# Chapter 7 Climate Change and Fishes in Estuaries

Bronwyn M. Gillanders, Matthew N. McMillan, Patrick Reis-Santos, Lee J. Baumgartner, Larry R. Brown<sup>†</sup>, John Conallin, Frederick V. Feyrer, Sofia Henriques, Nicola C. James, Andrés J. Jaureguizar, André L. M. Pessanha, Rita P. Vasconcelos, An V. Vu, Benjamin Walther, and Arif Wibowo

## 7.1 Introduction

Estuaries are among the most productive ecosystems worldwide but simultaneously among the most threatened by human activities (e.g. urbanisation, water diversion and fishing with consequent habitat loss, contamination and altered ecosystem properties), which can compromise their ecological functions and service provision (Wolanski et al. 2019). Superimposed on these impacts, estuaries face an array of additional direct and indirect impacts from global climate change. This is exacerbated by the multiplicity of processes through which climate change can affect estuaries, and in particular fish populations, across all levels of organisation from individuals to fish communities.

Global climate change is expected to lead to long-term, broad-scale changes in a suite of environmental drivers that pose emerging threats to biodiversity, ecosystem functioning as well as geomorphologic and biophysical processes in estuarine systems. Some changes will likely be swift whilst others will be pro-tracted but, in both cases, sound knowledge on estuarine fish communities and ecosystem functioning will be pivotal to recognise effects and predict future impacts in a timely manner. Only then will it be possible to safeguard the myriad ecological functions and services estuaries provide, in particular their nursery role which underpins the resilience and persistence of coastal fisheries for numerous valuable commercial species that depend on estuaries to complete their life cycles (Gillanders et al. 2011a, Hallett et al. 2017, Able et al. 2022, Blaber et al. 2022, Houde et al. 2022).

Overall, the impacts of climate change on estuarine fish communities and related fisheries raise many concerns, and whilst efforts are increasingly being taken worldwide to ensure long-term management of these key ecosystems, climate change may undermine current strategies. Moreover, the magnitude of impacts of climate change in estuaries will undoubtedly vary across biogeographic regions. Therefore, harnessing information to establish understanding from global to local scales of the vulnerabilities of estuaries and their fish communities will support informed adaptation and capacity building, enabling improved management and policy development (Sheaves et al. 2016, Williams et al. 2020).

In this chapter, we follow the functional guild classification of estuarine fish communities (Elliott et al. 2007, Potter et al. 2015) and provide an overview of the main drivers of change in estuarine systems,

<sup>&</sup>lt;sup>†</sup>Deceased

*Fish and Fisheries in Estuaries: A Global Perspective, Volume I*, First Edition. Edited by Alan K. Whitfield, Kenneth W. Able, Stephen J.M. Blaber, and Michael Elliott.

<sup>© 2022</sup> John Wiley & Sons Ltd. Published 2022 by John Wiley & Sons Ltd.

their expected causes and impacts on estuarine fish and fisheries. We start by presenting an analysis of the global, regional and local patterns of estuarine fish to identify drivers of taxonomic and functional diversity, and to predict how climate-induced changes may impact these ecosystems and their fish communities. Subsequently, we characterise the main environmental, climatic and biological stressors that will most likely impact estuarine fish and associated fisheries, and present a set of global case studies to illustrate the differences in expected impacts associated with different regions and types of estuaries and how these are not identical worldwide. Finally, we discuss key gaps in our knowledge and suggest pathways for future research to support estuarine ecosystem resilience, inform management and facilitate adaptation.

#### 7.2 Global, regional and local patterns

Climate change may lead to changes in the distributions and population densities of estuarine fish, but to understand how and where, it is necessary to understand the drivers behind patterns of estuarine fish at global, regional and local scales. The species composition of biological communities is influenced by a hierarchy of ecological mechanisms including dispersal limitation, environmental filtering and biotic interactions operating at global, regional and local scales (Figure 7.1) (Legendre et al. 2005, Rice 2005, Graham & Fine 2008, Barton et al. 2013). Dispersal limitation is the main predictor of fish species composition in estuaries at global and regional scales (this becomes evident when comparing composition between estuaries, Henriques et al. 2017b). Dispersal limitation results primarily from biogeographic barriers that isolate evolutionary processes, e.g. speciation and extinction, by limiting dispersal and thus giving rise to spatial differences in species composition among communities (Hillebrand & Blenckner 2002, Legendre et al. 2005, Barton et al. 2013). However, as spatial scales decrease, dispersal also becomes mediated by local connectivity and habitat patchiness (Hillebrand & Blenckner 2002).

