-driven sediment-water exchanges control nutrient and metal dynamics and can induce the release of secondary contaminants through sedimentary memory effects produced by old activities, underscoring the importance of legacy processes in lagoon management decisions (Viaroli et al., 2008; O'Higgins et al., 2014). The relative importance of these processes is amplified at air-water and sediment-water interfaces due to the high surface-area-to-volume ratio of shallow lagoons. Hydromorphology further governs lagoon functioning by controlling water circulation, residence time, sediment and pollutant transport, and biological connectivity, thereby shaping recruitment and the spatial and temporal structure of lagoon and adjacent marine populations (Ghezzo et al., 2015; Pérez-Ruzafa et al., 2019c).

The complexity of these interrelationships can hinder the detection of both the pressures on the ecosystem and of the impacts they produce, particularly in complex systems with top-down control of the trophic network in which nutrient concentrations can be kept low due to their utilisation by phytoplankton, which is in turn controlled by consumers, keeping low the values of the key variables (nutrients and chlorophyll *a*) used to detect eutrophication (Pérez-Ruzafa et al., 2002; 2005; 2019b).

All of this means that these variables cannot be used as indicators in isolation, but rather in a comparative and integrated manner, taking into account both spatial heterogeneity and temporal dynamics of each one.

These considerations must also be taken into account when establishing management or restoration measures, since the same measure (e.g. regulating freshwater inflows or dredging inlets) can have positive or negative effects on the functioning and ecological integrity of the ecosystem, its fish production (Pérez-Ruzafa & Marcos, 2012), or water quality, depending on the initial conditions, ecological status and lagoon characteristics. For example, dredging channels reduces residence times and exchanges with the open sea (García-Oliva et al., 2018; 2019), theoretically favouring the dilution of pollutants, but at the same time it softens extreme environmental conditions, reduces gradients with the open sea and favours the colonisation of new species, reducing environmental and biological heterogeneity and homogenising the ecosystem. The loss of gradients reduces fishing productivity (Pérez-Ruzafa et al., 2024), and the loss of complexity reduces homeostatic and self-regulating mechanisms (Pérez-Ruzafa et al., 2005b). Conversely, excessive or total isolation would cut off the minimum energy and biological flows necessary to maintain ecosystem functioning.

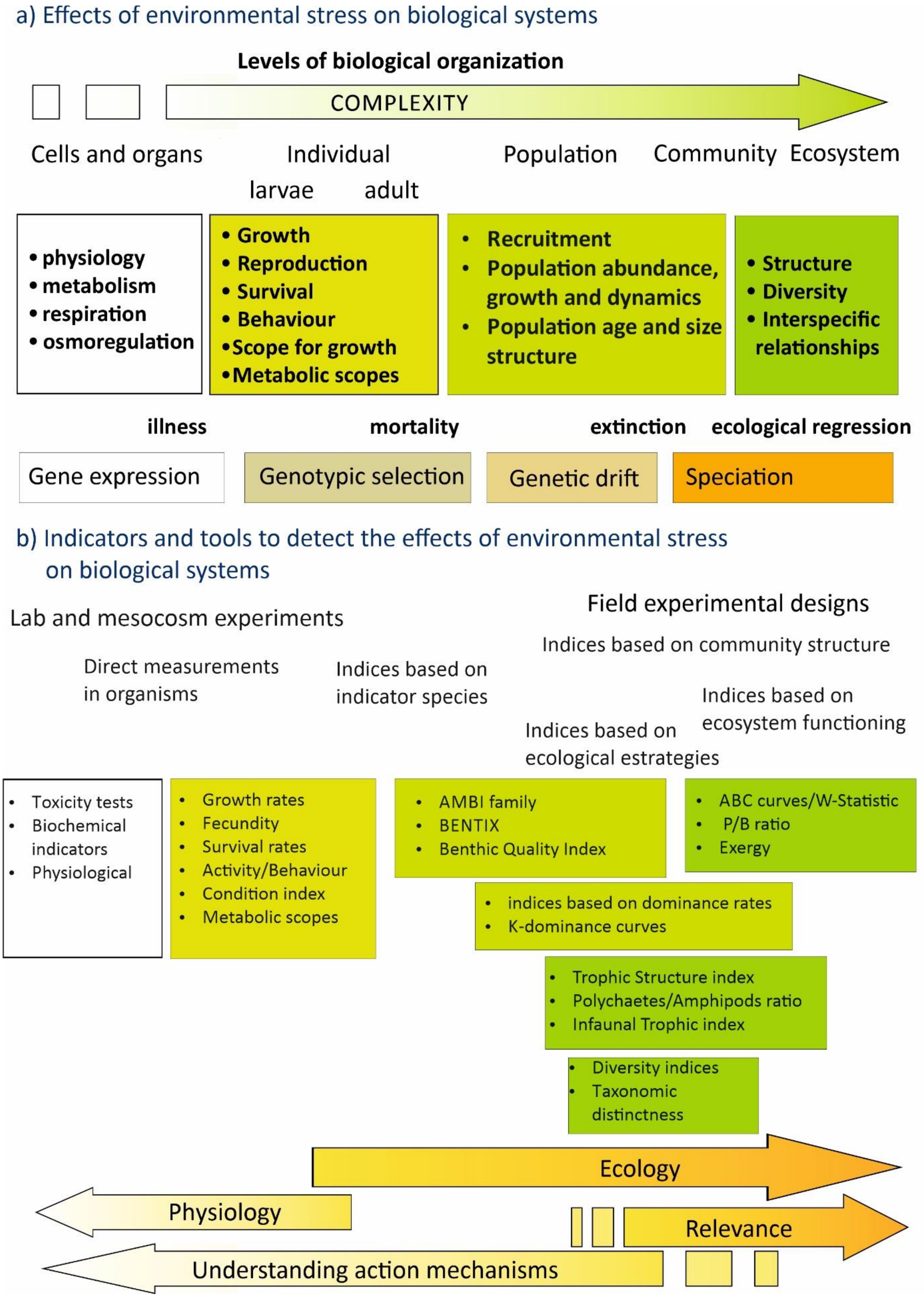
Another example of this lack of knowledge lies in the responsiveness of lagoon ecosystems to eutrophication and how much nutrients are needed for its functioning and when they are too much. In some lagoons, such as the Mar Menor, the response capacity to eutrophication is greater than traditionally assumed. This is driven by their intrinsic dynamism and ability to undergo rapid structural and functional changes. Self-regulation mechanisms rely on controlling phytoplankton through pelagic (e.g. gelatinous zooplankton) and benthic (e.g. increasing the dominance of filter and suspension feeders) pathways, while redirecting primary production toward benthic compartments. Enhanced abundances of filter-, suspension-, and deposit-feeding organisms promote the removal of organic matter from the water column, facilitating its export or sequestration in sediments, where hypoxic conditions can delay nutrient recycling. However, this responsiveness is modulated by species-specific traits such as life cycles and growth rates, as well as by broader system properties including habitat heterogeneity and connectivity with adjacent marine environments (Pérez-Ruzafa et al., 2026).

Understanding the thresholds within which the system can respond in one direction or another is one of the research challenges that must be addressed if we want to have reliable management and decision-making tools.

Regarding the response of biological populations and ecosystems to environmental or human pressures, the induced stress and pollutants affect living organisms at all levels of biological organisation (Maltby, 1999). In this way, biological indices to detect environmental impacts can be applied from cell to ecosystem scale (Fig. 29). However, from the management and decision-taking point of view, they provide different types of information (Pérez-Ruzafa et al., 2018b).

Lower levels of organisation provide mechanistic insights into biochemical and physiological responses (Corsi et al., 2003; van der Oost et al., 2003; Sarkar et al., 2004; Torres et al., 2008; Beyer et al., 2014; Dondero & Calisi, 2015), but their extrapolation to real consequences for populations or ecosystem levels is limited (Lloyd, 1991; Harding, 1992; Galloway & Depledge, 2001; Moore et al., 2004; Forbes et al., 2006). Conversely, higher levels capture ecosystem-scale impacts (Gray, 1982; Rombouts et al., 2013), being more relevant for diagnosis and management alerts, yet often obscure underlying mechanisms, constraining the development of effective mitigation and management strategies (Lloyd, 1991; Harding, 1992; Maltby, 1999; Sánchez & Porcher, 2009).

