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Planning hydrological restoration of coastal wetlands: Key model considerations and solutions

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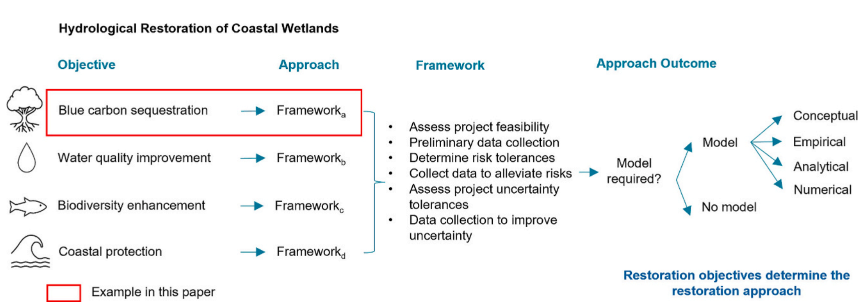
HIGHLIGHTS

- Hydrological restoration of coastal wetlands is emerging as a key tool for climate change mitigation and adaptation.
- Hydrological restoration has risks and uncertainties that can limit uptake and profitability while increasing planning costs.
- Data and models can help describe risks and reduce uncertainty.
- Developing low-cost data sources and easy-to-implement models may enhance the potential for hydrological restoration.

GRAPHICAL ABSTRACT



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ABSTRACT

The hydrological restoration of coastal wetlands is an emerging approach for mitigating and adapting to climate change and enhancing ecosystem services such as improved water quality and biodiversity. This paper synthesises current knowledge on selecting appropriate modelling approaches for hydrological restoration projects. The selection of a modelling approach is based on project-specific factors, such as costs, risks, and uncertainties, and aligns with the overall project objectives. We provide guidance on model selection, emphasising the use of simpler and less expensive modelling approaches when appropriate, and identifying situations when models may not be required for project managers to make informed decisions. This paper recognises and supports the widespread use of hydrological restoration in coastal wetlands by bridging the gap between hydrological science and restoration practices. It underscores the significance of project objectives, budget, and available data and

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offers decision-making frameworks, such as decision trees, to aid in matching modelling methods with specific project outcomes.

1. Introduction

The current global loss and degradation of coastal ecosystems is primarily attributed to anthropogenic actions (Dunic et al., 2021; Goldberg et al., 2020; Ostrowski et al., 2021). These ecosystems have high organic carbon storage in biomass and soil, called blue carbon (Macreadie et al., 2021). Blue carbon ecosystems and their soil organic carbon (SOC) stocks can be destabilised due to anthropogenic factors such as land-use change or climate-related factors such as sea-level rise (SLR) leading to CO_2 emissions that may result in the eventual loss of 25–100 % of the SOC in the system (Fourqurean et al., 2012; Sanderman et al., 2018; Pendleton et al., 2012). Such a loss of SOC would be equivalent to 3–19 % of the emissions attributable to global deforestation (Spivak et al., 2019). Because of the need to decrease CO_2 emissions, restore biodiversity and other ecosystem service benefits of coastal wetlands there has been increased interest in coastal wetland restoration globally (Waltham et al., 2021). This has been supported in part by the United Nations General Assembly recently declared "UN Decade on Ecological Restoration" for 2021–2030. However, many coastal wetland restorations have not been successful (Winterwerp et al., 2013; Lee et al., 2019; Bayraktarov et al., 2016). Low success in coastal wetland restoration may be due to poor project governance and site selection that does not sufficiently consider the biophysical and socioeconomic conditions required for restoration success (Primavera and Esteban, 2008; Lewis, 2005; Lovelock and Brown, 2019). Given the scale of restoration needs, and the increase in threats to coastal wetlands due to coastal development and climate change, there is an urgent need to learn from the past and rapidly identify strategies that improve restoration success.

Previous coastal wetland restoration projects have identified factors that are fundamental in designing a successful coastal wetland restoration project. Modification of the hydrological regime of restoration sites to replicate the hydrological regime of natural coastal wetlands is critical (Lewis et al., 2000). The desired natural reference states of coastal wetlands can include mosaics of different habitat types, such as seagrasses, mangroves, and tidal marshes of differing compositions and structures (see Supplementary A for examples). There is an increasing body of work on the importance of characterising inundation levels in ecosystem restoration (Rogers, 2021); however, the hydrological regime and its link to desired ecosystem types and services can be an overlooked component of restoration (Glamore et al., 2021; Gopal, 2013).

Hydrological modification of coastal floodplains intensified as agricultural, urban and industrial land-use increased throughout the 20th century, with the outcome being the loss of coastal wetlands and a decline in the structure and function of those remaining. Examples of hydrological modifications include widespread drainage of coastal landscapes for agriculture (Blann et al., 2009; Finlayson and Rea, 1999), impounding of floodplains for land uses like shrimp aquaculture, rice production or grazing (Blann et al., 2009; Pendleton et al., 2012), or construction of infrastructure that has resulted in changes in water flows and incidental degradation of wetlands (e.g. effect of highway construction on wetlands (Shuldiner and Cope, 1979)). These hydrologically modified coastal landscapes offer some of the largest opportunities for restoring coastal wetlands for their ecosystem services (Kroeger et al., 2017; Rowland et al., 2023). Some large government programs designed to restore the hydrological regime of coastal floodplains have partially achieved their objectives, such as the Napa Marsh and South Bay Salt Pond restorations in the United States of America and the Managed Realignment for Saltmarsh Restoration in Europe (Blackwell et al., 2004). Many additional opportunities to restore coastal wetlands remain that have yet to be identified, prioritised or actioned (Sasmito et al., 2023).

While early restoration works on coastal floodplains aimed to remediate acid sulphate soils, restore biodiversity or adapt to climate change, there is an emerging focus on restoring for climate change mitigation benefits, or blue carbon, as well as for co-benefits (e.g. biodiversity, climate change adaptation, water quality, flood storage and cultural heritage) (Hagger et al., 2022; Rogers et al., 2022). Restoration for blue carbon provides new impetus for restoring hydrologically modified coastal floodplains and their wetlands, but guidelines for hydrological restoration of coastal wetlands remain limited, often focused on particular regions, activities or risks (Supplementary B).

Here we synthesise current knowledge of the hydrological restoration of coastal wetlands. Decisions on the approaches used for hydrological restoration are complex and contingent on the overall objectives and project constraints, including tolerances of risks and uncertainties. This paper provides a framework, using the project objective of enhancing blue carbon capture as an example, that can be used by managers to evaluate simpler and less expensive modelling approaches, where appropriate, by providing guidance for matching modelling approaches with project objectives based on risk and tolerance of uncertainty. We describe the inherent complexity in restoration with the intention of establishing clear connections between hydrological science and modelling, and to support the accelerated global restoration of coastal wetlands through hydrological restoration.

2. Overview: hydrological restoration

Planning for coastal wetland restoration is a multi-step process, with steps undertaken sequentially, as described in detail by the Society for Ecological Restoration (Fig. 1). Here we focus on developing **restoration treatment prescriptions**. The approach for restoration follows stakeholder engagement, development of overall goals and specific objectives, identification of reference ecosystems and other components, that occur before project implementation. The resources available for restoration place constraints on all aspects of project planning and implementation but may be particularly important when considering hydrological restoration because of the complexity and risks of managing water flows. The description of the logistics of hydrological restoration, assessments of the resources required (human and financial), and cost-benefit analyses are critical components of restoration planning that may go through multiple phases of assessment in projects that use hydrological restoration, reflecting the complexities involved (Fejtek et al., 2014; Hansen et al., 2015; Jackson et al., 2022).

2.1. The importance of hydrological regimes for coastal wetlands

The hydrological regime influences both the physical (e.g., sediment movement and accumulation, water flows) and chemical (e.g., available nutrients, salinity and oxidation-reduction state) characteristics (often termed physicochemical) of coastal wetlands that have complex relationships with biota (Pérez-Ceballos et al., 2020). For example, hydrological characteristics influence the species present that are often differentially sensitive to inundation depth or variation in salinity, metabolic rates of biota, and propagule dispersal and plant and animal establishment success (Elster, 2000). For example, plant zonation varies with hydroperiod, which affects fish and wildlife use; and wading shorebirds utilise mudflats at specific water depths. Variations in hydrological characteristics can also enhance diversity. For example, transported sediments can interact with local geology to generate spatial heterogeneity, creating a variety of ecological niches (Mitsch and Gosselink, 2015). Variation in primary productivity, rates of organic matter decomposition, and organic matter export and burial are influenced by the hydrological regime thereby affecting organic matter accumulation and nutrient cycling (Mitsch and Gosselink, 2015). Ecosystem functions and processes directly or indirectly influenced by the hydrological regime influence the ecosystem services provided by coastal wetlands, such as habitat provision, improved water quality, carbon sequestration and coastal protection.