Environmental filtering (e.g. via habitat suitability or physiological tolerance) begins to shape composition of biological communities at regional and local scales, whereby variation in environmental conditions drives spatial differences in niche exploitation (i.e. niche-based assembly rules) (Legendre et al. 2005, Barton et al. 2013). In comparison, at broader scales (i.e. between estuaries), temperature is an important filter for fish communities, and at finer scales (i.e. within estuaries) salinity becomes a very important species filter. Connectivity is also important in shaping biodiversity in estuaries by allowing for movement of species across ecotones that meet in estuaries, e.g. seawater-freshwater (Basset et al. 2013). At local scales, biotic interactions (e.g. predation, competition, mutualism) also play strong roles in predicting community composition (Legendre et al. 2005, Rice 2005); and these interactions become important for fish communities within estuaries (Henriques et al. 2017b). There are thus several axes of variation in composition of estuarine fish assemblages.

Meanwhile, species richness (i.e. number of species) of fish assemblages in estuaries is driven by a series of hierarchical factors (evident when comparing richness between estuaries): at broad scales, it is driven primarily by latitude (and associated temperature gradients) and biogeography, and also by connectivity with the marine ecosystem; whereas at finer regional scales, it is driven by ecosystem area. At local scales (i.e. within estuaries), longitudinal variation (i.e. from upstream to downstream) in species richness reflects salinity gradients and possibly habitat mosaics or patches (Figure 7.2) (Vasconcelos et al. 2015, Henriques et al. 2017b).

Species diversity tends to increase with habitat heterogeneity and patchiness (Soininen et al. 2007). Estuaries can have high species diversity due to their habitat heterogeneity and transitional nature (freshwater-seawater gradients), but at the same time this transitional nature restricts the species pool that



**Figure 7.1** Hierarchical order of ecological processes that operate at (a) global, (b) regional and (c) local scales to determine composition of estuarine fish assemblages (Henriques et al. 2017b, adapted from Hillebrand & Blenckner 2002).



**Figure 7.2** Variables associated with spatial patterns of species richness in estuarine fish at the global, regional and local scales. Predictors of species richness are spatially hierarchical: first, processes that define which species can colonise an estuary (history contingency, energy dynamics and productivity variables), and second factors that regulate processes of colonisation (habitat connectivity and species–area relationship variables). \* = variables not analysed due to high correlation with others. \*\* = variables with weak responses (Vasconcelos et al. 2015).

can colonise and inhabit these ecosystems (Whitfield et al. 2012). Species diversity takes on great significance in conservation management, particularly through the lens of environmental changes (e.g. climate change, fishing, invasive species), since ecosystem resilience (capacity to withstand and recover from stress) is positively associated with diversity via functional redundancy, providing a buffer against ecological collapse (McCann 2000, Ives & Carpenter 2007). Preserving fish species diversity in estuaries and maintaining or improving conditions that support diversity (e.g. connectivity with adjacent habitats) are thus desirable to help buffer estuarine systems against the effects of a changing climate. Accurate assessment of estuarine biodiversity patterns at different spatial scales is therefore important to understand real patterns of change in space and time and to monitor resistance to change and enable resilience of ecological communities. To this end, the remainder of this section will examine the drivers of estuarine fish diversity at global, regional and local scales in detail.

## 7.2.1 Predictors of fish taxonomic diversity at global and regional scales

At global and regional scales, composition of fish assemblages in estuaries is shaped by assembly processes, namely dispersal limitation and environmental filtering. Patterns of taxonomic composition emerge at the global scale, i.e. among marine biogeographical realms. Large marine regions share evolutionary histories and exposure to common drivers (i.e. mainly water temperature and historical isolation), yield high levels of endemism and have internally coherent biotas at high taxonomic levels (Spalding et al. 2007). Historical isolation is fundamental to the evolutionary influence on taxonomic diversity and is caused by the occurrence of barriers between biogeographic regions. Hard barriers (e.g. isthmuses that separate ocean basins) physically constrain geographic dispersal of taxa, whereas soft barriers (e.g. depth, currents or abrupt environmental clines such as temperature and dissolved oxygen levels that often sag in areas of high turbidity) limit dispersal but may be more temporally diffuse, allowing for greater vicariance in fish assemblages among neighbouring regions (Cowman & Bellwood 2013).

