Estuarine, Coastal and Shelf Science 289 (2023) 108376

Contents lists available at ScienceDirect

Estuarine, Coastal and Shelf Science

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

Tidal barriers and fish – Impacts and remediation in the face of increasing demand for freshwater and climate change

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ARTICLE INFO

Keywords:
River
Estuary
Hydrodynamics
Fishway
Environmental flows
Diadromy

ABSTRACT

Worldwide, tidal barriers (e.g. barrages, dikes, tide gates) are constructed in the lower reaches and estuaries of rivers to limit saltwater incursion into upstream freshwater reserves, facilitate water diversion and abstraction, limit flooding, reclaim land and generate electricity. While performing these functions, tidal barriers also affect fish through: 1) reduced connectivity; 2) loss of tidal flux; 3) conversion of upstream estuarine habitats to freshwater; and 4) diminished freshwater discharge, which compresses the spatio-temporal salinity regime of downstream estuarine habitats. As such, tidal barriers commonly cause declines of diadromous and estuarine associated fish species, with a subsequent loss of ecosystem services. These impacts will be exacerbated as climate change promotes sea-level rise and alters freshwater flow regimes and will be amplified by increasing demands for freshwater by a growing human population. As a result, more tidal barriers are likely. Nevertheless, in estuaries with tidal barriers, management that promotes connectivity and more natural ecosystem function is increasing but remains complex from ecological, economic and engineering perspectives. We present case studies from the Netherlands, southeastern United States and southern Australia to characterise impacts on fishes in different biogeographical regions and document contemporary approaches to restoring ecosystem function and fish populations in systems with tidal barriers. To meet these goals, we suggest three key considerations for future research and management are provision of fish passage, reinstating tidal flux and delivering environmental flows.

1. Introduction

Estuaries are unique ecosystems at the transition between freshwater and marine environments. Humans are disproportionately concentrated around rivers and their estuaries due to the ecosystem services they provide, including fertile floodplains, transport and trade, fisheries, recreation and public amenity (Costanza et al., 1997; Small and Nicholls, 2003). Consequently, the majority of the world's rivers and estuaries have been anthropogenically modified (Dudgeon et al., 2006; Lotze et al., 2006). In estuaries and lower reaches of rivers, tidal barriers (e.g. barrages, dikes, tide gates) are commonly constructed to prevent saltwater intrusion into upstream freshwater resources, regulate freshwater discharge, limit storm surge for flood defense and to reclaim land for agricultural use (Burt and Rees, 2001), and occasionally, to generate electricity by harnessing tidal power (Retiere, 1994). While performing these critical functions, tidal barriers also fragment aquatic ecosystems and alter hydrodynamics, which in turn impact ecosystem processes and biota, including fishes.

The physical, chemical and biological nature of estuaries are influenced by the interaction of riverine flow and tidal exchange (Wolanski and Elliott, 2015). Tidal barriers alter these dynamics, while upstream diversion of riverine flow amplifies impacts. In many cases, freshwater

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https://doi.org/10.1016/j.ecss.2023.108376
Received 8 March 2023; Received in revised form 16 May 2023; Accepted 20 May 2023
Available online 22 May 2023

and estuarine habitats are separated, and tidal exchange is lost, with attendant changes to sediment and nutrient transport, and salinity gradients (Leentvaar and Nijboer, 1986; Burt and Watts, 1996). This fundamentally alters habitats and obstructs fish movement (Gough, 1996), and ultimately alters fish assemblages (Yoon et al., 2016, 2017). Tidal barriers, by nature of their position at the end of catchments, often have the greatest impact on diadromous and estuarine-associated fishes (Nunn and Cowx, 2012).

Estuaries are used by fishes with a diversity of life histories (the estuarine use life history guilds of Potter et al., 2015 are used hereafter). Estuaries represent critical habitats throughout ontogeny for solely estuarine species, migratory pathways between marine and freshwater environments for diadromous fishes, and nursery habitats for a range of marine species (Beck et al., 2001; Elliott et al., 2007). Consequently, the mechanisms by which tidal barriers impact fishes differ among species and life stages. Tidal barriers, in concert with altered riverine discharge, have been associated with declines in a range of estuarine associated fishes and fisheries worldwide, with associated ecological and economic consequences (Drinkwater and Frank, 1994; Raat, 2001; Gillanders and Kingsford, 2002).

Tidal barriers have a long history of impacting fish, particularly in the Netherlands and China, where for hundreds of years, polders have been used to transform estuaries to arable farmland (Hoeksema, 2007; Griffiths et al., 2013). From the 1800s, tidal barriers across river mouths became more prevalent as engineering became more sophisticated. Tidal barriers represent a significant contemporary ecological issue due to climate change and increasing human demands on coasts and catchments, primarily via sea level rise, coastal land subsidence and reduced freshwater inputs. Recent estimates suggest a rate of global mean sea level rise of 3.7 mm. yr^{-1} for the period 2006–2018, which may increase in subsequent years, while absolute global mean sea level rise of 0.28–1.02 m is predicted by 2100, relative to the 1995–2014 average (lower and upper confidence bounds of SSP1-1.9 and SSP5-8.5) (IPCC, 2021). This will increase the frequency of saltwater incursion events into the estuaries and lower reaches of many rivers, with associated flooding risks (McGranahan et al., 2007; Jongman et al., 2012). In heavily populated coastal regions (e.g. New York City, United States of America; Jakarta, Indonesia) the economic impacts may be catastrophic (Aerts et al., 2014). Mitigation of the impacts of rising sea levels will necessitate the construction of new tidal barriers, and amplification and increased frequency of closure of existing barriers (Umgiesser, 2020). This will be compounded by increasing consumptive and agricultural demand for freshwater with increasing human population, and particularly in arid and semi-arid regions, predicted climate-induced reductions in river flow (Palmer et al., 2008; Immerzeel et al., 2010). The result will be further decreases in freshwater discharge to many already freshwater-deprived estuaries (e.g. Zampatti et al., 2010).

The importance of estuaries for a diversity of fishes provides a context for the remediation of connectivity and hydrodynamic processes in systems with tidal barriers. Nonetheless, despite being a global issue, remediation of tidal barriers in relation to impacts on fishes has largely been restricted to parts of Western Europe (e.g. the Netherlands, the United Kingdom), the United States, Asia (e.g. South Korea) and Australia. Furthermore, in addition to considering the impacts of existing tidal barriers, there is a need to incorporate principals of connectivity and ecosystem sustainability into the construction of new tidal barriers.

To guide mitigation, we present a review on the impacts of tidal barriers on fishes and current remediation, and propose future directions. We describe tidal barriers in engineering and ecological contexts, including associated alteration of connectivity and estuarine hydrodynamics, and impacts on fish movement, habitats and populations. Subsequently, case studies from the Netherlands, southeastern United States and southern Australia are used to detail contemporary approaches and the complexity of rehabilitating fish passage and estuarine ecosystem function in systems with tidal barriers. The regions were selected to represent a range of climates, river hydrologies and remediation approaches, while also being the subject of considerable peer-reviewed research. We integrate insights from these case studies, and other published works, to suggest future directions for research and remediation in the short- (<30 years) and long-term (30–100 years), with a particular focus on three key areas, namely: the application of fish passage solutions; tidal restoration; and the provision of environmental flows. Ultimately, we propose a vision for the management of estuaries with regard to tidal barriers and fishes, and the predicted global impacts of climate change and increasing human population.

2. Defining tidal barriers

We generally define tidal barriers as structures built across natural flow paths in the tidal zones of rivers and estuaries with the aims of excluding saline water to create arable land or a freshwater storage; control tidal flux (including for the production of hydroelectricity); and/or to regulate freshwater discharge. They are varied in form and function but typically comprise levee(s) and water control structure (WCS) components (e.g. sluices, weirs, culverts) to regulate water flow. Burt and Rees (2001) categorised tidal barriers based on engineering characteristics and interaction with tide; we have adapted these categorisations and provided ecological context. We propose that tidal barriers fall within four main categories that range from minor/infrequent through to major/frequent alteration of tidal flux and freshwater flow, namely: 1) surge barriers; 2) part-tide barriers; 3) tide-exclusion barriers; and 4) pumping stations (Fig. 1).

2.1. Surge barriers

Surge or flood barriers are operated only during periods of extreme flood risk when spring tides and storm surges coincide, but otherwise the barrier gates remain open (Fig. 1a–b, 2a). There are, to our knowledge, eight large surge barriers protecting cities in Europe, Russia, North America and the United Kingdom (e.g. the Thames Barrier, England). Due to infrequent operation, these structures putatively have only limited impact on fish migration and estuarine hydrodynamics, and thus, are not a focus of this review. It should be noted, however, that rising sea levels and land subsidence are likely to make these structures more common and more frequently used (Umgiesser, 2020).

2.2. Part-tide barriers

Part-tide barriers are diverse in form and function and are defined as those that restrict but do not preclude tidal flux (Fig. 1 c-d, 2 b). They can be grouped by spatial-scale, namely: barriers that regulate lateral connections of local-scale creeks, wetlands, marshes and lakes, that have no or little regular freshwater flow; and large, main channel river sites with potentially high freshwater flow. The local-scale sites are ubiquitous in coastal nations worldwide and barriers are commonly used to prevent saltwater inundating agricultural land. Both local-scale and main channel part-tide barriers typically incorporate: 1) fixed-crest weirs that set a prescribed upstream water level but may be ‘overtopped’ by tides of a given height; and/or 2) tide gates, which may allow some upstream tidal flux, automatically closing during the flood tide and opening again during the ebb tide, when there is positive hydraulic head between upstream and downstream. In some cases, these structures may allow regular connectivity (e.g. daily) at specific stages of the tide (Seifert and Moore, 2018).