In coastal wetlands, hydrological regimes can be complex due to the mixing of hydrological sources (e.g. groundwater, surface water, rivers, ocean), each with its own spatial and temporal energy dynamics. The hydrological regime is influenced by both hydrological flows (from different sources) and the factors that moderate interactions between these flows which influence the environment (e.g. bathymetry, species composition, and sediments). The resulting complex behaviour of multiple water sources, their flows and moderating factors have been synthesised to create simplified indices such as hydroperiod (Box 1), which are useful in developing restoration designs (Lewis and Streever, 2000; Van Loon et al., 2016). Understanding and accurately characterising the hydrological regime in coastal wetland restoration sites is important for selecting appropriate sites and for implementing successful hydrological restoration (Balke and Friess, 2016).

2.2. Implementing hydrological restoration

The conversion of coastal wetlands to other land uses often involves the construction of tidal gates, levees, drains, or other water infrastructure which alters the natural hydroperiod. Therefore hydrological restoration of coastal wetlands often requires the reversal or modification of these works to restore connections to tides and achieve similar hydrological regimes as occurs in "reference" or unmodified sites (Haines, 2013). Reinstating tidal flows shifts the state of the system from one that is artificially maintained to support a particular land-use to one where natural hydrological processes are restored, allowing the system to develop in response to natural hydrological forcings over time (Fig. 2). With hydrological restoration, there is variation in the timing of recovery of different interacting processes affected by hydrological restoration (Fig. 2). For example, faunal use of newly inundated land may precede the establishment of mature plant communities and their carbon stocks (Rummell et al., 2023) (Fig. 2). Over time, sea-level rise, annual and interannual variation in sea level and other climatic factors (e.g. periodic intense storms, changes in rainfall patterns) can affect tidal inundation altering the hydroperiod, and this may cause changes in different processes and components of coastal wetlands (Fig. 2). For example, over time, changes may be observed in biological factors like species composition, plant cover and zonation (Fig. 2A) and in chemical characteristics of soils like acidity, salinity and oxygen levels (Fig. 2B). Additionally, the hydrology of sites may vary. For example, increases in magnitude of the tidal prism may alter channel width and flow velocities (Fig. 2C). Finally, processes like sediment accretion influence development of soils and site elevation and may alter sediment supply (Fig. 2D).

Reinstating natural hydrological conditions during restoration of modified coastal landscapes is challenging. For example, when breaching aquaculture pond walls and manually constructing tidal channels to restore mangroves, multiple corrections to the hydrology were made over time (Brown et al., 2014). This was because there was insufficient data to adequately inform the sizing of tidal channels during restoration. Affordable methods to measure variation in elevation over the project site were not available and there were no available methods to estimate changes in sedimentation as the hydrological regime was altered throughout the project (Brown et al., 2014).

Although there are examples of successful hydrological restoration projects, including the removal of tidal gates (Haines, 2013), dykes (Janousek et al., 2021), earth bund walls (Abbott et al., 2020) and other barriers to re-connect flow (Wolters et al., 2005), these examples are few compared to the larger number of other types of restoration projects reported in the literature (e.g. planting vegetation (Lee et al., 2019, Bayraktarov et al., 2016)). In fact, restoration of coastal wetlands is often considered easier when the tidal regime is unaltered. Earlier guidelines for restoring coastal wetlands suggest high-priority restoration sites should have an 'unimpeded or easily restorable hydrological regime' (McLeod and Salm, 2006). This is because coastal wetland

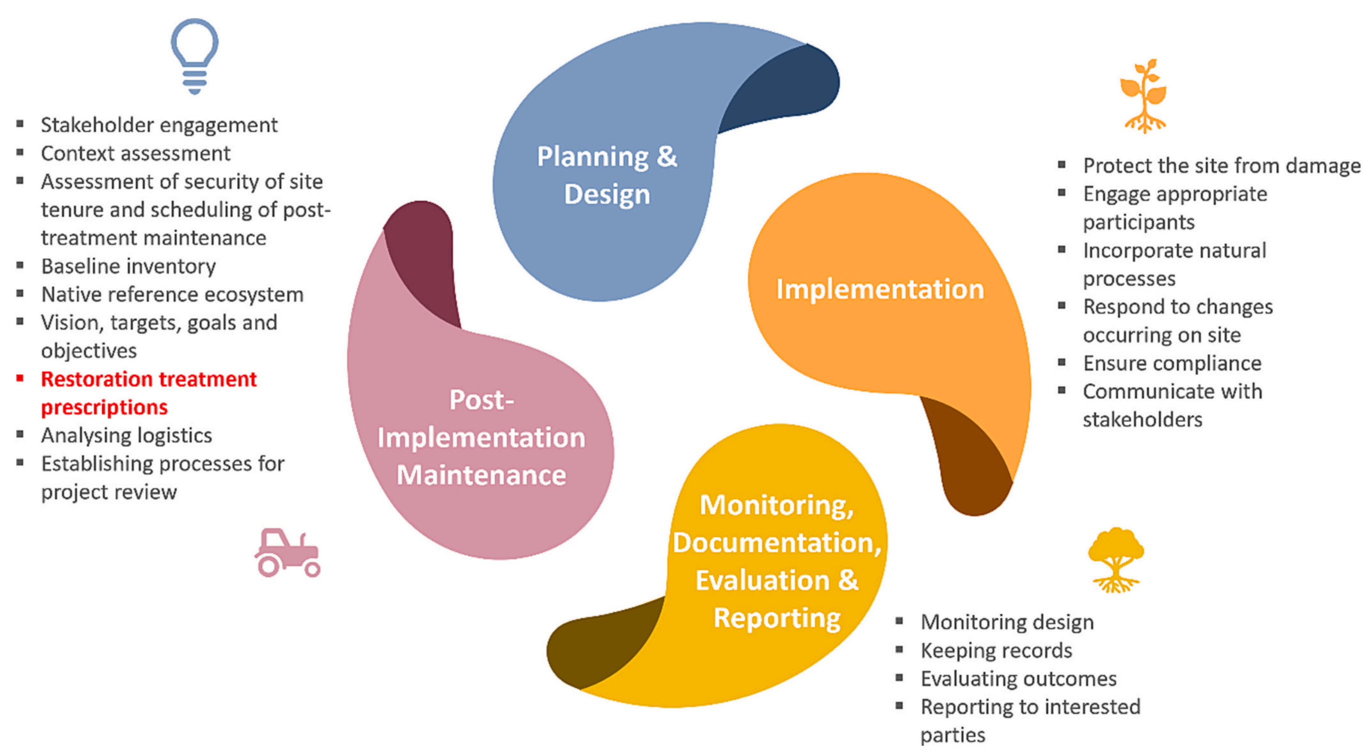

Fig. 1. The best practice process for restoration planning and implementation highlighting the gap this paper fills (in bold and red). (Adapted from Society for Ecological Restoration Australasia (2018).)

Box 1
Key terms and definitions.

Approach	The strategic process – For example, a project could use spontaneous regeneration of vegetation, facilitated (assisted), combined regeneration/reintroduction, or regrading – different approaches are matched to the level of site degradation and potential for recovery (Society for Ecological Restoration Australasia, 2018).
External hydrological sources	Sources of water from outside the system, including precipitation, run-off, and groundwater flows
Goal	Broad and overarching project aims brought to fruition through specific and measurable objectives
Hydrodynamics	The mechanical outcome of forces exerted by or acting on fluids
Hydroperiod	Frequency and duration of inundation
Model	Analytical – approximate solutions using mathematical equations that have a closed-form Conceptual – descriptive understanding of a system, e.g. the long-term geomorphological response to changes in forcing such as sea-level rise and other geomorphological models Ecogeomorphical – considers interactions between ecological and geomorphological processes Empirical – considers concepts derived from observations over various temporal and spatial scales Hydrological – simplification of a real-world system aiming to predict water resources' behaviour Numerical – approximate solutions to physical problems using mathematical equations Process-based – considers physical and ecological processes over temporal and spatial scales and is typically numerical Sedimentological – considers processes influencing sediment transport and characteristics
Objective	The intended outcome of a project, in this case, a restoration project, with respect to a reference site
Reference site/ecosystem	A non-degraded version of the ecosystem, complete with its flora, fauna (and other biotas), functions, processes, and successional states that would have existed on the restoration site had degradation, damage, or destruction not occurred (Society for Ecological Restoration Australasia, 2018).
Risk	Events with predictable probabilities of occurrence and outcomes can be estimated with some confidence (Infrastructure Australia, 2021).
System	The project area
Uncertainty	Events where probabilities of occurrence and magnitude are difficult to predict and outcomes are challenging to quantify (Infrastructure Australia, 2021).

ecosystems can often ‘self-repair’ if the natural tidal regime is not disrupted and propagules or seeds are available. Otherwise, propagules or seedlings can be planted (Lewis et al., 2000), or the natural supply can be augmented with seeds or propagules often with high levels of success (Lewis, 2005).