Figure 29. Biological processes and properties that are affected by environmental stress and pollution at different levels of biological organisation (a), and the main tools and types of bioindicators that can be used to detect and diagnose the impacts and changes occurring in them (b). It is generally advisable to implement multiple indicators at various levels in order to comprehensively understand the mechanisms of action and accurately assess the real impact on the ecosystem. In the case of integrative indices, the traceability of the variables that compose them should not be lost. Key to abbreviations: AMBI: AZTI Marine Biotic Index; BENTIX: Bentix index (Simboura & Zenetos, 2002); ABC: Abundance Biomass Curve; P/B: Production/Biomass.



Source: Modified from Pérez-Ruzafa et al. (2019a).

Traditional approaches and management tools used to determine the ecological status of coastal wetlands, as in the rest of continental waters, are based on tolerance and taxa richness of one or several biotic groups (e.g. Boix et al., 2005; Cañedo et al., 2012); or indicators based on pond or water characteristics (i.e. nutrients or other eutrophy indicators) including, in some cases, several items and some of them developed as rapid assessment methods (e.g. Vollenweider et al., 1998; Sala et al., 2004; Fennessy et al., 2007; Serrano et al. 2017). However, new proposals for tools or frameworks to improve ecological assessment have emerged in the last two decades. Five of these new approaches are size spectrum, aquatic metabolism, environmental DNA, biotic interactions, and metacommunity dynamics.

Size diversity index has been used to analyse biotic interaction and ecological temporal and spatial patterns of several assemblages in coastal wetlands (e.g. Badosa et al., 2007; Gascón et al., 2009; Quintana et al., 2015), and creation of indices to use them as a management tool have already been published (Reizopoulou & Nicolaidou, 2007; Quintana et al., 2008; Basset et al., 2021). The use of waterbody metabolism, estimated by means of high-frequency monitoring, first developed in lakes (e.g. Staehr et al., 2010; Obrador et al., 2014; Giling et al., 2017), has been recently implemented in coastal ponds (Bas et al., 2020). The results obtained using this approach are not only interesting for ecological assessment purposes, but also for phytoplankton ecology, such as seasonal succession, and the role of disturbance on pond zooplankton (Bas-Silvestre et al., 2024; Antón-Pardo et al., in press). Environmental DNA metabarcoding is a very powerful tool for conservation and management since it allows monitoring aquatic biodiversity and analyse the environmental impacts on community structure (Saenz-Agudelo et al., 2022; Wang et al., 2023; Xie et al., 2023) and has been used for detecting cryptic biodiversity in complex coastal lagoons like the Mar Menor (Corral-Lou et al., 2024). Ecological recovery following restoration is typically evaluated using metrics based on species diversity and composition (e.g, Badosa et al., 2008; Cabrera et al., 2019). However, increasing evidence suggests the success of long-term ecological recovery increases when more complex attributes were analyzed (Moreno-Mateos et al., 2015), such as ecosystem function indicators and biotic interaction networks (Cadier et al., 2020; Brown et al., 2021; Hernández-Carrasco et al., 2023). Finally, a new framework created to improve biological assessment, using metacommunities dynamics, has been designed for rivers, and although it seems a smart management-scientific suggestion, it is still not extended to coastal wetlands (Cid et al., 2020).

3.4.3 Integrating processes, variables and indicators in diagnostics and assessments

The existing technical guidelines for the assessment and monitoring of estuarine systems (habitat 1130) provide a partial framework that could be used as a reference for the development of protocols and variable selection for coastal lagoons (Gubbay & Garcia-Herrero, 2025). However, while these guidelines often define reference conditions and assessment indicators, their implementation remains largely based on expert judgement, resulting in considerable heterogeneity across ecosystems and among countries.

It is essential to undertake a more comprehensive evaluation of the relationships between indicator dynamics and the underlying ecological processes they represent. In particular, improved understanding of cause-effect linkages among indicators and their complex interactions is required to strengthen the assessment of ecosystem functioning and ecological integrity. As outlined in this document (Sections 3.3.1 and 3.3.2), this is particularly relevant in the case of coastal lagoons.

Coastal lagoons are essential ecosystems that require comprehensive assessments for effective management and conservation. Integrating processes, variables, and indicators in diagnostics is crucial to understanding their health and functionality. This integrated approach addresses both climatic and anthropogenic pressures, ensuring the continued quality and functioning of these vital ecosystems.

Coastal lagoons globally have suffered degradation due to anthropogenic pressures such as nutrient over-enrichment resulting in eutrophication (Zaldívar et al., 2008), habitat destruction (De Wit, 2011), and the inflow of contaminants (Munaron et al., 2012). In response, increased environmental awareness among citizens, alongside the implementation of Integrated Coastal Zone Management (ICZM) and specific EU legislation — including the WFD, the HD and the EU Birds Directive (2009) — has driven public policies within EU member states to prioritize valuing and protecting these ecosystems. These policies aim to enhance water quality, safeguard public health, conserve biodiversity, and optimise the ecological potential of coastal lagoons to provide essential ecosystem services. The WFD emphasises the ecological integrity of aquatic ecosystems, setting as its main goal achieving and maintaining 'good ecological status' in water bodies. Representing a comprehensive approach to water quality management, the WFD is recognised as one of the most significant pieces of European environmental legislation (Voulvoulis et al., 2017). Its primary objective is to improve the ecological status of water bodies by promoting a proactive strategy that aligns with the principles of restoration ecology. However, these EU directives have presented managers with significant operational challenges and created multiple barriers (Boutin et al., 2025). Firstly, mismatches between administrative and ecological boundaries weakened their ability to control key ecological processes, such as nutrient flows. Secondly, the proliferation of indicators, which are often perceived as being disconnected from local realities, has reinforced criticism of a 'management by numbers' approach. Thirdly, while the widespread use of regulatory exemptions was intended to adapt EU rules to local contexts, it frequently fueled persistent mistrust among stakeholders, especially in historically degraded environments. These challenges were further exacerbated by a siloed organisation of administrations, limiting coordination and adaptive management. Overall, these findings call for more integrated governance frameworks, a more critical and context-sensitive use of indicators, and greater transparency in derogation procedures (Boutin et al., 2025).

Assessing the health of coastal lagoons involves considering various variables and their interactions (Tables 8-11), such as:

— Water parameters, such as temperature, salinity, turbidity and dissolved oxygen, are essential for understanding the influence of different factors on ecosystems.

— Sediment parameters, including the assessment of sediment contamination through ecotoxicological studies. Biomarkers and experimental exposure methods provide additional insights into the impacts of pollution.

— Biological components to assess biodiversity and assemblage structure and to develop comprehensive biological indicators, or to understand the life cycle of key species.

To effectively manage lagoon ecosystems, it is crucial to integrate multiple indicators into a cohesive framework. This includes employing multifaceted indices that combine hydrological, biological and chemical data to accurately gauge ecosystem health; integrating indices based on diverse ecological metrics to allow for a better understanding of lagoon health; incorporating functional groups to provide valuable insights into ecological balance; and implementing automated systems to continuously collect data, thereby enhancing the management of lagoon dynamics and facilitating real-time monitoring of water quality and ecological indicators. Various studies highlight the implementation of integrated assessments in specific lagoons across different geographical locations, showcasing successful methodologies

The effective management and restoration of coastal lagoons require acknowledging societal values and stakeholder interests. Understanding how local populations perceive lagoon ecosystems and their desired states is vital for successful conservation efforts (De Leo & Gatto, 2001). Potential conflicts arising from competing uses, including fishing, aquaculture, tourism, and conservation/restoration initiatives, must be navigated by decision-makers (Souchu et al., 2000). Employing a multidisciplinary framework that incorporates ecological, hydrological and socio-economic parameters enables stakeholders to develop more effective management strategies. This integrative approach enables more accurate predictions of how ecosystems will respond to changes, enhancing the resilience of coastal lagoons and ensuring they can continue to fulfil their ecological functions and provide their services.