Large main channel sites also include run-of-estuary tidal power stations. Currently, there are approximately nine of these sites worldwide, with notable examples on the Rance River, France (Retiere, 1994) and Lake Siwha, South Korea (Kim et al., 2017). These structures typically allow tidal ingress through sluices during the flood tide, but during the ebb, sluices are shut and water is discharged through turbines to generate hydroelectricity. Several additional sites are under

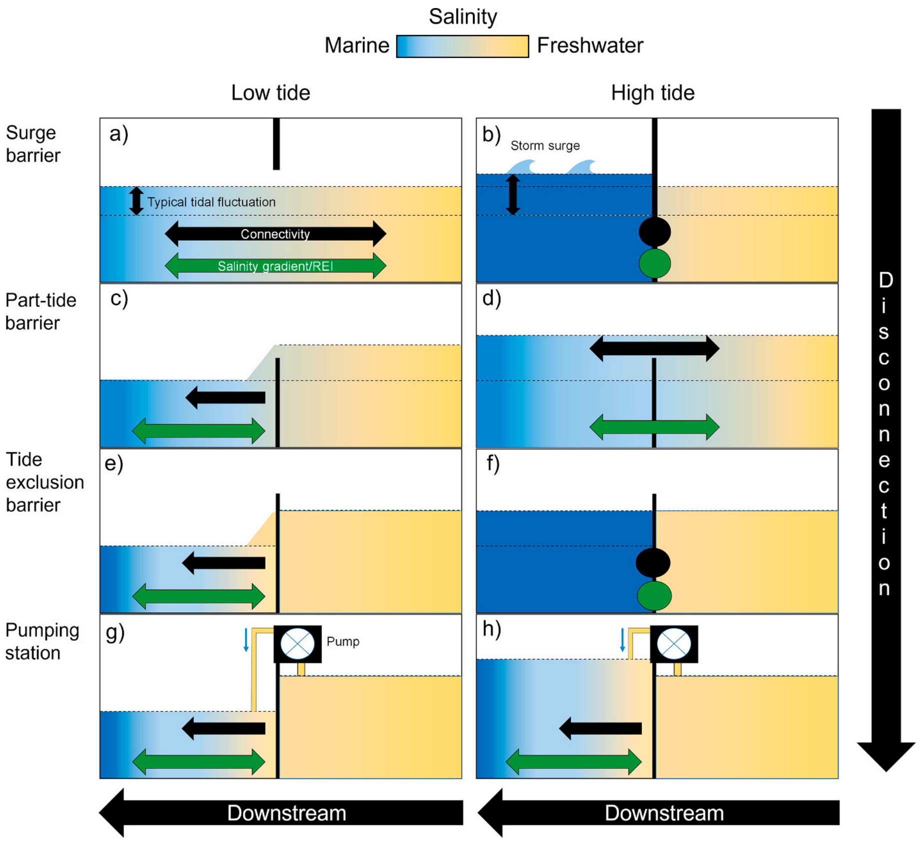

Fig. 1. Generalised conceptual diagrams of hydrodynamics experienced in systems with specific tidal barriers, namely surge barriers (a & b), part-tide barriers (note 'overshot' design used here) (c & d), tide exclusion barriers (note 'overshot' design used here) (e & f) and pumping stations (g & h). Black arrows indicate direction of connectivity (upstream and/or downstream) and green arrows the extent of the salinity gradient or river-estuarine-interface (REI). a) Surge barrier not in operation. Tidal fluctuation promotes extensive and dynamic salinity gradient. Free upstream and downstream fish movement; b) Surge barrier closure during storm surge. Upstream tidal propagation and connectivity temporarily lost; c) Freshwater discharged downstream during low tide and promotion of downstream salinity gradient. Downstream passage may be provided, upstream passage obstructed; d) Barrier overtopped at high tide. Promotion of upstream salinity gradient. Downstream and upstream passage may be provided; e) Freshwater discharged downstream during low tide. Upstream environment maintained as 'freshwater'. Promotion of downstream salinity gradient. Downstream passage may be provided, upstream passage obstructed; f) Discharge ceases at high tide or during drought. Upstream and downstream characterised as 'freshwater' and 'marine', respectively. No connectivity for upstream or downstream passage; g) Freshwater discharged downstream during low tide. Upstream environment maintained as 'freshwater'. Promotion of downstream salinity gradient. Downstream passage may be provided through 'fish-friendly' pump, upstream passage obstructed; h) Freshwater may be discharged downstream during high tide. Upstream environment maintained as 'freshwater'. Promotion of downstream salinity gradient. Downstream passage may be provided through 'fish-friendly'pump, upstream passage obstructed.

Fig. 2. Examples of tidal barriers including: a) the Thames River surge barrier, England (photograph by T. Corser, distributed under CC-BY/SA 2.0 license United Kingdom. Image cropped); b) tide gates, Louisiana, USA (M. Kimball); c) Tauwitchere Barrage, Australia (C. Bice); and d) Nieuwe Statenzijl, Netherlands (reproduced with permission of RWA Hunze en Aa's, Netherlands).

investigation for feasibility, some of which have been repeatedly considered (e.g. the Severn estuary, England), and it is likely such installations will become more common in association with a greater focus on renewable energy production (Hooper and Austen, 2013).

2.3. Tide-exclusion barriers

Tide exclusion barriers are the archetypal flow regulating structures in estuaries and aim to prevent intrusion of seawater to protect human assets and agricultural land, and freshwater reserves for consumptive use. These structures fundamentally alter estuarine and river hydrodynamics and represent significant barriers to fish passage (Fig. 1e–f, 2c-d). Notable examples include the Nile Delta Barrage (Nile River, Egypt) and Afsluitdijk (Rhine Catchment, the Netherlands). These structures are often long (hundreds of metres to kilometres), being located at the widest part of rivers near where they enter the sea and are commonly comprised of levees and WCSs of varying design.

A unifying hydrological feature for the majority of these sites is low seasonal baseflows. This includes dryland rivers, rivers in the dry tropics, and small temperate rivers. Key exceptions are the Zuiderzee and Delta Works in the Netherlands, which were built largely to protect upstream polders, and some barriers built to improve perceived aesthetics or recreational value (e.g. Cardiff Bay Barrage, Wales). In these cases, freshwater discharge may remain similar to that pre-barrier construction. For sites with low baseflows, however, the barrier not only prevents saline intrusion during low flows, but creates a storage of freshwater and, in most large rivers, is used to regulate flow from upstream dams (e.g. the Nile River, Egypt and River Murray, Australia). The result is that all impacts of flow abstraction in the upstream catchment accumulate at the tidal barrier, causing declines of freshwater discharge to downstream estuarine habitats.

2.4. Barriers with pumping stations

A unique subset of tide excluding barriers are those with pumping stations that enable water to be discharged even when downstream water levels (i.e. the seaward side) exceed those upstream of the barrier (Fig. 2 g-h). These barriers are constructed with the aim of excluding saltwater, prescribing a defined upstream water level and impoldering land for agriculture (e.g. in the Netherlands, Germany, Belgium, Bangladesh) and urban development (e.g. New Orleans, United States). They typically include single or multiple pumps and other WCSs (e.g. sluices) to freely discharge water when possible. Impoldering permanently changes tidal aquatic habitat to mainly terrestrial habitat, while downstream pumping of freshwater follows a stochastic pattern depending on rainfall and upstream water levels. Climate change and sea level rise are likely to increase the prevalence of tidal barriers with pumping stations, while seeing the function change from gaining new land to protecting existing land.

3. Impacts on fishes

The ecological impacts of tidal barriers vary due to differing purpose, design and geographical setting, but typically impact fishes via three primary mechanisms: 1) directly obstructing movement; 2) loss of upstream tidal flux and estuarine habitats; and 3) alteration of downstream estuarine habitats due to reduced freshwater flows and changed tidal flux.

3.1. Direct obstruction of movement

Direct obstruction of fish movement by tidal barriers most severely impacts diadromous species. Tidal barriers represent the initial barrier encountered by upstream migrant adults of anadromous species, and juveniles of catadromous and amphidromous species. In systems with tide exclusion barriers, including those with pumping stations, diadromous species may be effectively lost from upstream habitats (Raat, 2001; Yoon et al., 2016). Part-tide barriers, whilst providing greater levels of connectivity, also obstruct upstream passage for much of a tidal cycle (Russell et al., 1998; Silva et al., 2017), which may ultimately lead to failed migration or reduced fitness upon reaching spawning grounds. Downstream migration of diadromous species may also be obstructed. In many cases, tidal barriers are low structures (<5 m) and provide regular discharge, and therefore, effective downstream passage is often assumed. Nonetheless, during drought and in arid catchments where proportionally large volumes of water are used for human consumption and agriculture, barriers may remain closed for extended periods to retain freshwater supplies, obstructing migrations and impacting recruitment (Bice et al., 2018b). Downstream passage at tidal power generating structures and pumping stations present further issues for fishes, and like riverine hydroelectric installations, includes turbine passage-related injury and mortality of downstream migrants (Dadswell et al., 2018).