A global comparison of beta diversity (variability in species compositions: Anderson et al. 2006) found that estuaries tended to cluster together with other estuaries in the same marine biogeographic realm, despite some similarity with estuaries from neighbouring marine realms (some even despite hard barriers) and some dissimilarity generated by soft barriers (Henriques et al. 2017b). For example, estuaries along the east coast of the Americas in the Temperate Northern Atlantic, Temperate South American and Tropical Atlantic marine biogeographic realms clustered together with estuaries from east coast of Africa, as did estuaries in the Western and Central Indo-Pacific marine biogeographic realms. Seven major estuarine biogeographical realms therefore emerged: Indo-Pacific, Tropical East Pacific, Cold-Temperate North America, Temperate Australasia, Warm-Temperate West Atlantic, East Temperate North Atlantic and Temperate Southern Africa. The taxonomic similarity of estuaries within these realms was largely driven by marine biogeography, i.e. evolutionary processes associated with historical isolation and dispersal, but could not be entirely explained on this basis (Henriques et al. 2017b). At a regional scale, Whitfield (2005b) showed the relevance of such barriers, namely for the biogeography of fish in African estuaries: the closure of the Tethyan Gateway 12 million years ago in the north, and the development of the Benguela upwelling system in the south imposed a limitation to the movement and mixture between the east and west across both the northern and southern extremities of the African continent and promoted the current differentiation of faunas.

In addition to marine biogeography, the main drivers of taxonomic composition and richness in fish assemblages in estuaries are primarily the global latitude and temperature gradient (and global primary productivity gradient though much less important), local connectivity with marine ecosystems and secondarily local habitat size and suitability (Vasconcelos et al. 2015, Henriques et al. 2017b). Temperature influences the viable range within which physiological tolerances allow species to survive (Wiens 2011). Fish species richness thus tends to decline in estuaries towards the poles compared to estuaries at lower latitudes (with higher temperatures) (Blaber 2008), reflecting a temperature-latitude gradient recognised to influence species richness across all biological groups (Gaston 2000, Whittaker et al. 2001, Willig et al. 2003). This pattern is likely related to rates of speciation and extinction that vary along latitudinal gradients and holds true for marine (Tittensor et al. 2010), freshwater (Tisseuil et al. 2013) and estuarine fish assemblages (Vasconcelos et al. 2015). An example of temperature filtering is provided by the higher affinity (lower beta diversity) of estuaries in the high-latitude east American coast with estuaries from the high-latitude west American coast (although separated by the Panama Isthmus more than three million years ago) than with estuaries on the east American coast immediately to the south from which they are separated by a thermal soft barrier (Henriques et al. 2017b). The same was shown at a regional scale for estuaries in sub-Saharan Africa, where estuaries of adjacent biogeographical regions are not always similar as a result of temperature differences (Whitfield 2005b). Overall, this is indicative of the power of temperature to prevent colonisation of cold-temperate habitats by tropical species (Zapata & Ross Robertson 2007, Briggs & Bowen 2012).

While biogeography, latitude and temperature play important roles in influencing fish species richness (Vasconcelos et al. 2015) and composition (Henriques et al. 2017b) in estuaries, other factors also influence differences in these characteristics at global and regional scales, though to lesser extents. At these scales, estuaries with higher terrestrial primary productivity (a proxy of catchment nutrient input) also tend to have increased species richness (Vasconcelos et al. 2015). Terrestrial primary productivity is a proxy for energy availability, since it underpins energetic dynamics in rivers and estuaries where catchment nutrient inputs form the base of food webs. This relationship between primary productivity and species richness has also been observed in freshwater systems (Oberdoff et al. 1995) as well as among birds (Wright 1983) and mammals (Wylie & Currie 1993).

#### 7.2.2 Predictors of fish taxonomic diversity at local scales

As transition zones between marine and freshwater systems, estuaries attract both marine and freshwater fish species, with relatively few permanent estuarine-resident species (Blaber 2008, Mann 2009, Potter et al. 2015). Marine and freshwater species colonise estuaries to the extent their physiological tolerances permit, while diadromous species use estuaries periodically when moving between freshwater and marine habitats. In fact, most fish in estuaries are marine species pushing towards freshwater and freshwater species pushing towards the sea, both approaching the limits of their salinity tolerances (Elliott & Quintino 2007, Vasconcelos et al. 2011). Connectivity with adjacent habitats is therefore a key driver of estuaries represent a stressor for some species, for others, that are able to capitalise on these conditions, salinity variations represent a subsidy, yielding reduced competition, and allowing them to achieve high population densities in estuarine environments (Elliott & Quintino 2007).