Ecological indicators are quantitative representations of the drivers, pressures, responses, or states of an ecosystem. They provide synoptic information on ecosystem structure/function (e.g. nutrient concentrations, biodiversity, and productivity). The usefulness of integrated indices is often justified on the basis that they combine complex environmental factors into single values for management and public communication and can bridge empirical research and modeling as orientors/goal functions. In this context, the concern to find or design appropriate indicators is longstanding, particularly since awareness of environmental degradation has been growing, permeating management policies (Meadows, 1998; Waltz, 2000). With these expectations the WFD revived the search for biological indicators.

However, at the same time, the use of integrated indicators also involves significant risks that should be avoided. Integrated indicators frequently combine variables with different units, with synergistic or antagonistic effects, and with response levels that are non-standardised or difficult to standardise, resulting in a complete loss of traceability regarding the role played by each variable and the processes involved. Due to oversimplification via aggregation, and potential for confusing interpretation, the integration of variables can mask, and even nullify, the complex relationships between the different factors that affect and determine the various ecological processes. It hides the real causes of the malfunctioning of the ecosystem, sometimes even preventing its detection.

As mentioned earlier, an accurate diagnosis cannot be made on the basis of a single variable or indicator. Furthermore, a good indicator should meet the following criteria: scientific soundness, ecological and social relevance, understandability, effectiveness for decision-making, cost-efficiency and the maintenance of the traceability of the relative effect of its components (Meadows, 1998; Waltz, 2000; UNESCO, 2003; Salas et al., 2006). At the same time, from an ecological point of view, it should be easy to handle, sensitive to environmental stress, independent of reference states, widely applicable, and relevant to policy/management.

The selection of the adequate indicators depends on pollution type, community type, data requirements, and availability. Particularly, the use and interpretation of the indices designed for coastal waters face significant challenges when applied to coastal lagoons.

On the one hand, most indices respond to both anthropogenic pressures (e.g. eutrophication) and natural environmental stress (salinity, temperature, confinement gradients), making it difficult to isolate human impact and differentiate it from natural stressors. There may be numerous **confounding factors** acting at different spatial and temporal scales that must be controlled in experimental sampling designs (see Sections 3.1 and 3.4.1).

On the other hand, the experiences regarding the suitability of the various indices proposed for analysing communities' responses to environmental impacts are highly heterogeneous. Although studies comparing different indices have been published for some coastal lagoons, we are still far from understanding the causes of these discrepancies and the conditions under which some indices work better than others and some indices respond better to one type of pollutants or stress than to others. For example, AMBI responds mainly to organic pollution, while Margalef or Shannon diversity indices respond better to a wide range of stressors (Salas et al., 2006). Furthermore, it is frequent to find index discrepancies in the same ecosystem under the same stress; for example, in a study in the Mar Menor lagoon, EEI and E-MaQI often assign different ecological statuses to the same site due to their distinct methodologies (coverage vs. presence/absence) and different classifications of species into functional or sensitivity groups (García-Sánchez et al., 2012). In the same study, tested indices respond mainly to variables related to natural environmental stress, and their relationship with trophic parameters as chlorophyll *a* or ammonium concentration became evident only after removing the influence of these variables from the analysis.

The discrepancies between indices that use indicator species - whether for vegetation or fauna - and their weak response to certain sources of stress or pollution, or the differences in sensitivity when applied in different locations, are due to the intraspecific variability in both their strategies and the ability of populations of those species to adapt to the environments in which they live. Indeed, the bioindicator value of many species is not well-understood in lagoon environments, and their classification into groups (e.g. opportunistic vs. sensitive) can be inconsistent or inappropriate depending of local adaptations of lagoon populations and ecosystem history.

In this context, a series of **recommendations** should be taken into account:

— The key message is that no single indicator is sufficient. A suite of complementary indices should be used to assess environmental quality, ecosystem health and integrity effectively, rather than relying on a single index.

— The selection or design of indicators based on species composition should take into account the type of disturbance (organic, physical, toxic), the species plasticity, and data quality and availability (taxonomic resolution, abundance, biomass, physicochemical parameters). In this context, although some structural indicators may work well at the taxonomic level of family (Mistri & Rossi, 2001; Pitacco et al., 2019a, b; Sala-Mirete et al., 2025), for many indices based on the species tolerance to environmental stress, pollutants, or organic enrichment, higher taxonomic resolution (species-level) improves accuracy. Therefore, an incorrect taxonomic identification of species can lead to significant differences in the ecological strategies assumed for the species on which many indices are based.

— In aquatic ecosystems, benthic parameters (especially sediment variables and macrobenthos) are preferred over pelagic ones for assessing ecological status for long-term and site-specific impact assessments. Macrobenthic (>500 μm) assemblages are recommended due to their sessile nature, that permits relate spatially the source of stress and the impact degree, and their ability to integrate disturbances over time. Meiobenthos (<500 μm) responds faster but is taxonomically challenging.

— Indices selected should be based preferably on quantitative data (e.g. abundance, biomass, coverage) than on presence/absence.

- Conducting local descriptive studies that take into account the methodological clues/pieces of advice in Section 3.1, to understand species' ecological responses in the case study conditions before applying indices is necessary before assigning a pre-established category. Relating to this, it is important to differentiate between adaptations to natural stress and human-induced stress when defining ecological groups.
- It could be necessary to validate and modify officially accepted indices (particularly those that have not been specifically designed for lagoon environments) within a lagoon-specific domain, as they cannot be directly extrapolated from open coastal waters.
- Preliminary studies are also necessary to control for spatial variability when selecting comparable habitats and environmental conditions to be used as controls in sampling designs, taking into account the methodological clues and considerations in Section 3.4.1.
- Any variable or index used must be placed in the context of its dynamics and scales of temporal variability with observation periods as long as possible in order to assess its natural trends and fluctuations.

Faced with the problem of integrating different variables into a single index, multivariate ordination techniques like Principal Component Analysis (PCA) - used to explore spatial variation of sampling sites based on environmental conditions reducing the dimensionality of environmental data (e.g. salinity, temperature, nutrients, wave exposure) and identify the main combinations of variables explaining the gradients of variation among sites - or Detrended Correspondence Analysis (DCA), used to visualize differences and affinities between species and sampling sites along environmental gradients; i.e. related to eutrophication/nutrient concentration along one axis vs. environmental stress/confinement, salinity and temperature along the other axis - can be an appropriate way to analyse the interaction of environmental variables, stress sources and descriptors of the ecological status of the community while maintaining the identity of each one. In a similar way, structural equation models (SEMs) can also be used to better apprehend the causal pathways in the system.

3.5 From monitoring and diagnosis to forecasting of environmental quality and ecological integrity in coastal lagoons

Here, we provide a summary of recommendations and steps to establish a standardized yet adaptive framework for monitoring coastal lagoons, diagnosing their ecological status, and developing predictive capabilities (including digital twins) by addressing spatial-temporal variability and multi-source pressures (see also Box 1).

3.5.1 Previous analyses

3.5.1.1 Characterization of variability and heterogeneity sources

Identify and map the key drivers of spatial and temporal heterogeneity (section 3.1):

Physical: Substrate type, water masses (salininity, temperature), depth, confinement gradients (isolation from the sea-water renewal rates), water courses, inlets.

Biological: Communities and assemblages (benthenthic, planktonic), confinenment gradients (probabilities of colonization and settlement; connectivity).

Anthropogenic: Human activities, pressures (nutrient inputs, waste, hydrological alterations).

Scales: Determine the spatial (e.g., meters to km) and temporal (e.g., hours to seasons) scales over which each indicator varies.