The dynamic nature of estuaries and use of these environments by a diversity of species, means that migration past barriers is not only important for diadromous fishes, but also for freshwater, estuarine and marine species. Freshwater species are common in some estuaries during and following periods of high freshwater discharge (Whitfield, 2015) through incidental transport downstream, or active movement and use of newly accessible habitat. Species with limited tolerance for elevated salinity subsequently attempt to move upstream as flow declines and salinity rises (Baptista et al., 2010; Brevé et al., 2019). Aggregations can form in areas immediately downstream of tidal barriers and may result in high levels of predation and physiologically mediated mortality (Bendall and Moore, 2008). Many estuarine and marine fishes use estuaries for various purposes from foraging through to nursery habitats (Beck et al., 2001), and move extensively among different habitats (e.g. estuarine lagoon to wetland) and often over short temporal scales (e.g. tides) (Kimball et al., 2017). Tidal barriers can obstruct these movements (e.g. Kimball et al., 2015).

3.2. Loss of upstream tidal flux and estuarine habitat

Tide-exclusion barriers result in the disassociation of freshwater-estuarine-marine habitats and commonly, upstream habitats are transformed from brackish to freshwater with corresponding declines in estuarine-associated fishes and increases in freshwater species (Raat, 2001; Yoon et al., 2016). Part-tide barriers do not result in complete separation of freshwater and estuarine environments; nevertheless, there is a reduction of tidal fluxing upstream which is commonly associated with altered salinity and sedimentation regimes, dissolved oxygen stratification and hypoxia, and changes in fish assemblages upstream of barriers (Franklin and Hodges, 2015; Gordon et al., 2015). Nursery function is often diminished in these habitats (Scott et al., 2016) and in cases of hypoxia, fish kills may result (Beatty et al., 2018). Several studies have demonstrated the degree of dissimilarity in assemblages among estuarine wetlands (marshes) is associated with the degree of tidal restriction, with severely tidal-restricted wetlands typically characterised by lower densities and species richness than lesser restricted wetlands (Raposa and Roman, 2003; Ritter et al., 2008).

Tidal flux in estuarine environments creates a spatio-temporal diversity of hydraulic habitats (e.g. fast-flowing and still water), while resulting erosion and sedimentation influence physical habitat (Wolanski and Elliott, 2015). This diversity of habitats in turn supports a diversity of fish that are adapted to these dynamic environments. The action of tidal barriers, however, reduces the amplitude of tidally-driven water level fluctuations and current velocities upstream (Leentvaar and Nijboer, 1986) and subsequently leads to a loss of habitat diversity. Additionally, more constant water level leads to persistent shoreline erosion while reductions in current velocities alter the transport of suspended materials and increase siltation both upstream and downstream of barriers (Ferguson and Wolff, 1984; Zhu et al., 2017).

Ultimately, tidal barriers impact estuarine hydrodynamics in a way that alters hydraulic habitat and estuarine morphology to the detriment of habitat diversity.

3.3. *Alteration of estuarine habitats downstream due to reduced freshwater flows*

Many estuarine-associated fishes are dependent on or associated with a gradation of salinity from freshwater to marine. This gradient provides a primary habitat for feeding, refuge, spawning and recruitment (Whitfield, 2005). Among diadromous species, catadromous and amphidromous fishes can be grouped by the habitats on which their larvae or juveniles depend; strictly marine (e.g. larvae of Anguillidae) (Feunteun, 2002); or strictly estuarine (e.g. estuary perch *Macquaria colonorum*) (Walsh et al., 2013). The latter group is completely dependent on brackish salinities for completion of their life cycle, and recruitment can be linked to high freshwater inflows and an expansion of the estuarine gradient and habitat (Stoessel et al., 2018). The same largely applies to estuarine-associated marine-spawned fishes that use estuaries as nurseries for early life stages (Whitfield, 1990; Beck et al., 2001).

A common function of tide-exclusion barriers is to store and divert freshwater. These barriers not only divert flow locally but receive the cumulative impacts of all flow diversions in the catchment. In unregulated systems, during periods of low discharge, the freshwater-estuarine interface shifts upstream and, whilst diminished in size, provides an estuarine 'refuge' that supports critical life history processes and populations (Bate et al., 2002). In systems with tide-exclusion barriers, however, spatial and temporal shift of this interface cannot occur. Low flows and the presence of a tidal barrier act to physically compress the area downstream with salinities below marine (i.e. coastal squeeze; Pontee, 2013) and in the worst cases, where high rates of diversion and drought prevent any freshwater passing tidal barriers for prolonged periods, a hard freshwater-marine interface is created with no estuarine refuge (Zampatti et al., 2010). Such changes to estuarine salinity regimes commonly result in increased frequency of occurrence of marine straggler species and declines in abundance and distribution of estuarine-dependent species (Whitfield, 1999; Baptista et al., 2010), with flow-on effects for trophic dynamics and ecosystem function, as well as fisheries production (Gillanders and Kingsford, 2002; Robins et al., 2005; Gillson, 2011).

Altered timing and magnitude of freshwater discharge to estuarine and marine habitats can also impact cues for upstream migration. Indeed, upstream migration mediated by reduced salinity and odours of freshwater origin has been demonstrated for anadromous salmonids (Johnsen and Hasler, 1980) and lampreys (Meckley et al., 2014), and catadromous anguillid eels (Tosi et al., 1990), as well as juveniles of a range of marine-estuarine opportunist species (James et al., 2008; Havel and Fuiman, 2016). Reductions to freshwater discharge associated with tidal barriers have the potential to decrease these stimuli and impact estuarine ingress for numerous species.

4. Impacts on fisheries

Globally, estuaries support numerous commercial, artisanal, subsistence and recreational fisheries (Blaber et al., 2022); for instance, in the United States from 2000 to 2004, 46% of weight and 68% of value of commercial fish and shellfish landings were comprised of species reliant on estuaries at some stage of life (Lellis-Dibble et al., 2008). The Hilsa shad (*Tenualosa ilisha*), an anadromous clupeid distributed across Asia from Sumatra to Kuwait and a primary fisheries resource throughout its range (Hossain et al., 2019), exemplifies the influence of tidal barriers on migration and alteration of upstream habitats and subsequent fishery impacts. In the delta of the Ganges River, which comprises multiple branches and spans southern India and Bangladesh, the construction of tidal barriers (e.g. Farraka Barrage) has obstructed extensive upstream migrations of Hilsa shad leading to profound declines in upstream fisheries (Ahsan et al., 2014; Hossain et al., 2019). The construction of the Afsluitdijk in the Netherlands saw similar declines in economically important fisheries (see below case study). Furthermore, declines in freshwater flow to estuaries, which often accompany tidal barrier construction and operation, have also been implicated in declines of many fisheries worldwide (Rowell et al., 2008; Gillson, 2011). As such, together with ecological impacts, there are significant economic and social impacts associated with the influence of tidal barriers on estuarine habitats and fishes.

5. Case studies

To better specify the impact of tidal barriers on fishes and to introduce approaches to remediation we present case studies from: 1) the Netherlands; 2) the southeastern United States; and 3) the Murray-Darling Basin, Australia. These case studies represent a gradation of spatial scale from national to regional to catchment, and present different drivers for tidal barrier construction and operation, and approaches to remediation. Climatic conditions also differ and are considered as temperate, sub-tropical and semi-arid, respectively.

5.1. *The Netherlands*

The Netherlands coastline comprises the deltas of four major river

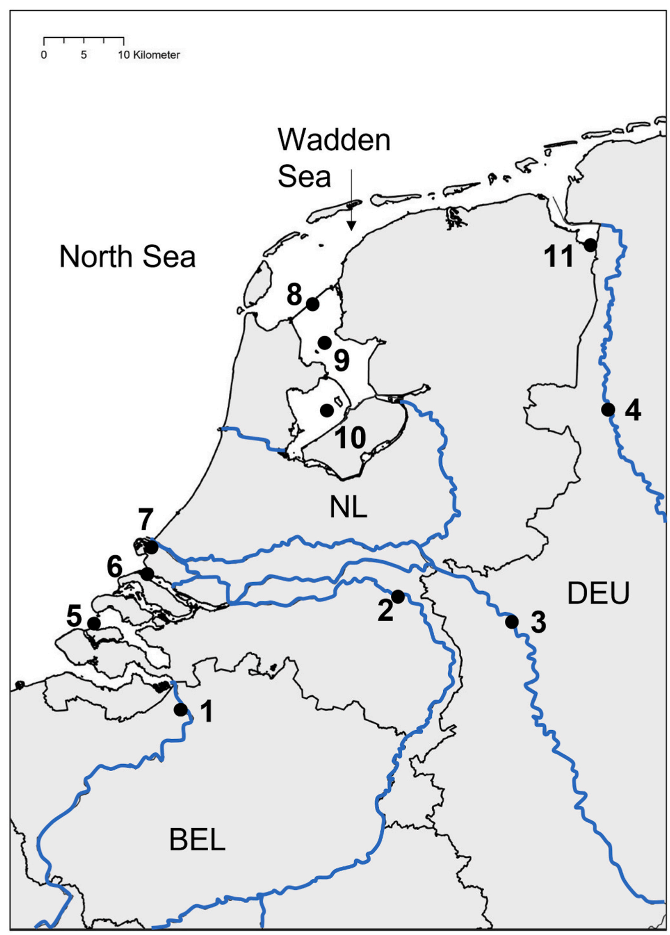

Fig. 3. The Netherlands (NL) and parts of bordering Belgium (BEL) and Germany (DEU). Major rivers are indicated including the Scheldt (1), Meuse (2), Rhine (3) and Ems (4). Additional geographic features and key tidal barriers are also indicated including the Oosterscheldedam (5), Haringvlietdam (6), Nieuwe Waterweg (7), Afsluitdijk (8), Ijsselmeer (9), Markermeer (10) and Nieuwe Statenzijl (11).