The challenging practicalities of hydrological restoration may inhibit the implementation of projects. The costs of hydrological restoration can be high, depending on the level of engineering required to restore water flows and mitigate for offsite flooding or erosion. Risks may be high, for example, given that wetlands act as flood storage by spreading flows across a greater area (Carter, 1996), the potential spread of water across landscapes during flooding may increase risks of flooding neighbouring properties. Data availability to model hydrological processes at appropriate scales can be limited, which may increase risks and uncertainty (Saunders et al., 2022). For example, modelling inundation at the scale of neighbouring land parcels requires fine scale elevation data that includes the depth of channel networks, but this is often unavailable (Saunders et al., 2022). While large-scale restoration targets have been proposed through nesting smaller projects within a landscape (Lovelock et al., 2022b) this gives rise to several hydrological challenges. Several small but hydrologically distinct projects may need to be assessed individually using fine scale data. In addition, water flows between them may interact requiring assessment at larger catchment scales (Saunders et al., 2022), which may not achieve anticipated economies of scale. Achieving ambitious wetland restoration targets will need to consider the practical challenges of implementation, particularly cost and our ability to plan using models of hydrological flows at a range of scales.

3. Approaches for hydrological restoration

Approaches to hydrological restoration of coastal wetlands can vary in complexity and cost depending on the restoration goals and the level of risk and uncertainty that are acceptable within a project.

3.1. Goals for hydrological restoration of coastal wetlands

Setting goals for restoration (identifying what project proponents want to achieve) is a fundamental principle of the design and implementation of all restoration projects (Society for Ecological Restoration Australasia, 2018). Clear goals for hydrological restoration projects influence project design and implementation (the approach) because different designs and management approaches achieve different outcomes with varying costs, risks, and uncertainty levels. For example, blue carbon projects which aim to optimise carbon abatement require regular tidal inundation (Negandhi et al., 2019) and may focus on establishing mangroves, which may be achieved in some drained landscapes by lowering tidal gates (Kroeger et al., 2017). Optimising restoration to improve water quality requires the retention of nutrient-rich waters in the wetland for long periods, which may be achieved by extending hydroperiods (Sarker et al., 2020), such as by raising tidal gates or plugging drains. Biodiversity goals such as providing habitat for birds typically require a mosaic of habitats (e.g. roosting, feeding, breeding for different species), including permanently flooded habitats (Kačergytė et al., 2021), which has previously been achieved by raising tidal gates to elevate water levels and through targeted levee construction (Glamore et al., 2021).

Clear objectives for restoration projects also enable the design of appropriate indicators for assessment of project progress and success (performance monitoring). For example, a project aiming to improve flood protection as an ecosystem service may monitor water levels, wave climate, sediment transport processes, surface accretion and erosion (Gijsman et al., 2021). In contrast, a project aiming to increase nesting habitat for shorebirds may monitor bird abundance and plant community composition (Ocean Studies Board, 2017).

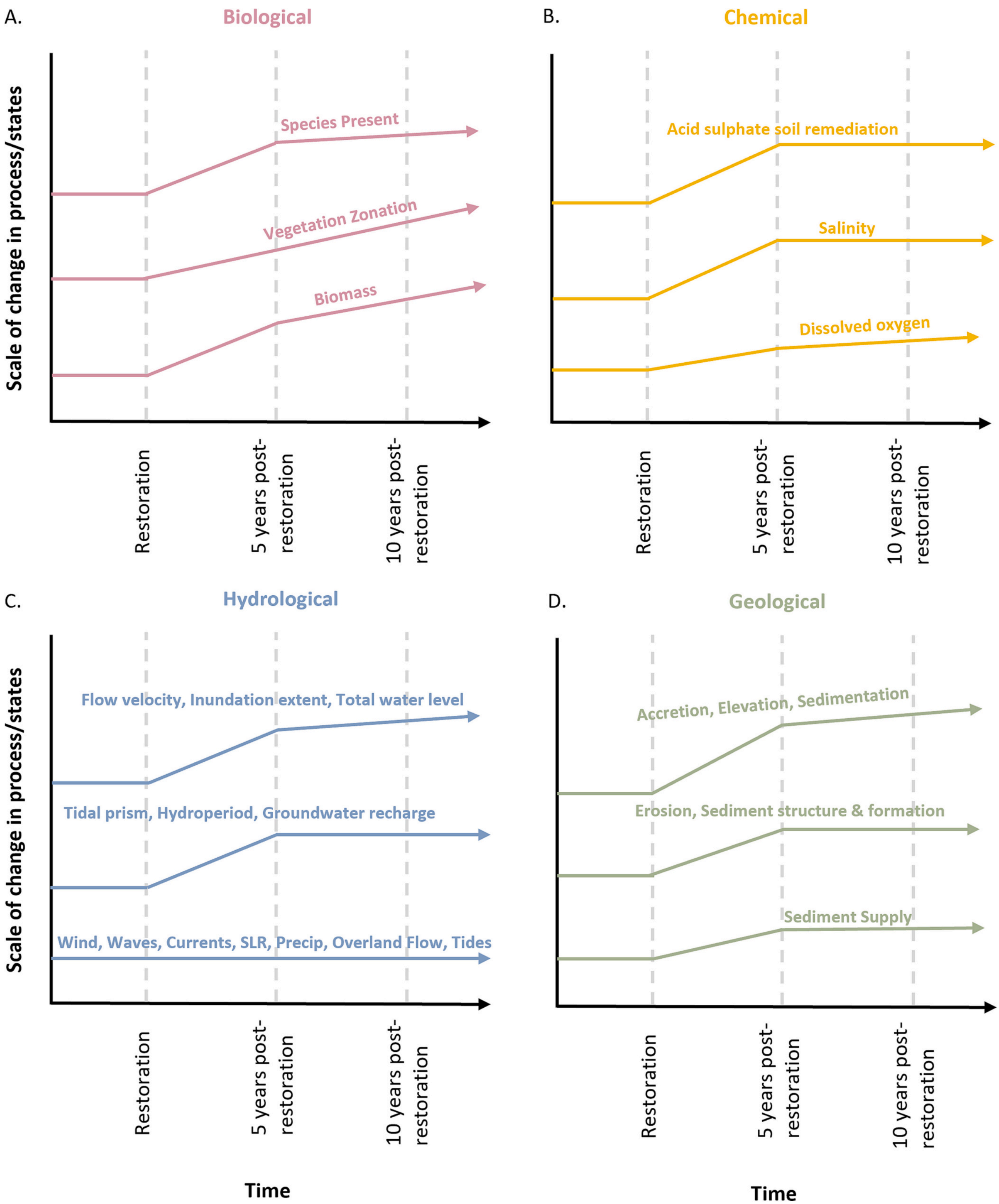

Fig. 2. Hydrological restoration shifts the state of the system from one that is artificially maintained to be suitable for certain land-uses (removed from tidal influences) to that exposed to natural hydrological processes, which affects coastal wetland processes and the long-term development of the system. Hypothetical changes in biological (A), chemical (B), hydrological (C), and geological (D) processes and states after the hydrological restoration of coastal wetlands.

3.2. Risks and uncertainty

Risks to projects are known adverse consequences of project activities that can often be quantified (as the probability of occurrence), while uncertainties in the context of engineering are factors that are difficult to quantify (Infrastructure Australia, 2021) (see definitions in Box 1). In risk analysis, outcomes of activities may be unknown, but the underlying probability distribution is often discernible. For example, the intensity and timing of future storms are unknown but the water levels for various storm events can be projected or estimated based on past events. Conversely, uncertainty arises when outcomes and their associated probability distributions are unknown. For example, water levels after removal of a specific tidal restriction are not known and data collection is required to reduce uncertainty in the outcomes. In cases where there is uncertainty in outcomes of hydrological restoration, decision-makers can evaluate their preferences for restoration actions in light of the possible chance distributions of outcomes (De Groot and Thurik, 2018) which may be elicited using expert opinion or other probabilistic models.

Risks arising from hydrological restoration projects include flooding of adjacent agricultural land, increased damage to adjacent habitats or species, the release of pollutants to waterways, damage to infrastructure, salinisation of coastal groundwater aquifers, changing soil and porewater chemistry, and severe erosion due to increasing velocity and volume of tidal currents. Risks associated with hydrological restoration are also those of project failure and can be unique to the project objectives (Table 1). For example, there is the risk that the restored ecosystem does not function adequately or provide the ecosystems services predicted, or it remains in a damaged state.

Uncertainties are factors that are not well understood and have limited data availability, yet contribute to uncertainty in predicting projecting outcomes. For example, future rainfall patterns with climate change may be an uncertainty in many regions. Data collection can be a solution to reducing uncertainty. However, an alternative approach to collecting more data to address data deficiencies is to use scenario analyses (e.g. multiple socio-economic pathways are explored in the Intergovernmental Panel on Climate Change (IPCC) reports (Riahi et al., 2017)) and to manage projects adaptively (Hamilton et al., 2013). Uncertainty limits confidence in projecting outcomes of restoration actions and is unique and inherent to all restoration objectives. The process of restoration of marine and coastal ecosystems and rivers has highlighted limitations in scientific and other information associated with these systems, limitations in analytical methods and tools, high levels of complexities in ecosystems, and the need to use expert judgements at all stages of restoration problem identification, analysis and implementation of solutions (Darby and Sear, 2008; Saunders et al., 2022). Ultimately, there are often high levels of uncertainty in input data and the tools used to process that data.