3.5.1.2 Experimental sampling design

Design a sampling strategy that captures (including all variability) or fixes (sampling in the same conditions) the identified variability sources at adequate spatial and temporal scales (section 3.4.1):

Site selection: Sectorization (basins/sectors/zones) according to detected sources and spatial scales of variability. (e.g., considering depth, inlets, water courses, inner lagoon, edges, substrate types, communities, human activities, etc.). Sampling sites according to stratified random sampling to cover spatial heterogeneity.

Sampling frequency: Matched to temporal variability (e.g., daily for dissolved oxygen, monthly/seasonal/yearly for benthic communities).

Replication: Consider a minimum number of locations/timings in each defined area and temporal scale to control stochastic variability or variability due to uncontrolled sources and to ensure comparability between situations.

3.5.1.3 Selection of key variables and indicators

To describe the different geomorphological, hydrological, biological and socio-economic processes involved in the functioning, ecological status and management of the ecosystem (section 3.4.2; Tables 8-11).

3.5.1.4 Historical context and successional status

Compilation of the historical evolution of climatic, hydrological, biological conditions and human activities that allow putting into context the ecological succession and long-term variability of the ecosystem. This is essential for establishing reference conditions and baselines.

3.5.2 Monitoring objectives

3.5.2.1 Regular monitoring system

The implementation of a standardized, long-term monitoring program must consider:

— Fixed stations that cover spatial heterogeneity (defined sectors and zones);

— Adaptation to variability scales (e.g., combine high-frequency sensors (high temporal resolution, low spatial coverage) + periodic field campaigns (lower temporal resolution, higher spatial coverage) + EO systems (high spatial and moderate temporal resolution);

— Use of harmonized protocols for intercomparability.

3.5.2.2 Diagnosis of specific problems and extraordinary events

Identify and select critical affected/impacted sites, controls and directional gradients. For specific impact assessments, define:

- Reference areas (ecologically equivalent, unaffected);
- Sources of stress and impacted zones;
- Spatial gradients to determine impact extent and recovery.

Also:

- Adapt sampling to the specific issue (e.g., eutrophication, urban or industrial wastes, coastal works or dredging and pumping sediments, hypoxia events, etc.);
- Use adequate experimental sampling designs;
- Establish spatial transects (from affected to control) and temporal frequency (pre-event; if possible, break, recovery) to determine spatial and temporal scales of the effects and recovery.

3.5.3 Forecasting and integration into a digital twin and predictive system: operational prediction and scenario testing.

3.5.3.1 Numerical models

These provide high spatio-temporal resolution. Requirements:

- Establish grid resolution and boundary conditions;
- Validate model predictions with regular field data;
- Provide input data with adequate temporal and spatial resolution for the modelled period according to spatio-temporal scales of the modelled processes.

3.5.3.2 Forecasting and developing a digital twin

This requires a modular architecture that integrates heterogeneous, near-real-time data streams from multiple sources (e.g., sensor buoys, piezometers, meteorological stations, and satellite imagery) into a unified and interoperable platform by coupling sectorial models:

- Meteorological models;
- Surface runoff and aquifer models;
- Hydrodynamic and lagrangian models;
- Sediment transport models;
- Biogeochemical models;
- Trophic models (and other ecological interactions models);
- Population dynamic model (important for key endangered species, IAS and fisheries);

- AI-driven predictive models: Global models (CatBoost, XGBoost) for multi-step autoregressive forecasting across multiple variables, and Specific models (Long Short-Term Memory; LSTM) for complex, non-autoregressive processes (e.g., streamflow response to rainfall);

The DT needs to be updated iteratively to improve predictive accuracy for ecological integrity indicators.

3.5.3.3 Key challenges to address (Diagnostic Quality Assurance)

When diagnosing ecological integrity and water quality, explicitly account for:

- **Scale mismatches:** Cause-effect relationships operate at different spatiotemporal scales across models;
- **Response times and magnitudes:** Quantify delays and amplitudes in cascading effects at different distances (e.g., rainfall → runoff → water table → salinity/nutrient inputs→ algal bloom → top-down responses);
- **Complex interactions:** Model synergistic or antagonistic effects, among variables with direct or indirect effects;
- **Integration of stochastic processes.** E.g., *confinement* as a probability function: likelihood of species colonization, establishment, and trophic integration.

3.5.4 Selection of data acquisition methods

Data acquisition methods should be chosen based on trade-offs in resolution and variable type (Table 12).

Table 12. Data acquisition methods and their trade-offs.

Method	**Temporal Resolution**	**Spatial Resolution**	**Use Case**
Fixed buoys and sensors	Very high (minutes-hours)	Very low (point)	Continuous physico-chemical monitoring
Field campaigns	Moderate (weeks-months)	Moderate	Biological and hydrographic sampling; sensors, EO and models validation
Earth Observation (EO)	Moderate (days-weeks, satellite-dependent)	High (meters)	Surface variables (chlorophyll, temperature, turbidity)
Numerical models	High (variable)	High (gridded)	Hindcasts, forecast, scenario testing (requires field validation)

Source: Own elaboration.

4 Knowledge gaps and future perspectives

4.1 Knowledge gaps for peat-forming inland wetlands

Despite the rapidly expanding evidence base on peatlands, several critical knowledge gaps continue to limit consistent assessment, restoration planning, and policy implementation across Europe.

- **Definition and mapping:** Peatland definitions vary between countries and regions, often reflecting agricultural rather than ecological or climate-related criteria. Common thresholds for peat depth range from 20-50 cm (national reporting), although shallower thresholds (e.g. 10 cm) may be more appropriate for capturing the full contribution of peatlands to climate regulation (UNEP, 2022). These differences strongly affect estimates of peatland extent, carbon stocks and restoration potential. In Europe, this has contributed to the underreporting of agricultural organic soils: in 15 of 28 countries assessed, the area of organic soils under agricultural use was underestimated compared with the European Peatland Map data, with agricultural peat soils estimated to be 37%, or 1.6 million ha, larger than officially reported (UNEP, 2022). More harmonized definitions, mapping standards and centralized data systems are needed to support inventories, greenhouse gas reporting, and restoration targeting (UNEP, 2022; UNEP, 2024; IUCN NL/REWET, 2025).

- **Classification and land use categories:** Some peatland types and regions remain poorly recognized in classification systems and policy frameworks. Highland peatlands, for example, are insufficiently represented in regional mire classifications and conservation strategies, despite their importance for biodiversity and water regulation (UNEP, 2022). In addition, tree-covered peatlands may be classified as forests, which can result in policies that support drainage or forestry development rather than peatland conservation (Gearey et al., 2025).

- **Biodiversity knowledge:** Species-level knowledge remains incomplete for almost all taxonomic groups, except birds and vascular plants, with particularly large gaps for invertebrates, fungi and microbes (UNEP, 2022; Flood et al., 2025). This restricts the ability to assess peatland condition using the full range of biological responses and may underrepresent biodiversity losses associated with degradation (Flood et al., 2025), or challenges with biodiversity improvement following restoration (Allan et al., 2024).

- **Hydrology and peat deposits:** Significant gaps remain in peatland hydrology, including site-level water-table dynamics, interactions with groundwater and basin-scale hydrology, and the status, depth and condition of peat deposits (UNEP, 2022). Peat soil properties relevant to different land uses are also insufficiently characterized (Flood et al., 2025). These gaps are critical because hydrology controls peat formation, degradation pathways and restoration potential. Field-based hydrological and soil measurements are needed to calibrate and validate EO-based indicators, and support evidence-based management.

- **Greenhouse gas emissions:** Data on peatland GHG emissions is fragmented, limiting the assessment of current condition, recovery potential and the climate benefits of rewetting (IUCN NL/REWET, 2025). Stronger links between field measurements, remote sensing and modelling are needed to improve GHG accounting and restoration evaluation (De Waard et al., 2024; Mursyid et al., 2025).