systems, the Scheldt, Meuse and Ems, in addition to the Rhine, the largest river catchment in western Europe (~185,000 km^2) (Fig. 3). The country has a long history of converting deltas for agriculture and residential use through the construction of dikes, weirs, polders and canals, and regulating tidal propagation and freshwater discharge (Hoeksema, 2007). Whilst works to reclaim land occurred as early as the late iron age (Lascaris and de Kraker, 2013), construction of the dikes and sluices that characterise the modern-day coast occurred predominantly in the 19th and 20th centuries. Notably, the Afsluitdijk was constructed in 1932, which closed off the Zuiderzee (~6000 km^2) from the Wadden Sea and converted this vast estuary into a freshwater environment. In 1953, a storm surge and catastrophic flooding (Gerritsen, 2005) prompted the closing of all estuaries in the Netherlands, except the Scheldt. This included the construction of the tide-excluding Haringvlietdam (length ~5 km with 17 × 60 m sluice gates) and large surge barriers at the Oosterscheldedam and Nieuwe Waterweg. The Wadden Sea in the north of the Netherlands was subject to smaller-scale empoldering and application of pumping stations and discharge sluices to manage water levels in formerly tidal creeks and marshes.

The combined measures to exclude upstream tidal propagation through the construction of barriers has resulted in the Netherlands being well protected from coastal flooding but had a substantial impact on fish and fisheries (Wolff and Zijlstra, 1982). The closure and conversion of the Zuiderzee into the freshwater Ijsselmeer and Markermeer saw associated changes to fish assemblages, including the extirpation of the Zuiderzee herring (*Clupea harengus*) (Redeke, 1939) and declines in numerous diadromous species including Atlantic salmon (*Salmo salar*), European sturgeon (*Acipenser sturio*), European eel (*Anguilla anguilla*), smelt (*Osmerus eperlanus*), river lamprey (*Lamptera fluviatilis*) and European flounder (*Platichthys flesus*) (Raat, 2001; Lotze, 2005).

In recent years, fish passage at tidal barriers in the Netherlands has received considerable attention (e.g. Philippart and Baptist, 2016). This has involved: 1) revised operation of discharge sluices and navigation locks to promote fish passage during short periods (minutes–hours) of limited head differential; and 2) the application of technical fish passes. Importantly, downstream tidal ranges are as high as 4 m and upstream water levels are often below sea-level. Consequently, fish passage systems are often highly technical and vary in design, size and function. Indeed, at single installations, upstream passage systems may incorporate traditional technical fishway components (e.g. vertical-slots), in addition to sophisticated fish collection basins, pumps and siphons.

One such site is Nieuwe Statenzijl (Fig. 2d), a tidal barrier on the River Westerwoldse Aa, which discharges to the Ems-Dollard Estuary on the border of the Netherlands and Germany. The barrier consists of four 8 m high x 5 m long sluice gates for discharging water, and a navigation lock. Research into facilitating fish passage at Nieuwe Statenzijl began in 2001 and investigated the use of the navigation lock for fish passage (Wintermans, 2003) and a revised operating regime for the discharge sluices (Leutscher, 2004). The original regime involved discharging freshwater during low tide and closing the sluices when the tide began to rise and headwater and tailwater levels equalised. The revised regime involved opening one sluice door for approximately 45 min to allow a limited window of upstream tidal propagation as tailwater levels rose above headwater levels, before eventual closure. This was successful in facilitating the upstream passage of European glass eels and three-spined stickleback (*Gasterosteus aculeatus*), but an undesirable consequence was the upstream transport of large volumes of sediment. To further improve passage at the site, but limit sediment transport, two smaller aperture valves (0.5 m diameter) were added to one of the sluice gates. These valves are opened at low tide to provide freshwater attraction flow and remain open for the initial part of the flood tide to allow limited upstream tidal propagation, facilitating passage of European eel, three-spined stickleback, juvenile European flounder and smelt (Krijnsen and Rondeel, 2019). In 2014, an 80 m long bristle elver pass was constructed on the eastern side of the structure which enhances the upstream passage of glass eels (Leutscher, 2004). Recent monitoring upstream of Nieuwe Statenzijl has documented European glass eels and European flounder in locations where these species had not been recorded for many years.

The larger tidal barriers in the Netherlands have historically received less attention regarding fish passage due to greater complexity and cost, but this is no longer the case. From late 2018, a revised regime for operation of the Haringvlietdam (the 'Kierbesluit') has been implemented to allow limited upstream fluxing of brackish water and promote fish passage (Beeldman et al., 2018). Additionally, at the Afsluitdijk, construction of the 'Fish Migration River' (FMR) commenced in 2020, which will incorporate a 4 km long by 25 m wide channel, as well as technical fishway sections (i.e. vertical-slots), and when completed (scheduled for 2024) will be the largest tidal fish pass in the world (Fig. 4) (Bruins Slot et al., 2018). This program is being conducted at an estimated cost of €20–30 million. Importantly, a research program is planned to determine the passage effectiveness and ecological benefit of the FMR, and to inform future operation (Griffioen and Winter, 2017).

In the coming years most tidal barriers in the Netherlands will be equipped with technical fish passes and/or adapted management to facilitate fish migration. The restoration of estuarine habitats upstream of barriers, however, has received much less attention owing to the need to protect agricultural and residential land from tidal incursions.

5.2. Southeastern United States

The southeastern United States coast extends ~3500 km from Texas to North Carolina and consists of highly productive estuarine tidal marshes that support numerous fisheries (NMFS, 2017) (Fig. 5). The two primary regions are the southeastern Atlantic coast, from North Carolina to southeastern Florida, and the northern Gulf of Mexico coast from Texas to south-western Florida. Key estuaries of the southeastern Atlantic coast are fed by the Pee Dee, Cooper and Santee rivers in South Carolina; the Savannah and Altamaha Rivers in Georgia; and the St. Johns River in Florida. They generally have semidiurnal micro- or meso-tidal ranges (1–3 m) and are characterised by extensive emergent marsh surface habitats, consisting of networks of tidal creeks, intertidal flats, and open waters. The northern Gulf of Mexico region receives freshwater inputs from 37 major rivers; the Mississippi and Atchafalaya rivers being the largest (Wilkinson et al., 2009). The Gulf region contains approximately 60% of the tidal marshes in the US and is characterised by semidiurnal micro-tides (<0.5 m), and estuaries comprised of large areas of emergent marsh surface habitats with creeks and open water habitats, and multiple large coastal lagoons (Mendelssohn et al., 2017).

Hydrology is managed in many estuaries of the southeastern United States and typically involves impoundment and regulation of water

Fig. 4. Conceptual image of the proposed Fish Migration River adjacent the Kornwerdezand sluices and navigation lock on the Afsluitdijk, Netherlands (reproduced with permission of Provincie Fryslân, the Netherlands – Feddes/Olthof – Landschapsarchitecten).

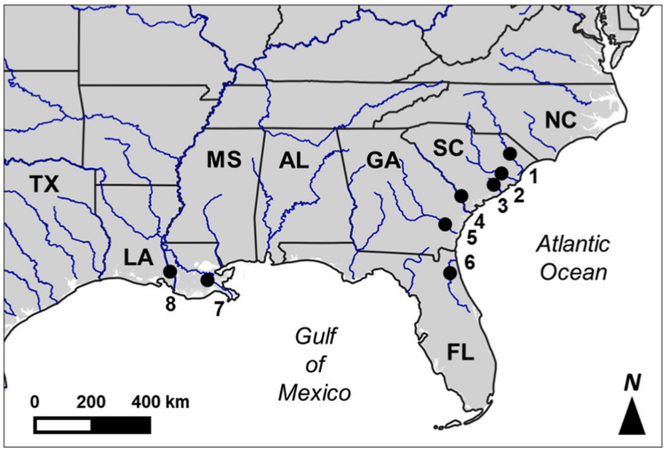

Fig. 5. Southeastern United States coastline depicting the US Atlantic coast (North Carolina, NC; South Carolina, SC; Georgia, GA; and the east coast of Florida, FL) and northern Gulf of Mexico (west coast of FL; Alabama, AL; Mississippi, MS; Louisiana, LA; Texas, TX). Major rivers include (marked with solid black dots with numbers): Pee Dee (1), Santee (2), Cooper (3), Savannah (4), Altamaha (5), St. Johns (6), Mississippi (7), and Atchafalaya (8).

levels with tidal barriers (fixed and variable-crest weirs, slotted weirs, and tide gates) for the purpose of agriculture, mosquito control and waterfowl production (Rogers et al., 1994; Brockmeyer et al., 2022). The extent of managed salt marsh habitats in the southeastern United States is unclear, with historic estimates ranging from 2% in Georgia to 14% in South Carolina (Miglarese and Sandifer, 1982; DeVoe and Baughman, 1986; Montague et al., 1987). Additionally, there is no comprehensive information on the number and types of tidal barriers in use, or whether these are passive (no manipulation possible) or active (can be manipulated or operated) structures; nonetheless, they likely number >10,000.