Lack of data or datasets with limited temporal scale and resolution and spatial scale and resolution, is a common problem for coastal wetland restoration projects that impose limits on the characterisation of risks and uncertainty. For example, water levels and inundation extent are often required to design hydrological restoration projects, hence datasets for nearshore and intertidal bathymetry and habitat structure and density are required (Beck et al., 2016). However, these datasets are often difficult to obtain because of high costs or because of the sensitive nature of the data (e.g. national security). Additionally, models are often used to describe outcomes of hydrological restoration and support decision making, but empirical datasets for validation can be scarce; reducing confidence in models used in making decisions about restoration projects or policies (Xie et al., 2019).

One of the largest sources of uncertainty for hydrological restoration is the limited availability of high-resolution elevation data for modelling hydroperiod and flooding. Accurate bathymetric and topography data are crucial for spatially relevant coastal modelling and monitoring (Pacheco et al., 2015). This is particularly the case as small errors in elevation data can have significant consequences for projections of vegetation establishment and ecosystem service provision. In some cases, in lieu of local data, national elevation models have been used to plan or prioritise coastal wetland restoration sites (Rogers et al., 2022) and global elevation data has been used to indicate potential for coastal wetland landward migration (Schuerch et al., 2018). However, global digital elevation models (DEM) such as those generated from the Shuttle Radar Topography Mission (SRTM) (freely available through the U.S. National Aeronautics and Space Administration (NASA)) have a nominal vertical accuracy of 3.6–16 m (Schumann et al., 2008). More recently another global elevation data set (WorldDEM Neo), with a 2.5 m absolute accuracy has become available, but the accuracy remains low compared to the common vertical accuracy of airborne Light Detection and Ranging (LiDAR) of 0.15 m (Schumann et al., 2008)., and the very high accuracy now being achieved from LiDAR fixed to remotely piloted aircrafts (i.e. drones). The coarse vertical accuracy of global elevation data and models derived from this data makes them less suitable for wetland restoration modelling because the elevation envelopes of many coastal wetland species are within the Residual Mean Square Error (RMSE) of the data products. For example, the species elevation ranges for *Nypa fruticans* are 0.407 and 0.431 m and *Sonneratia alba* are 0.212 and 1.083 m in Singapore (Leong et al., 2018) and Micronesia (Ellison et al., 2022) respectively. Validating and calibrating STRM-derived DEMs against site-specific data is a pathway to improving the accuracy of elevation models. However, even with high-resolution elevation data (e.g. from Light detecting radar, LiDAR) inadequate data processing can lead to inaccuracies. The presence of vegetation is a source of error (even with "bare earth" corrected LiDAR) (Simpson et al., 2017) and there are limited LiDAR datasets that are flown at low tide. Further processing of point clouds can improve the accuracy of the resulting

Table 1
Risks associated with hydrological restoration of coastal wetlands for different restoration objectives.

Project	Objective	Risks	Consequence	Reference
South Bay Salt Pond Restoration Project	Restore a mix of wetland habitats, provide wildlife-oriented recreation, flood management	Risks of reducing shorebird populations from altering salt pond habitat. Risks of flooding to adjacent vulnerable land-use and infrastructure (via construction of flood protection measures)	Reduced shorebird populations, mud flat habitat loss. Damage to adjacent infrastructure and land.	Rowan et al. (2011)
Caernarvon, Naomi and West Point a la Hache diversions	Reduce the rates of tidal marsh degradation	Risks of project failure due to increased nutrients from run-off that prevents vegetation establishment.	Wetland areas fail to establish, remain degraded and are more vulnerable to hurricanes.	Mascarelli (2011)
Eastern Dundas Tablelands, Victoria	Reduce acid sulphate soils via re-wetting	Risk of acid sulphate leakage due to excavations.	The use of trenches in rewetting enhances acid sulphate soils.	Gardner et al. (2018)
Surat Thani, Thailand	Revert abandoned shrimp pond to a mangrove forest	Risk of inappropriate hydrological regime. Risk of salinisation of adjacent vulnerable land-use.	Planted mangrove seedlings failed to establish as tidal hydrology was not properly restored. Decreased biodiversity and crop production in neighbouring areas.	Perillo et al. (2018)

Table 2
Overview of tools used in hydrological restoration projects to balance risks, uncertainties, and costs.

Tool	Overview	Use and examples
Risk assessment frameworks	Risk assessments are often used to guide restoration decisions and identify risks and uncertainties, often in conjunction with a cost-benefit analysis to compare each option's financial benefits and disadvantages. Risk assessment types include comparative risk assessment and multi-criteria decision analysis.	Guidelines exist for their application to restoration projects, such as the IUCN guidelines for 'using ecosystem risk assessment science for ecosystem restoration' (Valderrábano et al., 2021). Evaluation of the risks of restoring tidal flushing guided the proposed restoration measures for the Tomago Wetland Restoration (Glamore et al., 2021).
Cost-benefit analyses	Cost-benefit analysis (CBA) measures the benefits of an action/decision compared to the associated costs.	CBA has been used to weigh the costs and benefits of a mangrove restoration project compared with the 'do nothing' alternative, given the risks and uncertainties associated with climate change hazards (Agaton and Collera, 2022).
Value of information analyses	Evaluation of the cost of this data against its value towards reducing risk and uncertainty can be achieved using a Value of Information (VOI) analysis. In decision theory, the value of information is the difference between the expected value of an action before and after introducing new information (Raiffa and Schlaifer, 1961).	VOI has been used to investigate the cost savings obtained by including life-history information into a river connectivity restoration plan in Alaska (Sethi et al., 2017), investigate the worth of hydraulic conductivity data for the optimal restoration of an over-exploited aquifer in Greece (Sidiropoulos and Mylopoulos, 2015) and identify that meadow restoration was the best strategy in the face of uncertainty to manage endangered populations of whooping cranes in North America (Runge et al., 2011).
Modern portfolio theory	Modern portfolio theory (MPT) is an approach that allows minimisation of risk while providing an expected value of future returns by optimised diversification of investments (Markowitz, 1952, Markowitz, 1959, Elton and Gruber, 1997).	Given the various uncertainties in outcomes of ecological restoration, climate projections, population growth, species responses to climate change drivers, and even changes in ecosystem service valuation, among others, MPT can provide insight into allocation of investment in restoration and conservation efforts (Ando et al., 2018, Ando and Mallory, 2012, DuFour et al., 2015, Crowe and Parker, 2008, Carvalho et al., 2011) including application to coastal systems (Eaton et al., 2019, Runting et al., 2018, Vinent et al., 2019, Popov et al., 2022).

DEMs and achieve a more precise representation of site elevation (Agüera-Vega et al., 2020) thereby supporting the evaluation of hydroperiod for hydrological restoration.

3.3. Tools to balance risks and uncertainties, and costs

Several tools are available for hydrological restoration projects that can be used to identify, balance, and make decisions that compromise between risks, uncertainties, and costs (Table 2). Uncertainty is often assessed using analyses of scenarios which assesses the outcomes for different futures where the proposed future determines the modelling detail required, or through real options analyses that evaluate investment and decisions making in future scenarios taking into account the effect of alternative strategies (Infrastructure Australia, 2021). Some commonly used tools to evaluate risks include risk assessment frameworks. Cost-benefit analyses are used to compare the range of potential costs and benefits of different projects or activities. Decision theory tools such as the value of information analysis can be used to evaluate the costs and benefits of reducing uncertainty, and emerging methods such as Modern Portfolio Theory are used to evaluate the benefits of investing in multiple different projects (Table 2). Irrespective of methods used, hydrological restoration projects span a range of risks, uncertainty, and cost that are described in Box 2.

3.4. Models as tools for planning hydrological restoration

A variety of models are used in hydrological restoration and are particularly important for characterising outcomes and identifying risks and uncertainty (Box 2). Conceptualising the behaviour of hydrological processes at the spatial and temporal scales relevant to hydrological restoration projects is the most important step for planning hydrological inventions in restoration projects (for example, Tomago Wetland Restoration in Box 2).

Decisions on restoration approaches reached through best practice involve an interactive process among stakeholders and experts to create conceptual models. This is often effectively achieved in "workshop" settings where the outcomes of the workshops are descriptions of the dominant processes at a site, those influencing emergent patterns, mechanisms underpinning these processes, and the interactions between processes operating at large spatial or temporal scales. Often processes at smaller spatial and temporal scales, second- (and third-) order processes relevant to the goals of a restoration project are identified in this process and so more detailed data collection and monitoring can be planned.