- **Monitoring, remote sensing and data sharing:** New mapping and remote sensing technologies offer major opportunities, but many predictive modelling approaches still lack robust validation and uncertainty assessment (De Waard et al., 2024; Flood et al., 2025; Mursyid et al., 2025). Standardized long-term monitoring remains insufficient, particularly for restoration outcomes, water quality effects, and dissolved organic carbon dynamics in peatland-draining streams. Better integration of remote sensing with field ecology, hydrology and watershed science is needed, alongside transparent data sharing between scientists, practitioners, authorities and policymakers.

— **Sustainable land-use transitions:** Practical knowledge on transitions from drainage-based peatland use to wetter land-use systems remains limited. Paludiculture and other wet land-use alternatives are increasingly recognized, but their adoption remains low and transition strategies are unclear in many contexts (Tanneberger et al., 2021; UNEP, 2022; Gearey et al., 2025). Further evidence is needed on how to implement, monitor and support these systems across different peatland types and socio-economic settings.

— **Policy, governance and finance:** Peatland conservation and restoration are often constrained by fragmented policy frameworks, conflicting objectives between agriculture, energy and land-use sectors, insufficient finance, and weak enforcement (Gearey et al., 2025). Effective implementation requires coherent peatland-specific policies, mapping and monitoring systems, enforcement mechanisms, and transparent finance models that can combine public funding with private investment without replacing public responsibility. In addition, more inclusive public consultation and stakeholder engagement are needed to ensure that restoration strategies are socially acceptable, context-specific, and informed by local knowledge.

Overall, these gaps underline the need for a coordinated monitoring and governance framework that combines harmonized peatland definitions, improved mapping, field-validated EO indicators, long-term hydrological and GHG monitoring, and better integration of biodiversity, ecosystem services and socio-economic data. At the same time, knowledge gaps should not delay restoration action. Restoration should proceed through adaptive planning, monitoring, evaluation and learning, so that management strategies can be updated as new evidence becomes available.

4.2 Knowledge gaps for coastal lagoons assessment and management

Managing coastal lagoons, particularly in the Mediterranean, is hindered by a critical lack of established cause-effect relationships between human activities and ecosystem degradation. Key limitations include the lack of baseline data and fragmented knowledge. On the one hand, in most of the cases there are insufficient historical data on pre-impact hydrographical and biological conditions, making it difficult to distinguish human-driven changes from natural variability. On the other hand, despite the opportunity offered by the implementation of the WFD, lessons learned from individual lagoons are not systematically integrated to develop predictive tools and conceptual models for guiding regional and European policies. Furthermore, monitoring programs often fail to keep up with the rapid, anthropogenic forcing of lagoon evolution.

As proposed by Pérez-Ruzafa et al. (2011c) coordinated, multi-national effort is essential in a context of local observatories and monitoring programs integrated in a European and transnational network. This requires establishing shared observatories and databases, fostering researcher-state cooperation, and embedding these priorities in international research programs to build the foundational knowledge needed for effective lagoon management

— **Predictive understanding of temporal dynamics:** A definitive understanding of long-term ecological succession and inter-annual community changes in lagoons is lacking. The high stochasticity of colonisation and environmental variability makes predicting temporal trajectories difficult.

— **Quantifying genetic connectivity and adaptation:** While lagoons are recognized as genetic reservoirs, the rates and scales of genetic exchange with adjacent marine populations, and the specific adaptive advantages of lagoon genotypes for future climate scenarios, require further quantification.

— **Integrated response to multiple stressors:** The complex, non-linear interactions (synergies/antagonisms) between simultaneous pressures (e.g. nutrient loading, warming, contaminant mixtures, physical modifications) and their compounded effects on ecosystem function are not fully understood.

— **Efficacy and context-dependency of management actions:** There is insufficient systematic evidence on why the same management measure (e.g. inlet dredging, freshwater inflow regulation) can have positive or negative effects depending on the initial lagoon state and characteristics. It is necessary to develop a robust theoretical framework capable of responding both to the processes common to all coastal lagoons and to the unique characteristics and spatial-temporal variability of each one.

— **Standardized classification and typology:** Under regulatory frameworks like the WFD, the classification of coastal lagoons remains ambiguous, especially for systems with minimal freshwater input. A more scientifically robust and globally applicable typology is needed.

— **Lagoon-specific validation of ecological indicators:** Most biotic indices were developed for open coastal waters or estuaries. Their performance, appropriate boundary values, and the correct ecological classification of lagoon-adapted species within these indices require extensive local validation. One of the most significant shortcomings stems from the lack of integration of the information supposedly generated after 25 years of implementation of the WFD. The amount of data and experience gathered from the multitude of systems monitored and diagnosed during this time should serve to fill many gaps in knowledge and uncertainties that still exist.

— **Mechanisms by which coastal lagoons manage their high biological production and cope with eutrophication:** The pathways and fate of the exceptionally high primary production in lagoons (e.g. transfer to higher trophic levels, export to adjacent seas, or burial) are not fully quantified, limiting understanding of their role in regional carbon and nutrient cycles. Indeed, the self-regulating capacity of coastal lagoons in response to eutrophication is based on channeling production towards the benthic system by adapting the dominant trophic strategies in communities, exporting biomass outside the ecosystem through seabirds, migratory species and fishing, and sequestering surplus production through the accumulation of organic matter in sediments. Knowledge of the triggers and response capacities of these mechanisms is essential for developing effective and non-counterproductive management measures.

- **Impacts of climate change at ecosystem scale:** While general threats are known (sea-level rise, warming), precise, localized projections of impacts on key species (e.g. seagrass thresholds), community reshuffling, and synergistic effects with existing pressures are major gaps. In particular, it is necessary to understand the adaptations and response capacities of the populations settled in each lagoon, since these adaptations occur through selection of the lagoon conditions in each case, and based on a genetic pool determined by restricted and random colonisation.
- **Social-ecological system integration:** The translation of ecological diagnostics into socially relevant management actions is challenging. Better integration of socio-economic indicators, stakeholder perceptions, and ecosystem service valuations into holistic assessment frameworks is needed.
- **Application of novel monitoring tools:** Promising new approaches (eDNA metabarcoding, high-frequency metabolism sensors, size-spectrum theory, metacommunity frameworks) are emerging but have not been widely standardized or integrated into routine monitoring and assessment protocols for lagoons. An essential aspect of developing predictive models, including digital twins, is having monitoring systems capable of covering the different spatial and temporal scales, and temporal lags in the response, at which ecological processes and the complex causal relationships between the factors involved take place (Lorente-González et al., 2026). Furthermore, these models must be capable of integrating and differentiating these scales in their predictions, in order to be an effective management tool, and to be able to accurately identify where impacts occur and where management measures should be implemented.

4.3 Towards a Wetland and Coastal Lagoon Watch System: from monitoring protocols to operational observatories, predictive systems and digital twins

4.3.1 Towards an operational aggregation and reporting framework

Reporting under Article 17 and the NRR ultimately consists of a final condition assessment per habitat type, integrating multiple sources of information at Member State level. Given the limited nature of exhaustive sampling, EO systems should:

- Reduce the reporting burden on Member States;
- Improve comparability between regions;
- Integrate into the assessment process through clear indicator combination rules;
- Define criteria for missing data and temporal consistency;
- Differentiate early warning indicators from consolidated change indicators.

This development requires validation with national experts through operational case studies, and the integrated analysis of databases and reports generated since the implementation of the WFD and their contrast with research work on the ecological functioning of the different ecosystems.

The development of an operational observation and prediction system for coastal lagoons and wetlands requires a transition from fragmented monitoring protocols towards an integrated architecture based on EO, *in situ* data, numerical modeling, and digital twins. This framework must support: (i) routine monitoring for impact detection, recovery assessment following management actions, and risk anticipation; (ii) adaptive experimental sampling designs to evaluate responses to extreme events or unexpected impacts; and (iii) the foundation for a future EU Wetland Watch service, built on local observation systems integrated into a European-scale network. Such a structure must characterize and control the inherent ecological variability of these ecosystems, thereby enabling early warning systems and management policies aligned with their conditions, singularities, conservation objectives, and the rational exploitation of the goods and services that they provide.