In these managed habitats, the level of connectivity between impounded marshes and the open estuary influences fish communities (Rogers et al., 1992b; Rulifson and Wall, 2006). For example, during the mid-1900s, managed marshes in Florida were disconnected from greater estuaries by levees with no WCSs, primarily for mosquito control; the number of fish species in these managed habitats was low, comprising mostly solely estuarine species (Brockmeyer et al., 2022). Reconnecting these managed areas to the open estuary by installing WCSs (e.g. tide gates) resulted in an increase in species richness, especially marine-estuarine opportunists (Gilmore et al., 1982; O'Bryan et al., 1990). Nonetheless, WCSs still limit ingress, as evidenced by greater abundances of many species in unmanaged than managed marshes (e.g. Herke et al., 1992). In addition, specific marsh management strategies such as those used to promote waterfowl (Carswell et al., 2015) and control mosquitoes (Rey et al., 2012), including seasonal water-level manipulations with prescribed volumes of tidal exchange, rarely consider impacts on fishes, and commonly result in poor quality fish habitat and impediments to fish migration (Cianciotto et al., 2019). It should be noted, however, that managed marshes may serve as valuable nursery habitats for some marine species during their period of estuarine residency, especially where natural estuarine habitats are scarce (e.g. Elmo et al., 2021).

Despite their ubiquity, few studies have directly examined fish passage through WCSs in the southeastern United States. These typically used traps or nets to examine passage through specific WCS types on part-tide barriers (e.g. tide gates, fixed weirs) and documented obstruction of passage for multiple species (McGovern and Wenner, 1990; Stevens et al., 2006). As a means to improve passage of fish and other nekton, vertical-slots have been retrofitted to weirs in some instances. These work on the principal of a vertical-slot fishway baffle, exchanging water and facilitating fish movement through a portion of the water column for at least part of the tidal cycle (Kimball et al., 2010). Studies comparing unidirectional movement of fish through fixed-crest and flap-gated WCSs and structures with slots concluded that slots improve nekton passage (Rogers et al., 1992a; Rulifson and Wall, 2006). Despite improving passage, comparisons of numbers of individuals (species pooled) observed congregating at slotted WCSs with those that transited the structures (in both directions), derived from high-resolution acoustic imaging (DIDSON), suggested passage rates through slotted weirs were generally <10% (Kimball et al., 2010, 2015). Further studies using mark-recapture (visual implant elastomer) and electronic tagging (passive integrated transponders) have also noted relatively low species-specific passage rates through slotted weirs and various other types of WCSs for the solely estuarine mummichog (*Fundulus heteroclitus*) (Rudershausen et al., 2018) and marine-estuarine opportunists, including sea mullet (*Mugil cephalus*), ladyfish (*Elops saurus*), common snook (*Centropomus undecimalis*), red drum (*Sciaenops ocellatus*) and Atlantic tarpon (*Megalops atlanticus*) (Kimball et al., 2017; Mace et al., 2018; Cianciotto et al., 2019; Wilson et al., 2019).

The dimensions of WCS openings, including vertical-slots, varies depending on site-specific hydrological regimes. Studies at a small number of sites suggest that for structures with slot widths ranging from 0.1 to 0.6 m, width did not influence the number or size of fish successfully passing through the structure (Kimball et al., 2010, 2015, 2017). Nonetheless, it is reasonable to assume that smaller slot sizes may preclude movement of adults of larger species (e.g. red drum, Atlantic tarpon). Water velocities at these structures vary considerably (e.g. 0–1.5 m s^{-1}) as a function of tide and local weather patterns (i.e. storms), suggesting that at certain times, small species and early life stages may be obstructed by water velocities that exceed swimming abilities (Kimball et al., 2018; Rudershausen et al., 2018). It remains unclear, however, why a high proportion of individuals for some large species (e.g. ~50% of Atlantic tarpon) seemingly capable (i.e. not limited by swimming ability or body size) of transiting through WCSs, approach but do not pass (e.g. Mace et al., 2018). There may be behavioral impediments for these species to pass through vertical-slots or these large piscivores may be foraging and taking advantage of accumulations of smaller fishes. Further research is required to determine optimal WCS designs to facilitate greater passage of fish in these environments.

Understanding of the impact of WCSs and marsh management on connectivity and passage of fish among habitats within the estuarine seascape of the southeastern United States has been hampered by a lack of basic information on structures and focus of research efforts on relatively few geographic areas within the region. A census of the number, type, and location of WCSs in the southeastern United States, and their associated management operations (if any) is required; increased accessibility of satellite data (e.g. Sentinel) may facilitate this effort (Kimball et al., 2021). As a result of environmental damage caused by the 2010 Deepwater Horizon oil spill, extensive and ongoing estuarine restoration efforts are occurring throughout the northern Gulf of Mexico region (Baker et al., 2017; Carle et al., 2020), which may afford the opportunity to address some of these data deficiencies. These efforts could establish databases on fish passage in estuarine habitats similar to those for more inland riverine habitats in the region, which have informed prioritisation of subsequent research and restoration efforts (e.g. Martin, 2019).

5.3. The Murray Barrages, Murray-Darling Basin, Australia

The Murray-Darling Basin (MDB), covering 1,073,000 km^2, is Australia's most economically important river system, supporting 70% of the country's irrigated agriculture (AU$7 billion per year) (ABS/ABARE/BRS, 2009). At its terminus, the River Murray flows into a pair of large lakes (Alexandrina and Albert, cumulative area ~900 km^2) that subsequently discharge to an estuarine lagoon system (The Coorong) and ultimately, the Southern Ocean (Fig. 6). The Coorong is a reverse estuary, whereby freshwater enters the estuary close to the river mouth,

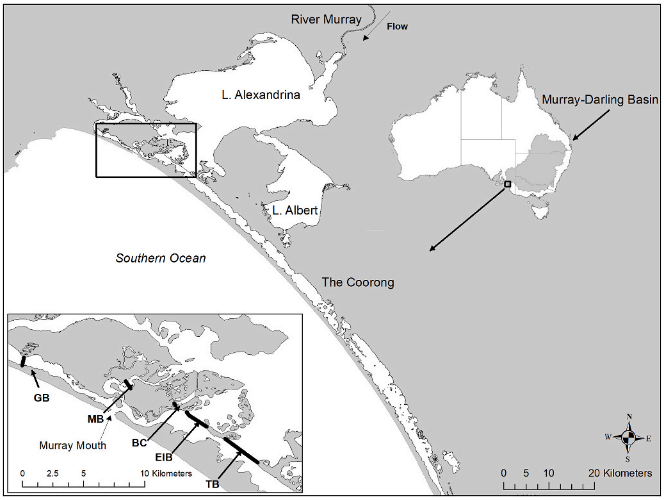

Fig. 6. Map of Lake Alexandrina, Lake Albert and the Coorong at the terminus of the Murray-Darling Basin Australia. Specific detail of the interface between Lake Alexandrina and the Coorong is provided in the inset, presenting the location of the Murray Barrages (GB = Goolwa Barrage, MB = Mundoo Barrage, BC = Boundary Creek Barrage, EIB = Ewe Island Barrage and TB = Tauwitchere Barrage).

but a narrow (<2 km) blind-ended lagoon extends for ~140 km parallel to the coast, with a characteristic salinity gradient ranging to hypersaline at the extremity (Geddes, 1987). In the 1930s, in response to increasing upstream diversion and abstraction of freshwater, and increased occurrence of saltwater incursion events, the Murray Barrages were constructed across the connections of Lake Alexandrina and the Coorong. The network comprises 7.6 km of levees and WCSs (radial gates and operable weirs) that regulate freshwater discharge and exclude saltwater. Upstream regulation and freshwater abstraction have reduced mean end of system discharge (~150 $m^3 s^{-1}$) to <40% of natural discharge (~390 $m^3 s^{-1}$) (CSIRO, 2008). As such, the combination of upstream water diversion and construction of the barrages transformed a dynamic lake and estuarine lagoon system to one that is characterised by a distinct division between freshwater and estuarine environments, with diminished freshwater discharge to downstream estuarine habitats (Mallen-Cooper and Zampatti, 2018).

A total of 103 fish species have been documented from the region, comprising freshwater, diadromous, estuarine and marine species (Bice et al., 2018a). The upstream fish assemblage is now characterised by freshwater species (Wedderburn et al., 2012), while assemblages in the Coorong are typically dominated by solely estuarine and marine-estuarine opportunist species (Bice et al., 2018a). Seven diadromous species have been recorded, including two anadromous lamprey (pouched lamprey *Geotria australis* and short-headed lamprey *Mordacia mordax*) and five catadromous species. Estuary perch was putatively common prior to barrage construction (Eckert and Robinson, 1990), but is now rarely encountered. Congolli (*Pseudaphritis urvillii*), whilst still common, contributes substantially less to commercial fishery catches than in the past, suggesting initial barrage construction and ongoing operation impact the species (Evans, 1991; Bice et al., 2018a).