Once a system has been conceptualised, and the risks and uncertainties identified, often more detailed modelling tools are used to further assess different restoration scenarios, including those that encompass the effects of sea-level rise or other uncertainties. Available data can be collated. If there are limited local data but global or region data are available, then options for downscaled estimates from available global or regional datasets may exist if expertise to do so is available (Section 3.2), else the expertise of those involved may contribute to decision-making.

If more complex models are feasible or necessary, then different numerical, analytical, empirical, and hybrid model types have been used in hydrological restoration projects (Fig. 3). Different types of models have varying levels of complexity and suitability for modelling hydrological scenarios for restoration (Supplementary C). Below we described some of the characteristics of these models and where they may be used most appropriately.

The financial cost for software licenses differs across models (see Supplementary D). Meanwhile, the expenses associated with model development by a specialist are contingent upon factors such as data availability, the level of risk requiring mitigation, the intricacies of calibration and verification processes, and the proficiency of the modeller. For any specific site, associated datasets and modelling goals, the labour hours needed to construct the model may vary significantly between a seasoned modeller and a less experienced modeller.

3.4.1. Conceptual models

Conceptual models can be useful tools for hydrological wetland restoration projects, particularly during the planning and conceptualisation stages. This simplified representation of a complex system can help project designers and stakeholders understand the key elements and hydrological processes of wetland function and how they interact. This approach highlights challenges and opportunities for wetland restoration and informs the development of a restoration plan tailored to the site's specific conditions and restoration goals. One of the major challenges of applying conceptual models is that hydrological processes important for wetland function can be complex and highly interconnected; for example, linking lagoon hydrology and inlet morphology (Behrens et al., 2015). Developing a conceptual model that accurately

Box 2
Examples of restoration at different scales, differing levels of risk and uncertainty, and the activities used to reduce them.

The actions used to reduce risk and uncertainty for hydrological restoration of coastal wetlands depends on the scale of the project as well as the projects tolerance of risks and uncertainty. Project planning processes often reflect the levels of risk and uncertainty. Additionally, some project sites may be exposed to higher levels of climate risks (e.g. marine heatwaves, intense storms, rainfall variation) and other uncertainties (e.g. management of catchment infrastructure) than others. Therefore, the goals, project scale, site factors, risks, and uncertainties of coastal wetland restoration projects strongly influence the selection of the actions used for hydrological restoration, including the types of modelling and analyses used.

Small scale, low risk, medium uncertainty: Bulimba Creek Wetland Restoration, Queensland, Australia

A motorway alignment was designed through a degraded wetland area at Bulimba Creek, Queensland. A small-scale hydrological restoration (15 ha) of a degraded wetland was used to increase the sustainable outcomes of the infrastructure build. The project involved restoring tidal flows to one area of the Bulimba Oxbow wetlands by removing a bund and constructing two causeways (weirs) to allow tidal ingress. The outcome of the restoration works included saltmarsh and mangrove recovery and rehabilitation of the waterway and fish habitat (Green, 2009).

Risk: The restoration site was part of infrastructure development and was engineered to withstand major floods, so the establishment of a coastal wetland posed little risk to the infrastructure project. There were risks that wetland plants would not grow.

Uncertainty: Baseline data were collected on hydraulic flows within the region, but no specific bathymetry surveys were conducted at the hydrological restoration site.

Actions to reduce risk and uncertainty: Regional hydraulic modelling was conducted to predict upstream flooding levels for the road over a variety of storm conditions. To reduce the risk of vegetation not surviving, weeds were removed from the site and minor design changes to drains and fence locations were made throughout the construction to allow the retention of individual trees. The entire site used laser-controlled levelling to design the elevation of the causeway to have tidal inundation that would be suitable for wetland plant growth.

Large scale, high risks, high uncertainty: Tidal Marsh Restoration in three Salt Ponds – California, USA

Medium-scale restoration (1175 ha) via dyke breaching for Ponds 3, 4, and 5 of the Napa-Sonoma Marshes Wildlife Area in northern San Francisco Bay was undertaken in 2006 (Brand et al., 2012). The goals of the restoration were to restore areas of formerly subsided, diked salt ponds to a vegetated marsh consisting of *Spartina foliosa* that would eventually transition to a higher marsh. The project aimed to enhance biodiversity and assist in the recovery of threatened and endangered species.

Risk: The restoration site had three major risks: 1) The risk of increases in upstream flooding; 2) the risk of not restoring high marsh habitat (large areas could remain mudflat or low marsh due to low rates of sedimentation). The creation of high marsh was a project objective and important to achieve for some of the project funding. The restoration approach used excavated channels in some areas to enhance water and sediment delivery, as well as constructing low berms to serve as sheltering wave breaks. 3) The risk of short-term environmental damage (and agency fines) from rapid release of concentrated salts from the former salt ponds. The need to mitigate this risk led to hydrodynamic modelling that resulted in use of temporary outlet structures to release salt and promote controlled mixing of water prior to full restoration.

Uncertainty: It was unknown whether rates of sedimentation would be sufficient to raise the mudflats to elevations suitable for tidal marsh plant growth. It was unknown how the project would affect remnant marsh at the site.

Actions to reduce risk and uncertainty: Research to investigate how to enhance sedimentation and to evaluate the area available in the ponds that would support *S. foliosa* colonisation was done as part of the "adaptive management" strategy. MIKE 21 modelling was used to inform management of initial salinity releases and to design measures to avoid potential flood impacts. Hybrid and geomorphic modelling were conducted to identify the potential loss of existing remnant marsh through channel scour. A geomorphic mass balance approach was used to understand the potential for mudflat erosion, and a 0D marsh sedimentation analysis was conducted to determine the rate of tidal marsh habitat development. Modelling helped set expectations for when the most saline ponds could be restored. The project cost, including land acquisition, technical design and construction, was around US$6 million.

Medium scale, medium risk, medium uncertainty: Eco-hydrology as a driver for tidal wetland restoration – New South Wales, Australia

This project focused on the tidal restoration of the Tomago Wetland in eastern Australia and aimed to recreate coastal saltmarsh habitat. This was accomplished by creating inundation depths and hydroperiods across the 410 ha site that were similar to natural reference sites.

Risk: The acid release from acid sulphate soils was a risk to waterways and biota. Additionally, there was a risk of inundation of neighbouring private properties with opening of tidal gates.

Uncertainty: There was limited available data on water levels, channel discharge, and topographic and bathymetric data (Glamore et al., 2021).

Actions to reduce risk and uncertainty: Data was collected for each of the uncertainties above. MIKE FLOOD was used to simulate the 1D/2D channel and overbank flows, MIKE 11 was used to simulate 1D flows in the channels, including through tidal restrictions and culverts, and MIKE 21 was used to simulate the 2D overland flow. The restoration was undertaken in three stages over 8 years. The restoration strategy and scenario testing were designed to encompass all restoration stages to ensure an adequate hydroperiod was created to support coastal wetland plant communities (Glamore et al., 2021).

Small scale, Low risk, low uncertainty: Restoration of disused aquaculture ponds in Indonesia

The project restored several disused shrimp ponds in Indonesia that experienced disease and poor water quality (Stevenson et al., 1999).

Risk: Low risks because ponds were in an unproductive landscape. Without restoration of vegetation the risk of erosion risks be enhanced (Stevenson et al., 1999).

Uncertainty: There was limited data availability for tidal flows or elevation.

Actions to reduce risk and uncertainty: Conceptual model development. Breaching of pond walls followed by monitoring of outcomes and adaptive management. Adaptive management was used to modify excavations of pond walls and channels.