To that purpose, monitoring systems require a dual approach:

- Standardized regular monitoring: designed to capture relevant spatial and temporal heterogeneity;
- Event or problem-specific monitoring: based on adaptive experimental designs including affected areas, equivalent reference areas, and impact gradients.

This scheme requires explicit selection of complementary data acquisition methods:

- Fixed buoys and sensors: high temporal resolution, low spatial resolution;
- Field campaigns: intermediate spatiotemporal resolution;
- Earth Observation (EO): high spatial resolution and synoptic coverage, with sensor-dependent temporal resolution;
- Numerical models: high-resolution spatiotemporal integration, conditioned by input data quality and *in situ* validation.

4.3.2 Models and transition towards digital twins

The implementation of a Digital Twin (DT) for coastal lagoon ecosystems, such as the Mar Menor, requires a modular architecture that integrates heterogeneous, near-real-time data streams from multiple sources (e.g., sensor buoys, piezometers, air quality stations, and satellite imagery) into a unified and interoperable platform (Ye et al., 2026). Therefore, the evolution towards DTs demands the integration of sectorial models (meteorology, surface and subsurface hydrology, hydrodynamics, lagrangian, sediment transport, biogeochemistry, and ecological dynamics).

This objective faces significant structural limitations:

- Multi-scale integration of cause-effect processes across heterogeneous models;
- Representation of time lags and cumulative effects, particularly in cascade processes (i.e. precipitation - runoff - aquifer - wastes (nutrients, pollutants) – hydrography - biological interactions);
- Quantification of non-linear, synergistic, and antagonistic interactions among ecological variables;
- Inclusion of critical stochastic processes related to lagoon confinement, understood as the probability of species colonization, settlement, and persistence, and their role in the trophic web.

These factors currently constitute a bottleneck for fully integrated operational forecasting systems at both local and European scales.

4.3.3 Structural limitations of current systems

Progress towards an operational watch system faces fundamental constraints:

— Insufficient spatial and temporal resolution of EO products relative to functional ecological units;

— Scarcity of harmonised *in situ* data for calibration and validation across lagoon typologies and biogeographical regions;

— Lack of long-term baseline and time series to establish interannual variability, stages of ecological succession and degradation trajectories.

These limitations must be explicitly taken into account in the design of the system, as they directly affect the robustness of the derived products and their use in regulatory assessments under the WFD, particularly when applying the 'one out, all out' aggregation rule, and ultimately, may result in water bodies being incorrectly classified as 'degraded' when they actually reflect their particular natural geomorphological or hydrological conditions.

4.3.4 Priority research needs

To overcome these barriers, several priority lines should be considered:

— Development and calibration of a combination of EO proxies, modelling and in-situ network of sensors for non-directly observable variables, including:

  - Water table depth, and its influence on surface and groundwater flows to wetlands and coastal lagoons, including under conditions of torrential rainfall, or with repetitive patterns, as well as the delay and persistence times of the flows,
  - Sediments and soil properties (organic vs. mineral components, including heavy metals),
  - Biogeochemical fluxes in interfaces and ecosystem boundaries,
  - Species colonization (related to confinement and restricted connectivity,
  - Adaptability and functioning of food webs in a context where a high proportion of their components undergo random changes.
  - Energy balances (kinetic energy versus work produce due to restricted connectivity) in the communication channels between the lagoon and the open sea

— Definition of quantitative regional thresholds for coastal lagoon indicators.

— Construction of an operational aggregation framework compatible with European reporting requirements (including Article 17).

4.3.5 Standardization of indicators and need for operational ranges

Although existing data and expert knowledge provide a good understanding of how key variables behave under both reference and impacted conditions, further work is needed to strengthen the indicator system for coastal lagoons. In particular, the relationships between environmental pressures, indicator values, and ecosystem responses need to be quantified, including the identification of critical threshold values that may signal significant ecological change. Some examples are:

- Critical levels of variables related to eutrophication: Chlorophyll-a ranges by trophic status (oligotrophic–mesotrophic–eutrophic) adapted to lagoon types, or the maximum (or minimum) loads that an ecosystem can withstand (or needs) to maintain its communities and fisheries yields;
- The role of N/P ratio imbalance in hypoxia events;
- The water and sediment conditions that determine the bioavailability of pollutants retained in sediments;
- Permissible alterations in geomorphological alterations, or in the inlets;
- Operational classifications of confinement (open, semi-enclosed, closed);
- Associated thresholds for salinity and expected biological communities;
- The effect of salinity on the capacity for homeostatic responses to eutrophication and the risk of dystrophic crises.

All these aspects still need to be evaluated and adapted to the geomorphological, climatic, hydrographic, and biological conditions and characteristics of each lagoon. Without this standardization, EO-based assessment systems cannot be robustly integrated into services such as Copernicus Land Monitoring Service (CLMS) Lake Water Quality or into European regulatory frameworks.

A critical issue is that WFD-type indices applied to coastal lagoons respond primarily to natural gradients (confinement, salinity, extreme temperature), while the anthropogenic component emerges only after statistically removing these factors. This implies that any EO system integrated into WFD-type assessments may overestimate degradation in naturally confined lagoons or when comparing lagoons with different salinity or size, generate biases amplified by one-out-all-out aggregation rules, or induce classification errors under Article 17. This problem must be explicitly considered in the design of the assessment system.

4.3.6 Roadmap towards the EU Wetland Watch Service

The results described in this report should be understood as an intermediate step towards an operational European Wetland Watch service. Its consolidation requires:

- Integration of existing efforts in Member States (avoiding duplication);
- Joint participation of ecologists, modelers, and EO experts;
- Definition of operational needs from the perspective of end-users of reporting;

In this context, a European workshop (DG ENV, KCBD, BioAgora; Brussels, 2026–2027) is proposed to:

- Present existing national systems;
- Validate the EO–ecology translation approach;

— Define the service architecture;

— And evaluate its integration into EU reporting processes.

5 Conclusions

This report provides a scientific basis for the ecological characterization of two contrasting Annex I wetland habitat groups: peat-forming inland wetlands and coastal lagoons. It demonstrates that meaningful assessments require more than mapping habitat extent or recording species composition; conditions must be interpreted through ecosystem functioning, pressure-response relationships, natural variability, and the ecological context in which indicators are applied.

For each habitat group, the report identifies key functional processes, pressures, degradation pathways, and candidate indicators, structured into tiered sets of essential and complementary variables with guidance on the role of EO versus *in situ* measurement. In both cases, no single indicator is sufficient; robust assessments require multiple lines of evidence combining field-based measurement, remote sensing, and expert ecological interpretation.

This report should be understood as a foundation for assessment, not as a fully operational protocol. A critical remaining requirement is the development and calibration of habitat-specific thresholds capable of distinguishing favourable condition, early degradation and severe degradation, across the range of peatland types, lagoon contexts and biogeographical regions. This was beyond the scope of the present work and would require coordinated analysis of monitoring data, field validation and systematic use of reference conditions across Member States.

Long-term conservation and restoration of these sensitive and important ecosystems will depend not only on biophysical assessment but also on effective socio-ecological governance that integrates stakeholder interests, competing uses and transparent management. The frameworks and indicator logic presented here can support more consistent, functionally-grounded assessments under the EU policy frameworks (HD, WFD and NRR). Translating this scientific basis into practical assessment protocols requires refining indicator sets, developing and validating threshold ranges across habitat types and biogeographical regions, improving monitoring network representativity, and establishing transparent decision frameworks for combining indicators into conclusions about condition and degradation risk. These are the necessary next steps to support conservation status assessment, restoration prioritization, and long-term monitoring of wetland recovery. These goals should be reached through coordinated transnational monitoring networks, harmonized observatories, and European-scale databases to transform fragmented data into predictive frameworks that give scientific support to adaptive management, restoration priorities, and to a consistent implementation of EU environmental policy.