The Murray Barrages were constructed without provision for fish passage and it was not until the early 2000s when technical fish passes were formally considered (Barrett, 2004). From 2003 to 2018, eleven fishways were constructed, including nine of varying vertical-slot design, as well as nature-like and trapezoidal fishways (Bice et al., 2017a). Fishway design and construction was an iterative process as knowledge on local fish migration and fishway function in Australian improved (Baumgartner et al., 2014). Due to a lack of empirical data on fish movement, the design of initial vertical-slot fishways (internal hydraulics: max velocity ≤2.0 $m s^{-1}$, turbulence ≤95 $W m^{-3}$) was based on the perspectives of commercial fishermen and emphasised the importance of movement for large-bodied (adult length >300 mm) estuarine (e.g. black bream *Acanthopagrus butcheri*) and freshwater species (e.g. golden perch *Macquaria ambigua*) that may move to habitats downstream of the Murray Barrages during high flow. These fishways were effective for golden perch but were not used by large-bodied estuarine fishes, while the passage of fish <100 mm was largely obstructed (Stuart et al., 2005; Jennings et al., 2008). Subsequent monitoring indicated fish <100 mm dominated the migratory fish community year-round, particularly juveniles of catadromous species and displaced freshwater fishes (Zampatti et al., 2010). Thus, additional vertical-slot fishways were constructed with internal hydraulics favourable for the passage of small-bodied fish (max velocity ≤1.1 $m s^{-1}$, turbulence ≤25 $W m^{-3}$). These fishways effectively facilitate the passage of small-bodied fishes under low flow conditions, but during periods of high freshwater discharge, absolute passage was compromised by low fishway discharge, relative to barrage discharge, and subsequently limited attraction of fish (Bice et al., 2017b).

The most recent fishways constructed at the Murray Barrages (2016–2018) include a trapezoidal and two dual vertical-slot fishways designed to produce internal hydraulics that are generally favourable for the passage of small-bodied fishes (max velocity ≤1.7 $m s^{-1}$, turbulence ≤40 $W m^{-3}$), but also discharge moderate-high volumes of water to promote attraction even during periods of high freshwater discharge. The two largest and most commonly operated barrages (i.e. Tauwitchere and Goolwa, Fig. 6) feature multiple fishways that target different size classes of fish, and operate most effectively under different hydrological conditions (i.e. low-flow and high-flow), and thus can be viewed as complementary (Bice et al., 2017a, 2017b). Data collected during standardised monitoring since 2006 suggests that fishway construction has enhanced the abundance of the catadromous congolli (Bice et al., 2018a).

Reduction in freshwater discharge through the Murray Barrages has altered the downstream salinity regime of the Coorong with subsequent impacts on fishes. Most notably, there has been a reduction in the spatio-temporal distribution of meso- and polyhaline regions (salinity 5–30), with associated increases in euhaline and hypersaline regions (salinity >40) (Aldridge et al., 2018). Consequently, in years of low freshwater discharge, for solely estuarine fishes and juveniles of marine estuarine-opportunists that prefer salinities 5–35 distribution is reduced and limited to the region immediately downstream of the barrages and near the mouth of the river (Ye et al., 2016). The commercially important mulloway (*Argyrosomus japonicus*) uses estuaries as nurseries throughout its anti-tropical Indo-Pacific range, and in southern Australia, the Coorong is a crucial nursery for the species (Griffiths, 1996; Ferguson et al., 2014). Year class strength of adult populations in coastal environments in the proximity of the River Murray is associated with years of high freshwater discharge from the Murray Barrages during the year of spawning, and putatively, enhanced nursery function associated with broad areas of low salinity estuarine habitats (Ferguson et al., 2008). While discharge to the estuary is largely a function of broad-scale catchment water availability, the presence and operation of the Murray Barrages influences nursery habitat availability and quality, and recruitment of this iconic species.

In recent years, the importance of freshwater discharge from the Murray Barrages to support downstream estuarine habitats has garnered increasing recognition. 'Environmental water, freshwater allocated specifically to support the needs of ecosystems is now delivered on an annual basis within the framework of the Murray-Darling Basin Plan (MDBA, 2012) and guided by a set of predetermined 'Environmental Water Requirements' (EWRs; Lester et al., 2011). These prescribe return intervals for given annual flow volumes that are required to meet salinity targets within the Coorong that have been developed considering flow-related requirements for a range of indicator taxa, including estuarine associated fishes (Rumbelow, 2018). Uncertainty remains regarding the capacity of environmental flows to support longer term ecological recovery of the Coorong toward pre-development conditions, but to-date, has been successful in achieving several fish-related outcomes. This includes supporting continuous operation of fishways on the barrages since 2010, and specific flow events to promote upstream migrations of diadromous species (Bice et al., 2018a), and to provide conditions favourable for spawning and recruitment of the solely estuarine black bream (Ye et al., 2019).

6. Remediating tidal barriers for fish

Worldwide, tidal barriers presently impact fishes, but under future climate change and sea level rise the nature of these impacts may change. As such, there is a need to consider research and remediation at short- (reactive, <30 years) and longer-term scales (proactive, >30 years). The above case studies present international examples of impacts and approaches for mitigation over the short-term, and highlight that this process is complex, site-specific, often iterative and constrained by competing management issues (e.g. the need to protect upstream land from tidal ingress). Below we draw upon these case studies, and other published works, to summarise current knowledge and suggest future research directions for three key interactive remediation activities at tidal barriers. These are: 1) fish passage solutions; 2) restoration of tidal propagation; and 3) the delivery of environmental flows. In addition, we provide a commentary on potential long-term remediation in light of predicted changes to the nature of estuaries under climate change (Passeri et al., 2015).

6.1. Fish passage

Considerable multidisciplinary (ecology, physiology, engineering) research has resulted in increasingly informed fish passage solutions in riverine settings (Katopodis and Williams, 2012). Fish passage at tidal barriers, however, has not received commensurate attention; for instance, Silva et al. (2018) provides a well-considered contemporary review of fish passage science and application to identify key areas for future research and management, yet tidal barrier passage is not specifically addressed. Specific focus is warranted given tidal barriers present notable ecological and hydraulic challenges regarding fish passage. These include: 1) passage is required by a diversity of species and size classes, including small individuals with weak swimming abilities (e.g. juveniles of catadromous species); 2) tailwater levels are variable over a range of temporal scales (hourly, daily, seasonally) as a function of tide and discharge; 3) upstream water levels can be below mean sea level; 4) barriers are often long, and therefore freshwater discharge may be diffuse, meaning optimising attraction to fishways can be difficult; and 5) freshwater discharge can be highly variable on daily, seasonal and inter-annual scales, particularly in arid and semi-arid systems.

To supplement the case studies, we conducted a search of peer-reviewed literature pertaining directly to fish passage at estuarine barriers using a Web of Science Core Collection search of All Fields (Clarivate Analytics) on January 15, 2022 using the term [(fishway* OR fish passage*) AND (estuary* OR tidal*)]. This search resulted in 212 published records, which were then supplemented by 29 additional publications known to the authors and not identified in the search. Of these 241 records, 61 were deemed to directly focus on fish passage solutions at estuarine barriers (Supplementary Material Table 1). Most studies (87%) specifically addressed upstream passage, whilst a lower proportion (34%) investigated downstream passage (note that percentages amount to >100% as several studies investigated multiple aspects of fish passage). Half (51%) of the relevant studies had a focus on direct passage through WCSs (e.g. sluices, culverts), and often revised operation (e.g. opening for longer periods of time) or alterations to structures (e.g. addition of slots or valves) to enhance passage, while 45% had a focus on application of technical fishways. Approximately 13% of studies detailed passage through navigation locks.

6.1.1. Passage without fishways

Fish passage at many tidal barriers may be improved without the application of specific fish passes, particularly at small-scale, part-tide barriers (e.g. tide gates). Passage at these structures, in part, is a function of frequency and timing of opening; typically targeting a narrow window when water levels are near equal but avoiding reverse flow. Despite the potential simplicity of this solution, assumed and actual opening frequencies of tide gates can vary (Seifert and Moore, 2018). As such, enhanced passage at these structures may be promoted by more rigorous approaches to structure management and maintenance by local natural resource and water management agencies.

Despite apparent connectivity for parts of the tidal cycle, part-tide barriers may represent velocity and behavioral barriers to movement (Russon and Kemp, 2011). Several studies document improved passage by reducing flow velocities and increasing periods of connectivity by providing orifices/valves or vertical-slots within tidal barriers (Kimball et al., 2015; Wright et al., 2016), while general guidelines exist to inform the construction of culverts that are favourable for fish passage (Chanson and Leng, 2020). Many of these actions promote greater levels of passage for many species and are relatively cost-effective and are thus viable at many small-scale, part-tide barriers that impact lateral connectivity. Hydraulic modelling has great promise to inform these decisions, including physical modifications and refinement of barrier operation to promote fish passage (Guiot et al., 2023).

The capacity of altered operation of WCSs to improve passage is not limited to smaller part-tide barriers but is also possible at large tide-excluding barriers where upstream and downstream water levels approach equilibrium during part of the tidal cycle. This has been demonstrated with varying success at several sites in the Netherlands including the sluices of the Afsluitdijk (Bij de Vaate et al., 2003), the Haringvlietdam (Brevé et al., 2019) and Nieuwe Statenzijl (Krijnsen and Rondeel, 2019). In the case of the latter two sites, this has involved incorporation of additional culverts or orifices/valves to assist passage (Brevé et al., 2019). Progress on altered sluice management at these structures has been underpinned by targeted research and modelling (Huisman, 2017). The experience at Nieuwe Statenzijl suggests that, under certain circumstances, improved sluice management may pass greater numbers of fish than technical fishways (Bangma, 2015; Krijnsen and Rondeel, 2019). Nonetheless, despite positive results at specific sites, actions to promote fish passage through WCSs on large tide-excluding barriers is relatively rare and represents a priority for research.