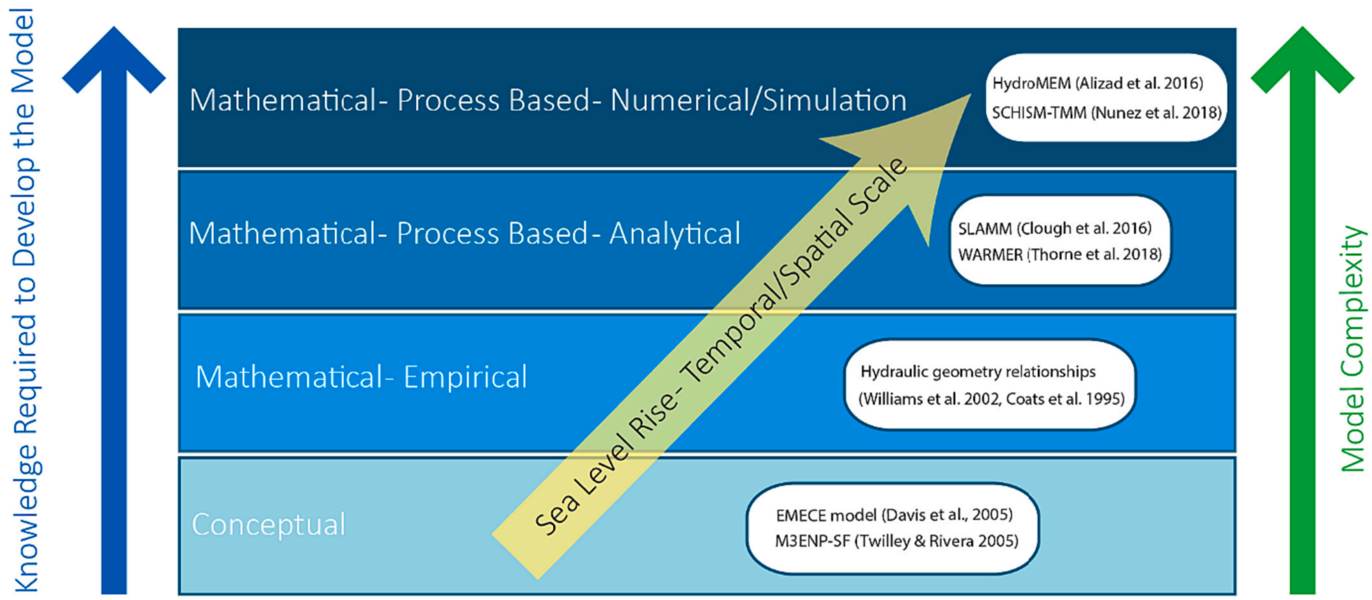

Fig. 3. Examples of model types available for hydrological restoration (arrows denote an increase in magnitude) showing the increasing performance for including SLR and finer temporal and spatial scales with examples of models (white boxes).

represents these processes may require significant data integration from multiple sources including rainfall-runoff data, hydrodynamics, groundwater, elevation, accretion models and vegetation data. At the simplest level, some restoration projects use the conceptual model augmented by an elevation model to project coastal wetland development (Cahoon et al., 2019). Attempts have been made to integrate hydrodynamic and ecogeomorphological modelling within conceptual models but this involved gross simplification and 10-year time steps due to computational limitations. In contrast, it is possible that full integration of hydrodynamic and ecogeomorphological models within conceptual models with smaller time steps (e.g. daily) could be achieved with cloud computing (Kurtz et al., 2017).

3.4.2. Empirical models

Empirical models are based on data from previous studies and observations, and are used in many cases to predict a restored wetland's performance based on specific design parameters (e.g. predicting the tidal prism based on wetland creek geometry (Coats et al., 1995, Williams et al., 2002)). These models are useful for making projections based on the system's behaviour observed in the past or in reference sites. Applying empirical models that rely on statistical relationships between variables can be helpful in wetland restoration projects because they are based on data and observations that reflect the specific conditions of a particular site. However, they may not be transferable to other sites, and their predictive capabilities are limited to the specific conditions of the data used in the model.

3.4.3. Process-based models: analytical models

Process-based analytical models are used to simulate the physical, chemical, and biological processes in a system. These models are designed to provide a quantitative understanding of the interactions between different processes and to help predict the responses to changes in environmental conditions and/or management practices. Ecogeomorphic models are analytical models that integrate ecological and geomorphic processes to simulate the dynamics of coastal wetland ecosystems and thus can be modified for use to explore outcomes of hydrological restoration.

Most of the ecogeomorphological models are less computationally complex than hydrodynamic models, and they simulate the primary processes contributing to the elevation adjustment of coastal wetland surfaces, including factors like accretion, subsidence, and the effect of sea-level rise. Given that restoration projects are long-term, they can consider the influence of sea-level rise on hydrological processes. However, future projections of ecosystem elevations and plant communities are challenging to constrain, given uncertainties in the drivers, the accuracy of parameterisation data, and non-linear feedbacks inherent to coastal wetland ecosystems (Koch et al., 2009). Modellers have used various solutions to simulate the impacts of sea-level rise. The simplest models increase the water level and surface elevation at set rates through time (e.g. SLAMM (Clough et al., 2016)).

Ecogeomorphic models are typically conceptualised as non-spatially dependent (0D) models that include the addition of mineral and organic materials, losses or volume associated with diagenesis, organic matter decomposition, and auto-compaction arising from the consolidation of sediments. These 0D models can be parameterised in three dimensions to simulate the development of coastal wetlands (e.g., MEM (Alizad et al., 2016) and WARMER (Thorne et al., 2018)), although their application typically relies on simplified assumptions of inundation time or frequency to drive ecosystem responses. The simplification of hydrodynamic and geomorphological processes improves the capacity for geomorphological models to include variation in tidal plains or hydrodynamics that vary over spatial and temporal scales but does not capture the full complexity of hydrodynamic and sediment transport processes and feedbacks (Mogensen and Rogers, 2018).

3.4.4. Process-based models: numerical/simulation models

Numerical models include any mathematical model that uses numerical methods to solve equations or simulate physical processes. Numerical models can simulate a wide range of physical and chemical processes, such as sediment transport, groundwater flow, and ecosystem dynamics that can be used to predict future water levels and environmental changes caused by the removal of tidal restriction devices. These models use numerical methods to discretise the equations that describe the physical processes and then solve the equations using iterative algorithms (providing an approximate solution).

Several numerical models have been proposed to describe coastal wetland persistence under different scenarios of SLR (e.g., Temmerman et al. (2003), van Proosdij et al. (2006), D'Alpaos et al. (2007), Kirwan et al. (2016), Mariotti (2016)) which is particularly important when predicting vegetation changes based on changing water levels. These models quantify the evolution of tidal marshes under different physical and ecological drivers. However, in many of these models, the sediment transport dynamics are highly simplified, representing only the starting point for the system rather than the spatial distribution and evolution over time.

Hydrodynamic models are numerical models that simulate fluid flow and transport processes, such as the movement of water and sediments in rivers, estuaries, and coastal regions. Hydrodynamic models typically use the Navier-Stokes equations, a set of partial differential equations that describe fluid flow that simulate the physical processes in a system (Teng et al., 2017). These models are used to study the hydrodynamics of natural and engineered wetland systems and to predict the effects of environmental changes, such as changes in river flows or SLR on these flow regimes. Other models use equations such as the Exner equations that are particularly relevant for sediment supply (Deng et al., 2017).

Hydrodynamic models can be valuable tools in wetland restoration projects, as they can simulate and predict factors and processes involved in wetland habitat development, such as water movement, flow velocities, sediment transport, accretion, and erosion. For instance, these kinds of models can predict water flow through a restored wetland, allowing project designers to optimise water distribution to improve ecosystem function (Karim et al., 2021) and simulate sediment transport within a wetland, which is essential for restoring natural sedimentation patterns, maintaining healthy wetland habitats and increasing carbon storage (Pérez-Ceballos et al., 2020). Moreover, highly resolved outputs from hydrodynamic models can help optimise restoration design decisions, including the placement of structures and the design of water flow paths, as well as to predict erosion patterns and develop strategies to prevent soil erosion, which is crucial for long-term stability of these ecosystems (Alizad et al., 2016; Zhang et al., 2020; Nunez et al., 2020).

Hydrodynamic models are on the computationally intensive end of the spectrum of available tools. It is challenging to use many existing hydrodynamic models (e.g. SCHISM-TMM (Nunez et al., 2021), HydroMEM (Alizad et al., 2016), Delft3D (Brew and Williams, 2010)) or to compare potential management scenarios within them due to the intensive computational requirements required to capture the time scales of interacting non-linear dynamics and arising ecogeomorphic feedbacks. Balancing complexity and computational efficiency can be difficult, but frameworks can help efficiently align model design and complexity with the restoration objective (Larsen et al., 2016). For example, Larsen et al. (2016) developed a decision-tree that can be used to identify trade-offs between the level of detail required by a model and computational resource requirements.

Hydrodynamic models typically have simplified treatment of vegetation responses and vegetation community transitions and generally lack consideration of below-ground biological processes, although they could feasibly be parameterised to account for these factors (Hui et al., 2022). They also simplify sediment transport, with limited capacity to account for anything more than suspended sediment concentrations (i.e. no information about grain size or mass etc.) (Papanicolaou et al., 2008). Improving these models by increased parameterisation and computational resourcing may be costly. For smaller site-specific areas (local catchment scale), HydroMEM and other hydrodynamic models, are well suited to describing hydrodynamics and dependent processes in restoration sites.

3.4.5. Models for understanding the impacts of sea-level rise

Uncertainty in elevation data and changes in soil surface elevation over time are key limitations for many models, but these parameters are important for projecting the impacts of SLR on coastal wetland restoration sites. While many ecogeomorphological models include SLR by using relationships between inundation and the depth below the highest astronomical tide (or other tidal plane), most hydrodynamic coastal wetland models implement boundary conditions that indirectly or directly include SLR along with sediment fluxes through open water boundaries such as the mouth of an estuary (Kumbier et al., 2022). These models are dependent upon accurate parameterisation of the 3-dimensional space defined by the boundary conditions, which is a key limiting factor in low gradient (low slope) landscapes that are typical of coastal wetlands (Larsen, 2019) where accuracy and precision errors in elevation and bathymetric data can be orders of magnitude greater than the hydrodynamic or geomorphological change that a model simulates. Therefore, the development of high-resolution DEMs can be a solution to reduce uncertainty in the models.