The development of an operational observation system for coastal lagoons and wetlands requires not only technological integration, but also a conceptual transformation: a shift from ecological descriptions and fragmented monitoring towards a predictive, calibrated, and operationally relevant system. The main challenges are not exclusively technical; they also involve methodological and ecological harmonization. Overcoming these challenges will require simultaneous advances in the translation of ecological variables into EO proxies, the definition of operational thresholds, and the multi-scale integration of models. Only then will it be possible to build a coherent, scalable, and effective system for assessing the conservation status of European wetlands.

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List of abbreviations

Abbreviations	Definitions
AMBI	AZTI Marine Biotic Index
DOC	Dissolved Organic Carbon
DT	Digital Twin
DTM	Digital Terrain Model
EEI	Ecological Evaluation Index
E-MaQI	Expert-Macrophyte Quality Index
EO	Earth Observation
GHG	Green House Gases
HD	EU Habitats Directive (92/43/EEC)
IAS	Invasive Alien Species
KCBD	EC Knowledge Centre for Biodiversity
NEE	Net Ecosystem Exchange
NRR	EU Nature Restoration Regulation (2024/1991)
PAH	Polycyclic Aromatic Hydrocarbons
POC	Particulate Organic Carbon
SAR	Specific Absorption Rate
WFD	EU Water Framework Directive (2000/60/EC)
WT	Water Table

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7 Annexes

Annex 1. Glossary

Bioindicator: A species, group of species, or biological trait whose presence, absence, abundance, or physiological condition provides quantifiable information on environmental quality or anthropogenic stress, used to infer ecosystem health.

Bog: A bog is a type of peatland that is rainwater fed (ombrotrophic), and therefore very acidic and poor in nutrients (Ramsar Convention on Wetlands, 2018; UNEP 2024).

Coastal lagoon: A shallow, coastal water body separated from the open ocean by a barrier (e.g. sandbar, barrier island), with restricted exchange through one or more inlets; generally shallow and typically characterized by salinity ranging from brackish to hypersaline depending on climatic and hydrological conditions.

Confinement**:** A hydrological gradient describing the degree of restriction in water exchange with the open sea, integrating tidal amplitude, freshwater input, and residence time as a key ecological driver in transitional waters. It has been also related to the probability of colonisation of a lagoon or the different areas inside it by organisms from the open sea.

Connectivity**:** The degree of hydrological, biological, or material exchange among aquatic habitats or between a water body and adjacent systems (e.g. ocean, watershed), influencing population dynamics, nutrient flux, and ecosystem resilience.

Earth observation (EO)**:** The collection, processing, and analysis of information about Earth's physical, chemical, and biological systems via remote sensing technologies (e.g. satellites, airborne sensors, in situ platforms) or direct measurement, enabling synoptic monitoring of environmental variables, land cover, habitat distribution, and ecosystem change across spatial and temporal scales.

EO-direct**:** Earth observation data obtained through direct, in situ measurements or proximal sensing techniques (e.g. field surveys, deployed sensors, drone-based imagery) that capture environmental variables at high spatial and temporal resolution without intermediary inference or modeled approximation.

EO-proxy**:** Earth observation data derived from remotely sensed signals (e.g. satellite reflectance, backscatter, thermal emissions) that serve as indirect surrogates or modeled estimates for environmental variables (e.g. chlorophyll-*a* concentration as a proxy for phytoplankton biomass, sediment reflectance as a proxy for benthic composition), requiring calibration, validation, or transformation to represent the target parameter.

Estuary**:** A semi-enclosed coastal body of water with a free connection to the open sea, within which seawater is measurably diluted by freshwater derived from a river.

Fen: A fen is a peatland that, in addition to precipitation, receives water that has been in contact with mineral soil or bedrock (e.g. groundwater or surface runoff). As a results, fens are generally less acidic and richer in nutrients than bogs (Ramsar Convention on Wetlands, 2018a; UNEP, 2024).

Gradient**:** A measurable, unidirectional change in an environmental variable (e.g. salinity, oxygen, confinement) across spatial or temporal scales, structuring ecological communities and ecosystem processes.

***Inlet*:** A narrow water passage connecting a semi-enclosed coastal basin (e.g. lagoon or estuary) to the open ocean, governing tidal exchange and hydrological connectivity.

***Levels of biological organisation*:** The hierarchical framework structuring biological systems from molecular to biospheric scales, encompassing genes, cells, organisms, populations, communities, ecosystems, and the biosphere. As we move through the various levels, biological and structural complexity increases, which typically results in a greater capacity for self-regulation within the system and greater efficiency in the management of resources and energy flows.

***Macrobenthos*:** Benthic organisms retained on a 0.5 mm or 1.0 mm mesh; includes polychaetes, mollusks, crustaceans, and larger fauna.

***Meadow*:** A continuous or patchy benthic assemblage dominated by seagrass or macroalgae, characterized by cohesive spatial structure, high primary productivity, and provision of complex habitat for associated fauna.

***Meiobenthos*:** Benthic organisms that pass through a 0.5 mm mesh but are retained on a 0.063 mm mesh; predominantly metazoans such as nematodes and copepods.

Mire: A mire is a peatland in which peat is currently accumulating; i.e. an actively peat-forming peatland (Ramsar Convention on Wetlands, 2018a; UNEP, 2024)

Peat: is the organic deposit made up of plant material that has accumulated where it grew (*in situ*) and has only partly decomposed. It forms under almost permanently waterlogged, oxygen-poor, and under low-temperature conditions, where microbial activity is suppressed and only the more decay-resistant parts of certain plant species persist, gradually building up a layer of recognizable but partially decayed plant remains (Ramsar Convention on Wetlands, 2018a; UNEP, 2022).

Peatland: In this report, *peatlands* are defined as areas of land where a natural layer of peat has built up at or close to the surface. Peatlands develop where the ground is saturated with water for long periods, so that oxygen is limited and the decomposition of plant material is strongly slowed, allowing peat to form and build up over time. The term covers both peat-forming ecosystems that are still actively accumulating peat ("mires"), as well as degraded areas where peat formation has stopped and the remaining peat is now being lost (Ramsar Convention on Wetlands, 2018a; UNEP, 2022).

***Pseudoreplication*:** The use of inferential statistics to assess differences between treatments when the samples used as replicates are not truly independent. This occurs when multiple measurements are taken at the same location or time, such that spatial or temporal replicates are treated as independent replicates but do not actually capture the system's true patterns and scales of variability; this increases Type I error rates and compromises the validity of statistical inference and the results.

***Residence time*:** The average duration that a water parcel or dissolved constituent remains within a defined water body before being flushed out; a key hydrological parameter governing nutrient retention, oxygen dynamics, and ecosystem susceptibility to eutrophication.

***Sampling experimental design*:** The structured framework defining the selection, spatial arrangement, temporal frequency, and replication of sampling units, intended to enable statistically robust inference, minimize bias, and allow discrimination of treatment effects from natural variability.

***Seagrass*:** Flowering angiosperms adapted to complete submersion in marine or brackish environments, forming extensive underwater meadows that provide critical habitat, sediment stabilisation, and carbon storage.

***Transitional water*:** A collective term established by the European Water Framework Directive for estuarine, lagoonal, and other coastal water bodies with intermediate characteristics between freshwater and marine systems, characterized by gradients in salinity, confinement, and ecological structure under significant terrestrial–marine influence.

***Watershed*:** The topographic drainage divide delimiting the land area from which surface runoff and groundwater flow contribute to a common receiving water body.

***Yield*:** The quantity of a resource (e.g. biomass, organic matter, or specific compounds) produced or harvested per unit area or time (e.g. by fishing); in benthic ecology, often refers to secondary production or harvestable biomass from a system.