6.1.2. Fishway solutions

In our review of literature we found evidence of 36 fishways that had been constructed on tidal barriers; this was supplemented by further fishways not found in the literature to create a final list of 57 fishways (Supplementary Material Table 2). As just one of the authors (MMC) knew of 11 fishways in Australia that were not in the literature, it is very likely that globally there are many other undescribed tidal barrier fishways. Of the 57 fishways, vertical-slot (35%), nature-like (24%), pool and weir (7%), submerged orifice (7%) and cone (7%) designs were the most common, followed by eel passes (4%), trapezoidal (4%), Denil (4%), Larinier pass (2%), fish sluices (2%), siphons (2%) and fish locks (2%) for general description of fishway types see Clay, 1995). The designs adopted are a function of factors commonly considered for riverine fishways, including target species, estimated biomass and barrier height. Yet, at tidal barriers, tailwater level variation is a primary consideration. In regions with large tidal ranges, fishways may only function for short periods as the tailwater may recede drastically at low tide yet exceed headwater levels at high tide. At sites where fishway function is possible throughout tidal ranges, designs that can best accommodate variable tailwater levels have been preferred (e.g. vertical-slot fishways; Bice et al., 2017b).

Pool and weir designs have been applied in several instances, particularly in the northern hemisphere where salmonids have been a primary target of passage (Russell et al., 1998), but when applied for the purpose of passing fish assemblages, have had limited success (Yoon et al., 2016). The application and refinement of vertical-slot fishway designs has been a theme of fish passage at tidal barriers in Australia, where the passage of whole fish assemblages is often a priority. Several studies have demonstrated the effectiveness of this design in passing a broad range of species and life stages (e.g. <30–950 mm in length) (Mallen-Cooper, 1999; Stuart and Mallen-Cooper, 1999; Stuart and Berghuis, 2002; Bice et al., 2017b). On small tidal barriers, where fishway discharge is a substantial proportion (e.g. ≥10%) of overall discharge, vertical-slot fishways with appropriate internal hydraulics are likely effective (O'Connor et al., 2019), yet on large tidal barriers, this design suffers from poor attraction during periods of high tailwater level and high freshwater discharge (Bice et al., 2017b). Nonetheless, providing auxiliary flow to fishways is now a common approach to improving attraction efficiency and is likely well suited to application at tidal barrier fishways (Adam, 2012; Schütz et al., 2021).

At tidal barriers where headwater levels are stable, or fishway exits are set to be engaged at prescribed upstream water levels, nature-like (rock ramp) fishways can be effective (e.g. Sumiya et al., 1995). Novel cone fishways are also applicable in these situations and have been developed to produce very low turbulence (<25 W m^{-3}) and pass small fish (10–100 mm) – these are technical fishways comprised of pre-fabricated cone-shaped baffles within a concrete channel and produce internal hydraulics similar to step-pool nature-like fishways (Marsden and Stuart, 2019; Stuart and Marsden, 2021). At several tidal barriers in northern Australia, these fishways are effective in passing a range of species and sizes including juveniles of catadromous species 10–100 mm in length (Stuart and Marsden, 2021). Importantly, both of these fishway types, along with trapezoidal fishways, can be designed with wide channels, allowing high discharge through a deeper middle section, which provides the large volumes of water often required to attract fish to fishways in the case of tidal barriers.

We found only one example of a fish lock being applied on a tidal barrier, at the Nagara Estuary Barrage, Japan (https://www.water.go.jp/chubu/nagara/27_english/07/04/02/index.htm). Notwithstanding, fish locks have potential on tidal barriers given they can be effective at passing small fish with limited swimming abilities (Clay, 1995), a common passage objective at tidal barriers. In our literature search, navigation locks were identified in 14% of studies as a means of promoting passage. Several studies documented the successful use of navigation locks to enhance both upstream and downstream movement past tidal barriers when these structures are specifically operated in a manner that promotes attraction and passage (Vincik, 2013; Silva et al., 2017; Bice et al., 2018b). At tidal barriers that incorporate navigation locks, revised operation with consideration of fish passage is a useful adjunct to other fish passage solutions.

6.1.3. Informing passage solutions

Revised operation of tidal barriers and integration of technical fishways requires an understanding of species-specific movement motivation, timing and behaviour, and hydraulics. Worldwide, research pertaining to these factors at tidal barriers remains scant. Indeed, whilst studies that quantify fine-scale movement behaviour or assess attraction efficiency to fishways are common at riverine barriers and fishways (Cooke and Hinch, 2013), few studies of fish movement exist that used telemetry methods (e.g. acoustic, radio and PIT-telemetry) at tidal barriers, although these are becoming more common (see Kimball et al., 2017; Beatty et al., 2018; Bennett et al., 2021).

At the Murray Barrages in southern Australia, in the absence of such empirical data, fishways have been located in close proximity to navigation locks and frequently operated sluices. Under conditions of low discharge, this approach that follows basic tenets of fish passage science, seems appropriate (Clay, 1995; Silva et al., 2018). Yet, at the terminus of river systems, fishway discharge at tidal barriers is often a small proportion of overall discharge; as such, maximising fishway attraction is difficult and few fishways with dimensions and discharge to perform well under such conditions have been constructed on long tidal barriers. To resolve this problem, at many tidal barriers, multiple fish passage solutions have been applied on single barriers, with positive results (e.g. Nieuwe Statenzijl, Netherlands; the Murray Barrages, Australia; Nagara Estuary Barrage, Japan). Alternatively, the Fish Migration River, proposed for the Afsluitdijk in the Netherlands, represents the first effort to provide a singular integrated fish passage that discharges a large proportion of overall barrier discharge, but will be constructed at great cost.

The burgeoning field of ecohydraulics is a priority area of research and will inform future decisions on fish passage solutions at tidal barriers (Mawer et al., 2023). Hydraulic modelling is now a commonly used tool to assess internal hydraulics and attraction conditions to inform fishway design (Bombač et al., 2014) and integrating information on fine-scale fish movement and behaviour, is a logical extension. Complex agent-based models, with defined decision rules founded on knowledge of fish movement and behaviour, that can be integrated with hydrodynamic models, have great promise to inform fish passage at tidal barriers (Benson et al., 2021). These approaches can test barrier and fishway operation and modification in simulated environments to inform real-world application.

The provision of effective fish passage at tidal barriers is best achieved using structured adaptive management approaches where new knowledge informs refined barrier operation and fishway application (Birnie-Gauvin et al., 2017). Ultimately, site-specific, integrated approaches that include revised ('fish-friendly') operation of WCSs and navigation locks (if present), and application of technical fishways, are likely to be the most successful in promoting connectivity. These approaches are complementary and can result in fish passage being provided across a broader range of hydrological conditions than one approach alone and should be considered for existing and new tidal barriers. Nonetheless, all fish passage solutions should be set in a framework that includes subsequent monitoring to evaluate effectiveness and provide feedback to refine operation and approaches.

6.2. Reintroducing estuarine tidal flux

The reintroduction of tidal flux upstream of barriers has potential to promote biological connectivity and more natural hydrodynamics, and rehabilitate estuarine ecosystem function. In many cases, this is a contentious proposition given the initial objective of tidal barriers to limit saltwater incursion and flooding, and reduce potential risks to upstream potable water supplies, agriculture and urban areas. Nonetheless, the deliberate removal of tidal barriers that regulated local-scale estuarine wetlands, lakes and tributaries has occurred in many countries and there is substantial literature on subsequent changes to biotic assemblages (Wolters et al., 2005), including fishes (Lechêne et al., 2018; Sun et al., 2021). Removal has typically occurred when barriers have become redundant (e.g. changed agricultural practice) and when there has been little perceived risk to stakeholders. In other cases, restoration of tidal flux has occurred because of legislative requirements to compensate for losses of other estuarine habitat (Cox et al., 2006).

A concept underpinning tidal marsh restoration in the Scheldt estuary of Belgium and the Netherlands is the use of Controlled Reduced Tide (CRT) in Flood Control Areas (FCAs) (Meire et al., 2005; Maris et al., 2007). These are sections of estuarine floodplain bounded by man-made dikes, specifically a high outer dike and low inner dike, originally constructed for land reclamation and agriculture. Now, these reclaimed agricultural lands, repurposed as FCAs, aim to promote water storage capacity and dampen upstream tidal propagation, lessening flood risk of populated areas, while potentially promoting ecological benefit (Cox et al., 2006). CRT is applied in FCAs by using culverts and sluices to allow a dampened tidal regime, thus combining flood protection and ecological rehabilitation (Beauchard et al., 2011). CRT has been associated with positive ecological responses from various biota including vegetation (Jacobs et al., 2009), birds (Beauchard et al., 2011) and fish (Van Liefferinge et al., 2012). Thus, CRT appears to be an approach that could be applied at other leveed estuaries with lateral tidal barriers, where complete removal is not possible. An important caveat is that the design of WCSs thoroughly considers the bi-directional passage of fish and other aquatic biota.

Facilitating a degree of tidal fluxing at large run-of-river tide-excluding barriers in the world's densely populated estuaries, where they are critical to protect agriculture, infrastructure and human lives, is more problematic, but not impossible. The changed operation of the Haringvlietdam in the Netherlands is a key example. The project was first proposed in the 1990s and ultimately realized in 2018 (Buitenhuis and Dieperink, 2019). This has involved the upstream relocation of agricultural and potable water offtakes and implementation of monitoring programs to assess the influence of reintroduced tidal flux on upstream salinities and fish assemblages (Beeldman et al., 2018). This project demonstrates that through appropriate research, design, monitoring and nuanced structure operation, it may be possible to allow some tidal flux and at the same time, manage risk. Such approaches would be barrier specific, and a function of multiple factors, notably river discharge, tidal range, risk/safety and nature of the barrier. Hydrodynamic modelling presents a useful tool to run hypothetical scenarios and inform trials of tidal restoration at run-of-the river barriers.