The limitations and uncertainties surrounding sediment supply and deposition, vegetation cover and structure, and other key factors in estuarine systems necessitate a focused effort to address knowledge gaps for use within complex models. While projected rates of relative SLR are broadly available and dependent on climate predictions (e.g., Garner et al. (2022)), sediment supply to these systems is generally poorly known and in some areas are difficult to predict due to variable weather patterns. An example is California's Mediterranean climate, where rainfall is sporadic, and most estuaries have hardened shorelines that may prevent sediment transport (Hanak and Moreno, 2012). Similarly, vegetation cover (Buffington et al., 2016; Holmquist et al., 2021), used to initiate starting conditions for models may not be up to date or operate dynamically to incorporate geomorphological or vegetation cover changes occurring within a system over time. Where parameterisation is limited, efforts can be directed to addressing knowledge gaps that would address key uncertainties. While long-term water levels and existing topographic and bathymetric surveys and surveys of vegetation cover and condition can improve the performance of models, fluxes in these systems can be poorly known. When all input factors to models are not well known the uncertainty in projections can be analysed using a range of values that may have a known distribution (e.g., use Monte Carlo simulations (Sun and Zhu, 2019)). Alternately, sensitivity analysis can be conducted to understand how influential variables are to the modelled outcomes which can help guide the development of qualitative decision-making frameworks (Cai et al., 2018; Sun and Zhu, 2019; Zhang and Shen, 2021).

4. Guide to hydrological model selection

Given the wide range of models (Section 3.3) and methodologies (Supplementary C) available, coastal wetland restoration practitioners need to be able to identify the best approaches for their project. Modelling products vary in cost and function, and the model most suited to a project depends on the project objectives, budget and the available data to support model parameterisation. Decision trees are a tool that can be used to match methods with specific desired project outcomes (Toth et al., 2001), including biodiversity enhancement, carbon sequestration, water quality, and flood protection. In Fig. 4, a decision tree illustrates how model selection decisions may be made when the restoration objective is to maximise blue carbon sequestration. Key junctions in the decision pathway are described below.

4.1. Evaluating the need for hydrological models in planning for hydrological restoration for blue carbon

4.1.1. Project feasibility

The feasibility stage of a hydrological restoration project offers an opportunity to assess the need for a hydrological model based on the project objectives (Figure 1 and 4A). Building a conceptual model (Section 3.4.1) is the first step to understanding the features of a potential restored wetland site, identifying key processes and interactions, and determining knowledge and data gaps. Typically, at the project's inception, a project developer will have a good idea as to whether they have the capacity and data to build a hydrological model or not. The decision to build a more complex process-based hydrological model (or not) at the start of the project alleviates the risk of wasting resources on

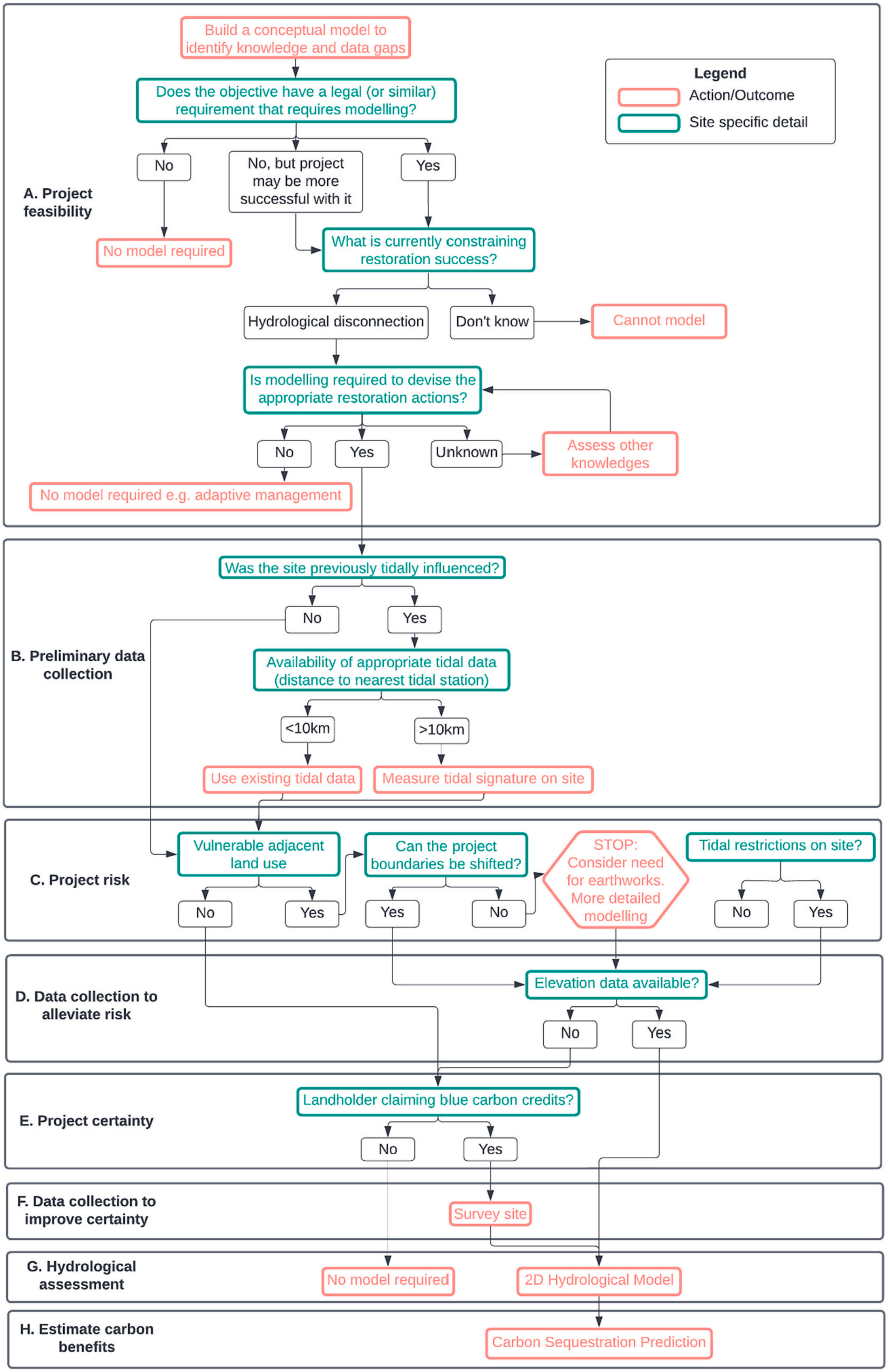

Fig. 4. An example of decision trees that can be used for the selection of hydrological models to plan for a blue carbon project using hydrological restoration.

making a model that is not required particularly when resources are scarce and may be more effectively deployed elsewhere.

If a hydrological model is not required, but there is some uncertainty regarding project outcomes, a hydrological model may still be useful. However, it is important to identify the primary constraints to restoration success before deciding whether to pursue modelling as a part of project planning. If the constraints are unknown, the development of a hydrological model would be ineffective, as the purpose of a model is to improve predictions by addressing known limitations.

In cases where modelling is not necessary, alternative approaches such as adaptive management can be explored. If a constraint (such as limited available space for wetland expansion or limited project budget) can be addressed through other design approaches such as through data collection on elevation and water levels that can be linked to vegetation development, or the constraint is not significant enough to affect project success, modelling may not be required. Hydrological modelling may not always be appropriate in project planning, and other methods such as conceptual models combined with cost-benefit analyses (Table 2) may be more effective (Agaton and Collera, 2022).

4.1.2. Preliminary data collection

Once project feasibility has been established, preliminary data may be gathered to fill knowledge gaps identified by the conceptual model (Fig. 4B). Since the focus is on tidal restoration, the first question that affects hydrological model selection is whether the site was previously tidally influenced. While the term 'restoration' implies that the site is returning towards a more natural state (e.g. hybrid stage (Sheaves et al., 2021)), some projects may be focused on habitat creation or enhancement, which may be part of an adaptation strategy. Additionally, soil subsidence may bring sites that were previously non-tidal within the intertidal zone. Knowing whether a site was or was not tidally influenced in the past can indicate the type of activities (e.g. earthworks) that may be required to return tidal influences on the site.