A comprehensive list of terms and their definitions of ecological concepts used in coastal lagoons and transitional waters ecology can be found at EcoPortal, LifeWatch ERIC (Titocci et al, 2026).

Annex 2. Indicators

Table A1. Some indicators frequently used in the literature to diagnose the ecological integrity of biological communities in estuaries and coastal lagoons.

Indicator Name	Formula / Calculation Key	Description Key Applications	Bibliography
Based on Indicator Species			
AMBI (AZTI's Marine Biotic Index)	Classifies species into 5 Ecological Groups (EG I: sensitive, to EG V: tolerant). Index calculated from relative proportions of each EG	Based on species classification into ecological groups relative to tolerance. Organic enrichment, general pollution. Requires species-level ID. Borja et al. (2000; 2003) contain a broad species list (~3000 taxa), tested in many areas, and free software	Glemarec & Hily, 1981; Borja et al. 2000; 2003
BENTIX	Simplified version of AMBI using 3 groups instead of 5	Developed for Mediterranean waters	Simboura & Zenetos, 2002
Indicator Species Index (ISI)	Requires prior classification of species sensitivity values based on local data	Applies similar principles as AMBI (species classification). Developed for Norwegian/Swedish coastal waters	Rygg, 2002
Bellan's Pollution Index	Characterizes conditions by dominance of pollution-indicator vs. clean-water species. Based on polychaete communities	Organic pollution studies	Bellan, 1967; Bellan-Santini, 1980
Bellan-Santini's Index	Characterizes conditions by dominance of pollution-indicator vs. clean-water species based on amphipod communities	Organic pollution studies	Bellan-Santini, 1980
Benthic Response Index (BRI)	Uses ordination on a calibration set to quantify a pollution gradient based on species tolerance	Pollution gradient detection	Smith et al., 2001
Ecological Evaluation Index (EEI)	Based on coverage of two Ecological State Groups (ESG I: pristine, ESG II: degraded)	Macrophyte-based, for transitional/coastal waters	Orfanidis et al., 2001; 2003
Based on Ecological Strategies			

Indicator Name	Formula / Calculation Key	Description Key Applications	Bibliography
Macroalgal / Seagrass Indicators	Presence/absence/density of key species e.g. *Cystoseira*, *Posidonia oceanica* indicates good quality; *Ulva* indicates eutrophication	Submerged vegetation (macrophytes). Water quality	Niell & Pazo, 1978; Pergent et al., 1999; Pérez-Ruzafa, 2003
Infaunal Trophic Index (ITI)	ITI = (100 - %SS) + (%C), where %SS = % surface deposit feeders, %C = % carnivores	Assesses disturbance via benthic macrofauna feeding strategies	Word, 1979
Feeding Structure Index (FSI)	Ratio of % subsurface deposit feeders to total deposit feeders	Assesses disturbance via benthic macrofauna feeding strategies	Petrov & Shadrina, 1996
Polychaetes/Amphipods Ratio (P/A)	Ratio of polychaete to amphipod abundances	Indicator of organic/oil pollution	Gómez-Gesteira & Dauvin, 2000
Nematod/Copepod Ratio	Simple ratio of Nematod/Copepod abundances	Meiobenthic response to stress	Raffaelli & Mason, 1981
Meiobenthic Pollution Index	Benthic meiofauna (Polychaetes)		Losovskaya, 1983
r/K Strategies Index	Scores species based on biological trait analysis	Based on life-history traits of all benthic taxa	De Boer et al., 2001
Diversity Indices and network analyses			
Species Richness (S)	Simple count of species.	General disturbance indicator. Standard metric	Magurran, 2004
Margalef Index (d)	d = (S - 1) / ln(N) where S=species, N=individuals.	Detects changes in species number. Performed well in case studies	Margalef, 1958; Magurran, 2004
Shannon-Wiener Index (H')	$H' = -\Sigma(pi * \log_2 pi)$	Diversity incorporating richness and evenness. Information theory-based	Magurran, 2004
Pielou's Evenness (J')	$J' = H' / \log_2(S)$	Equitability of species abundances	Magurran, 2004
Simpson's Index (λ)	$\lambda = \Sigma(pi^2)$	Dominance-weighted diversity	Magurran, 2004
Berger-Parker Index (d)	d = N_max / N	Dominance of the most abundant species	Magurran, 2004

Indicator Name	Formula / Calculation Key	Description Key Applications	Bibliography
Average Taxonomic Distinctness (Δ+)	Average taxonomic-phylogenetic distance between species in a sample	Biodiversity measure independent of sampling effort	Warwick & Clarke, 1995; 1998
Total Taxonomic Distinctness (TTD/s+)	Abundance-weighted version of taxonomic distinctness	Similar to Δ+, but weighted by abundance	Warwick & Clarke, 1995; 1998
K-dominance Curves	Plots cumulative abundance vs. species rank.		Lambshead et al., 1983
Ascendency (A)	A=T*H Where T is the total system throughput and H is the average mutual information (a measure of flow organization derived from the logarithms of conditional probabilities between compartments)	Quantitative index in ecological network analysis that measures the organized complexity and overall performance of an ecosystem by quantifying both the magnitude and the constraint of trophic flows within its network structure	Ulanowicz, 1986
Biomass/Abundance			
SAB Curves	Comparison of species rank vs. abundance and biomass curves		Pearson & Rosenberg, 1978
ABC Method & W-Statistic	W = 2*(Area between cumulative biomass and abundance curves). W-statistic quantifies the difference. Ranges from -1 (polluted) to +1 (unpolluted)	Compares dominance patterns in biomass vs. abundance curves.	Warwick, 1986; Warwick & Clarke, 1994
Based on Thermodynamics			
Exergy Index (Ex)	Ex = Σ (β_i * B_i) where B_i is biomass, β_i is weighting factor based on genetic information	System-level indicator of biomass and genetic information	Jørgensen & Mejer, 1979; Nielsen, 1990; Jørgensen et al., 1991; Marques et al., 2003
Specific Exergy (Ex_sp)	Ex_sp = Ex / Total Biomass	Exergy per unit of biomass. Reflects the "quality" or complexity of the biomass	
Integrative Indices			

Indicator Name	Formula / Calculation Key	Description Key Applications	Bibliography
Index of Biotic Integrity (IBI)	Includes physico-chemistry, diversity, composition, trophic structure		Nelson, 1990
Benthic Index of Biotic Integrity (B-IBI)	Combines scores from several selected metrics (e.g. % sensitive species, trophic composition) into a single index	Integrates multiple metrics (diversity, composition, tolerance)	Weisberg et al., 1997; Van Dolah et al., 1999
Benthic Index of Environmental Condition	Includes various benthic community metrics		Engle et al., 1994; Macauley et al., 1999
Trophic Index (TRIX)	TRIX = ½ [log (Chl *a* + DIN + $P\text{-}PO_4^{3-}$) + 1.5], where DIN = $N\text{-}NO_3^-$ + $N\text{-}NO_2^-$ + $N\text{-}NH_4^+$	Integrates key water column parameters.	Vollenweider et al., 1998
Toxic Pollution (Biomarkers)			
Various Biomarker Methods	Not a single formula. Includes methods like: Metallothionein induction, Lysosomal stability, Imposex measurement, etc.	Specific to contaminant type (e.g. PAHs, heavy metals). Useful for specific toxic effects but hard to integrate into general quality status.	Working Group on Biological Effects of Contaminants (WGBEC; ICES, 2024)
Bioaccumulation Measurement	Measurement of pollutant concentration and toxic substances (e.g. heavy metals, PAHs) in tissues of accumulative species, including Molluscs (e.g. *Mytilus*, *Cerastoderma*), amphipods, polychaetes, algae.		Goldberg et al., 1978; Reish, 1993; Newman et al., 1991; Storelli et al., 2001
Ecologic Reference Index (ERI)	Specifically using mussels. Standardizes contaminant concentrations in organisms for comparison.		Baan & Vaan Buuren, 2002

Source: Salas et al. (2006).

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