Conceptually, promoting limited tidal flux of marine/brackish water upstream of tidal barriers would provide greater areas of estuarine habitat and improve fish passage. Importantly, this process would promote passage of species that do not readily use fishways, particularly many commercially important estuarine and marine-opportunist species that utilise selective tidal stream transport (Gibson, 2003). In most cases, restoration of tidal exchange upstream of barriers will require the collaboration of multiple stakeholders to manage risks and maximise benefits, but potentially represents a management action that could promote positive outcomes for fish populations at many of the world's tidal barriers.

6.3. Flow management to enhance estuarine habitats downstream

For regulated rivers, knowledge of the importance of freshwater flow regimes in driving physical, chemical and biological function has precipitated the development of 'environmental flow science' as a means to restore ecologically relevant aspects of natural flow regimes and support native biota, including fishes (Dyson et al., 2003). Many methods for determining the freshwater flow requirements of aquatic ecosystems have been devised and have progressed from simple hydrological frameworks to holistic ecosystem-based approaches (Tharme, 2003). Nonetheless, research and application of environmental flows have traditionally focused on riverine reaches, whilst application to estuarine environments has lagged, despite the importance of freshwater inputs being well understood (Alber, 2002; Adams, 2014). Notwithstanding, this is now a field of considerable interest and research globally (Chilton et al., 2021).

Fish are now a key biotic target of estuarine environmental flow programs in many countries, including Spain (Peñas et al., 2013), the United States (Reis et al., 2019), South Africa (Adams et al., 2016), Australia (Robins et al., 2005) and China (Sun et al., 2015). These are typically guided by species-specific or assemblage-based hydro-ecological relationships, and then prescribe freshwater flows to support these relationships (Van Niekerk et al., 2019). Three fundamental and related processes most commonly considered are: 1) promoting conditions suitable for residence (i.e. maintaining favourable salinity regimes); 2) facilitating critical life history processes (spawning and recruitment that is linked to hydrology and salinity); and 3) providing flow-related migratory cues. In systems with tidal barriers where upstream freshwater flow is diverted, there is a critical need to provide areas of estuarine refuge habitat downstream, particularly during times of drought (Baptista et al., 2010). Not only is this critical in influencing fish distribution, but also spawning, growth and recruitment (Jenkins et al., 2010); indeed, flow management that aims to support sustainable populations of long-lived species should consider multi-year flow regimes that include seasons of elevated freshwater discharge to enhance recruitment, year class strength and population resilience (Morrongiello et al., 2014).

The delivery of environmental flows has long been hampered by perceptions that freshwater discharged to estuaries and the ocean is wasted, particularly in regions where trade-offs between agricultural use and ecosystem function are complex (Gillson, 2011). Nonetheless, in an ecological sense, this view is outdated, with numerous studies demonstrating the principal role of freshwater flow on estuarine associated fish productivity (e.g. Lonergan, 1999; Kimmerer, 2002), along with numerous other ecosystem responses (Mallin et al., 1993). It can be argued that this is particularly important in estuaries with tidal barriers, where upstream transition of the river-estuarine-interface is precluded during times of low flow.

6.4. A long-term view on remediating tidal barriers

The rapid rates of global mean sea level rise currently being experienced and projected under climate change (IPCC, 2021) are likely to greatly increase saltwater incursion and storm-related flooding risks for many major cities and agricultural areas (Aerts et al., 2014). This situation has already reached a crisis point in some locations, necessitating new barriers and reinforcement of existing tidal barriers. In 2005, flooding in New Orleans associated with Hurricane Katrina prompted construction of the Greater New Orleans Hurricane and Storm Damage Reduction System, including the construction of tidal barriers and pumping stations, at a cost of $14 B USD (completed in 2018). Levee subsidence and greater rates of sea level rise than projected at project inception (2007) mean this system may cease to provide adequate risk reduction as early as 2023, necessitating further investment (https://www.scientificamerican.com/article/after-a-14-billion-upgrade-new-orleans-levees-are-sinking/).

Another notable case is the upgrade to the Afsluitdijk in the Netherlands, where works began in 2019 to heighten and strengthen the structure, and add pumping stations, at a cost of > €500 million. These examples illustrate the immense cost of conventional engineered coastal defenses in the face of a rapidly changing environment. Whilst these structures are critical to protect human lives and infrastructure in dense population centres, it is not necessary to view these methods as a template for broader coastal defense under climate change. In less populated areas projected to require flood defense, but currently undeveloped regarding conventional coastal protection, ecosystem-based flood defense and re-thinking of the position of river-estuarine interfaces may be more sustainable and cost-effective (Temmerman et al., 2013). While ecosystem-based flood defense is beyond the scope of this review, the interaction of fishes with tidal barriers and other flood defense actions is a necessary consideration of these broader approaches.

Protecting a static view of estuarine ecosystems in space and time may be unproductive in the long-term, leading to incrementally higher tidal barriers over time and increasingly challenging mitigation for fish passage, and freshwater flow and salinity management. A biogeographical view of the spatio-temporal flux of estuaries – for example, over the last 10,000 years – could provide a useful, if controversial, perspective. The consensus of most predictive modelling studies is for greater saline intrusion and a shift upstream of the freshwater–estuarine interface (Chen, 2017; Vargas et al., 2017), although there is regional and seasonal variability in estimates based on differences in estuary morphology and region-specific influences of climate change on rainfall and river flows (Robins et al., 2014). In many cases, these shifts in salinity are predicted to drive concomitant upstream shifts in the distribution of certain biotic communities (Little et al., 2017). Hydrodynamic modelling could assist planning for transitioning estuarine habitats upstream to new, currently freshwater locations. As such, the construction of new tidal barriers, where necessary, should be cognizant of such shifts and occur as far upstream as possible to limit coastal squeeze of estuarine habitats (Little, 2012). Furthermore, the reinforcement and heightening of existing barriers should be traded-off against the ecological and economic costs of relocation upstream. This is not without precedent, with Indonesia establishing a legal framework in January 2022 to facilitate the relocation of the nation's capital from the city of Jakarta (https://www.washingtonpost.com/world/2022/01/18/indonesia-capital-city-jakarta-borneo/), in part, due to land subsidence, sea level rise and increased flood risk (Erkens et al., 2015).

7. Conclusion

The ecologically sensitive management of tidal barriers is essential to sustain the function of estuaries and minimise impacts on ecosystem services. This is now a global conservation priority caused by the growth of coastal populations and increasing demands for freshwater, and exacerbated by climate change and sea level rise, and increasing impact of extreme weather events (Michener et al., 1997). It urgently requires both reactive and proactive attention.

Whilst approaches to remediation of tidal barriers are region, catchment and barrier-specific, we propose that in all cases, the objectives should be to minimise impacts on tidal flux, ensure connectivity for fish and promote natural estuarine ecosystem function. To achieve these objectives, we advocate generation of knowledge to understand: 1) estuary-specific hydrodynamics, water physico-chemistry and productivity, and predicted/observed changes under altered freshwater flow regimes, sea level rise, and barrier construction; 2) fish species-specific estuarine use and impacts of habitat loss and obstruction of movement; 3) the effectiveness of fish passage solutions; and 4) the benefits of remediation approaches on population dynamics. In many developed nations, the first three points are partially understood, but in developing nations, empirical data on fish movement, habitat use, and flow requirements are limited; for instance, the prevalence of diadromy in Asia's Mekong River, one of the world's great catchments, is just becoming apparent (Vu et al., 2020, 2022). Estuarine ecosystems in Asia, Africa, and South and Central America are increasingly subject to anthropogenic development and the above tenets can guide approaches to barrier construction, modification and operation. Furthermore, we advocate better global exchange of knowledge on impacts and remediation of tidal barriers; lessons learned in Europe, the United States and Australia could inform application in developing nations through established academic (e.g. Estuarine and Coastal Sciences Association), not-for profit (e.g. World Fish Migration Foundation) and international development agencies (e.g. Food and Agriculture Organisation of the United Nations).

Globally, tidal barriers have enabled increased agricultural production, freshwater storages close to coastal settlements, and protection of lives and assets; yet their impacts on ecosystems are undeniable. The challenge now is to integrate these functions with ecological restoration. In 2019, the United Nations General Assembly declared a 'Decade on Ecosystem Restoration' (2021–2030) with the aim of stimulating the restoration of degraded ecosystems worldwide in the face of key threats to biodiversity, and water and food security (e.g. climate change). This initiative provides a context for the restoration of estuaries impacted by tidal barriers at multiple scales.

CRediT authorship contribution statement

Christopher M. Bice: Writing – review & editing, Writing – original draft, Methodology, Investigation, Conceptualization. **Jeroen Huisman:** Writing – review & editing, Writing – original draft, Methodology. **Matthew E. Kimball:** Writing – review & editing, Writing – original draft, Methodology. **Martin Mallen-Cooper:** Writing – review & editing, Writing – original draft. **Brenton P. Zampatti:** Writing – review & editing, Writing – original draft, Conceptualization. **Bronwyn M. Gillanders:** Writing – review & editing, Writing – original draft.

Declaration of competing interest

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

Data availability

No data was used for the research described in the article.

Acknowledgements

We thank George Giatas, Peter-Paul Schollema, Martyn Lucas and an anonymous reviewer for input on earlier versions of this work. Thanks also to Josh Fredberg for assistance with mapping. This research did not receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecss.2023.108376.

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