Where possible, acquiring freely available (or more financially desirable) information is usually more financially desirable than collecting data on-site. This is particularly relevant to projects with limited funding. But the suitability of freely available data may vary. For example, the suitability of available tidal data declines with the distance between the site location and the nearest available tidal data. This is because the greater distance from a tide gauge will influence how representative the data is of the tidal regime at the restoration site. If there is low correlation between the tidal regime at the site and at the tide gauge, collection of site-specific tidal data may be required to improve the accuracy. Alternatively, validation of the tidal regime of a site over a short period against the tidal regime at a tidal gauge, and subsequent calibration can be used to extrapolate data from a tide gauge to a restoration site (Jian et al., 2017). While there are currently no simple predictive methods to estimate tidal height in the upper reaches of rivers or estuaries based on data at the mouth of rivers, it is known that channel geometry (narrowing, widening, cross-sectional area, bends) (Williams et al., 2002) and roughness (sediment type, submerged vegetation present, obstructions) play a large role in determining fluid flow characteristics (Chow, 1959). In the decision tree, the '10 km' distance between the tide gauge and the site has been arbitrarily selected, however some testing should be done to validate and calibrate this irrespective of distance. This is particularly important along estuaries where tidal amplification and attenuation can be profound. Site-specific data is likely a better alternative for accurate predictions of inundation at the site, but this may vary depending on local geometry and roughness of the system.

4.1.3. Project risk

Evaluating the risks the project poses to other adjacent or downstream ecosystems, properties with different tenure and nearby assets and infrastructure important components of planning (Fig. 4C) and may be identified and options weighed using risk tools such as comparative risk assessment or multi-criteria decision analysis (Table 2). For tidal reintroduction projects the potential risk of the project causing flooding to adjacent properties or draining areas that are currently inundated, may have negative environmental or financial consequences. In this context, the risk is a project limitation because this sets up non-negotiable design requirements, such as ensuring adjacent land outside of the project boundaries is not flooded. If the project boundaries cannot be shifted (e.g. adjacent land cannot be brought into the project), then extensive earthworks may be required. In this case, hydrological modelling is likely needed to evaluate the consequences of removing tidal restrictions (e.g. MIKE 21 to investigate the opening of tidal gates (Glamore et al., 2021)).

4.1.4. Data collection to alleviate risk

After identifying the potential risks that a project may pose to adjacent land and the environment, collecting data that can help accurately quantify the likelihood and consequence of those risks can be used (Fig. 4D). Conducting a Value of Information analysis would identify the data that is most useful to collect based on the restoration objective (see Table 2 for examples). Generally, for the case of flood risk assessment, the topography of the site plays a crucial role in predicting the movement of water on site (see examples in Box 2). Therefore, the collection of elevation data may be necessary to alleviate the identified risk. The use of DEMs has a dual advantage; they can be utilised to develop hydrological models to predict potential risks, such as flooding caused by tidal introduction, and can help to quantify the potential for blue carbon benefits (see Section 4.1.8). However, they can be costly to develop (Januchowski et al., 2010). The trade-off for collecting fewer data, specifically elevation data, is reduced resolution or inability to develop site-specific ecogeomorphic and hydrodynamic models.

4.1.5. Specific project requirements (specific to aims)

For tidal restoration projects that aim to sequester carbon using a carbon standard (e.g. VERRA method, Needelman et al. (2018), or the Australian blue carbon method (Lovelock et al., 2022a)) additional requirements of the method may need to be addressed. Other restoration goals (e.g. for biodiversity or water quality) may also have requirements that are specific to those goals (see Supplementary C for examples of project approaches for different objectives). Projects using a market-based method may have a wide range of requirements for project registration. These additional requirements could be those associated with proving that there are limited environmental impacts of the project, or because investors require estimates of likely returns on investment. For both these cases a process-based numerical model (Section 3.4.4) would be beneficial. Models such as TUFLOW or HEC-RAS (see Table S4 for examples) can be used to explore the potential implications of flooding, or MIKE 11 or MIKE 21 (see Table S4 for examples) can be used to investigate changes in water quality with tidal introduction on site. The type of hydrological model needed could be method dependent (Supplementary C). For example, for projects using the *Tidal Restoration of Blue Carbon Ecosystems Method* (Emission Reduction Fund, 2022) carbon sequestration of restored ecosystems is linked to land elevation and therefore a DEM may provide appropriate data to project carbon sequestration, providing hydroperiod is closely related to elevation. Landholders can opt to commit different areas of available land to a blue carbon project based on a cost-benefit analysis where the landholder projects their financial return on committing land to a project versus keeping their existing agricultural (or other) practices. Without an accurate DEM, or where a DEM cannot be related to hydroperiods, emissions abatement could be overestimated or underestimated, adversely affecting project decisions. In addition to increasing accuracy through finer scale models, empirical or process-based may provide projections of sediment transport, water quality including dissolved oxygen, salinity, algae concentrations, pollutant dispersion, and the effect of different management strategies using models like Corps of Engineers Quality Width averaged 2D (CE-QUAL-W2) (Bowen and Hieronymus, 2003), Hydrological Simulation Program – FORTRAN (HSPF) (Shenk and Linker, 2013), or Soil and Water Assessment Tool (SWAT) (Upadhyay et al., 2022).

4.1.6. Data collection to reduce uncertainty

If a project has an objective to restore for carbon credits but a suitable DEM is not available, site surveys of elevation can enhance the accuracy in predicting carbon sequestration (Fig. 4E). The land elevation is often required to link vegetation growth (and therefore biomass) to hydrological limits (e.g. Napa-Sonoma Marsh Ponds project, Box 2). Accurately characterising the elevation of water storage areas like creeks and drains can be challenging because they are not well captured by LiDAR or satellite-derived bathymetry due to their high turbidity, variable wave climate, and depth (Pacheco et al., 2015). Consequently, these characteristics, may be inaccurately represented in DEMs because of high turbidity, breaking waves, and varying water depth in coastal environments. In these instances, the best course of action could be to outline the risk and uncertainty in hydrological processes and associated carbon sequestration predictions so the landholder/project can adjust decision-making appropriately.

4.1.7. Hydrological assessment

A hydrological assessment can be used to provide valuable insights into inundation patterns and to establish relationships between hydrological regimes and vegetation distribution and conditions (Fig. 4F). Other factors such as the relationship between elevation or distance from water bodies and vegetation characteristics could also be assessed. Such relationships can then be used to generate testable hypotheses about the impacts of hydrological modifications (or restoration works) on the hydrological system and its components. For example, a process-based hydrological model (analytical or numerical) can be used to predict the impacts of different interventions (e.g. drain, bund and gate removal) on future flooding extent and to investigate the impacts of sea-level rise on the restoration project (see Haines (2013), Rayner and Glamore (2010) and Karim et al. (2021) in Supplementary C). Some models can assess sediment fluxes that could arise due to these hydrological alterations (e.g. TUFLOW (Haines, 2013), OpenFoam (Le Minor et al., 2019)). Models to simulate sea level rise by adding a water level to a DEM (often called bathtub models and undertaken using a geographic information system) can be used to identify if the project is likely to result in flows of water off the project area which can be used to evaluate the need for more detailed models (see Dittmann et al. (2019) and Luke et al. (2017) in Supplementary C).

4.1.8. Estimating carbon benefits

Estimating carbon sequestration benefits can be linked to hydrological models using knowledge of the association between hydrological conditions, usually hydroperiod (Supplementary A), and rates of vegetation growth, soil carbon accumulation and elevation gains. In some empirical models based on spatial relationships (e.g. SLAMM) the program developer provides a carbon sequestration module (Clough et al., 2010). In other process-based analytical models, carbon sequestration is estimated using site-specific data on the inundated extent and tidal levels (e.g. Blue Carbon Accounting Model) (Clean Energy Regulator, 2022, Lovelock et al., 2022a).

5. Conclusion

The impediments to the widespread adoption and effectiveness of hydrological restoration of coastal wetlands include high project costs, risks, and uncertainty. Hydrological models can be used to assess risks and uncertainty. However, the selection of the most appropriate hydrological model for a restoration project depends on the project's objectives, and therefore, the need for expensive, time-consuming, and data-intensive models may not be warranted in all cases. In many cases, restoration project objectives can be accomplished with the use of simple models. Relying on detailed models as a default solution may not always guarantee optimal restoration outcomes, although they can be useful and are sometimes needed. To improve the uptake and success of hydrological restoration projects, practitioners could consider, in the first instance, whether complex hydrological models are needed, and whether simplified models will suffice.

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CRediT authorship contribution statement

Alice J. Twomey: Writing – review & editing, Writing – original draft, Visualization, Project administration, Conceptualization. **Karinna Nunez:** Writing – review & editing, Writing – original draft, Visualization, Conceptualization. **Joel A. Carr:** Writing – review & editing, Writing – original draft, Visualization, Conceptualization. **Steve Crooks:** Writing – review & editing, Conceptualization. **Daniel A. Friess:** Writing – review & editing, Conceptualization. **William Glamore:** Writing – review & editing, Conceptualization. **Michelle Orr:** Writing – review & editing, Conceptualization. **Ruth Reef:** Writing – review & editing, Conceptualization. **Kerrylee Rogers:** Writing – review & editing, Conceptualization. **Nathan J. Waltham:** Writing – review & editing, Conceptualization. **Catherine E. Lovelock:** Writing – review & editing, Writing – original draft, Supervision, Resources, Project administration, Funding acquisition, Conceptualization.

Declaration of competing interest

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Data availability

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