The spatial boundaries of species physiological tolerances are highly changeable because estuaries are prone to changes in conditions at various spatial and temporal scales, e.g. salinity or temperature may change daily with tidal cycles or seasonally with freshwater inflows (and estuary mouth closures) (Basset et al. 2013). This relative instability leads to frequent turnover in compositions and distributions of estuarine fish communities in time and space (Sosa-Lopez & Mouillot 2007, Villéger et al. 2008) but in a predictable manner as a result of changes in the main drivers (temperature, salinity, river flow and connectivity with adjacent ecosystems). Environmental instability does not favour the evolution of specialists (Whitfield 1994b), which are relatively lacking in estuaries compared to more stable marine environments. It follows that estuaries favour robust euryoecious species with wide environmental tolerances (especially euryhaline ones) that can exploit a wide range of conditions; species richness in estuaries is therefore lower than in adjacent marine habitats (Wallace et al. 1984, Martino & Able 2003, Barletta et al. 2010). As a result, estuarine fish communities are dominated by few but persistent 'core species' with robust physiological tolerances that dominate in abundance, but experience periodic intrusions by less robust 'occasional species' that may exploit estuaries when conditions allow (Magurran & Henderson 2003).

Species composition of estuarine fish assemblages are driven by physiological constraints associated with salinity. There has been a wide discussion on whether estuaries can be considered as an ecotone [i.e. area of frequent and rapid environmental and community change e.g. Reid (1961)], or ecocline (i.e. area of gradual, continuous change in species composition across an environmental gradient), or even a dual ecoclinescape (with a marine cline and a freshwater one, and not true estuarine species) (Attrill & Rundle 2002). According to Whitfield and Elliott (2011), there is no consensus that the ends of an estuary represent ecotones, i.e. that there is one ecotone between the river and the estuary, and another between the estuary and the sea. This discussion on how aquatic species diversity changes from freshwater to more marine areas has also involved multiple proposals and rebuttals of suitable models. Early on, the seminal Remane diagram (Remane 1934) suggested that marine and freshwater components comprise equal number of species, both decreasing with the progression into transitional waters and the space also occupied by a small but significant number of 'brackish' species. Recently, Whitfield et al. (2012) proposed a revised conceptual species diversity change model which is more suited to estuaries worldwide and aligned with the Venice classification (Figure 7.3). In this model: freshwater species are fewer than marine and only a few extend to mesohaline areas and below; marine species extend to all areas (and dominate all except oligohaline); estuarine/brackish species can occur in all areas including freshwater, their diversity highest in mesohaline and polyhaline waters; and finally diversity decreases above salinity 40 and plummets above 50, with few species able to occupy salinities 0-100. Regardless of the landscape/diversity model, the



**Figure 7.3** Proposed conceptual model for estuarine biodiversity (species) changes covering the salinity continuum from freshwater to hyperhaline conditions, proposed by Whitfield et al. (2012).

species (and number of species) present and their longitudinal (upstream-downstream) distributions within estuaries are likely to vary with key conditions, e.g. fluctuations in salinity gradients, which in turn are heavily influenced by events such as freshwater inflows and connectivity of the estuary with the adjacent marine ecosystem.

Connectivity between estuaries and marine ecosystems is an important driver of changes in species richness and composition, and can be measured via proxies such as estuary mouth morphology (open versus closed, including frequency, timing and duration of opening; Whitfield 1999), mouth width (Nicolas et al. 2010, Vasconcelos et al. 2015) and tidal range. Generally speaking, permanently open estuaries have greater species richness than temporarily closed estuaries due to their greater connectivity with adjacent habitats and less disrupted salinity gradients (Vasconcelos et al. 2011). Estuaries prone to mouth closure (temporarily open-closed estuaries) usually occur in some temperate regions like southern Australia and South Africa but also other regions. Such estuaries typically have smaller areas, mouth widths and drainage basins than permanently open estuaries (Vasconcelos et al. 2015). As such, fish assemblages of estuaries prone to mouth closure may have greater contributions from species that complete their life cycles in estuaries compared to estuaries in other regions where marine species dominate (Potter & Hyndes 1999, Hoeksema et al. 2009). The influence of estuary-sea connectivity on community richness and composition has also been shown at a more regional/local scale among 10 South African estuaries, with the degree of similarity of fish community composition largely defined by estuary mouth status and estuary size (Vorwerk et al. 2003).

In estuaries with periodically or intermittently closed mouths, freshwater flows are often essential to overcome physical barriers to connectivity between estuarine and marine habitats, e.g. through overtopping and erosion of sand berm at the mouth. In such cases, peak immigration by marine species is likely to follow dissipation of the river flooding and a resumption of tidal processes (Whitfield 1999). In barrier island systems, such as the northern Gulf of Mexico, physical disturbance, sedimentation and storm surges associated with major tropical cyclones can open or close inlets to alter connections of enclosed estuaries with the adjacent ocean (Conner et al. 1989, Stone et al. 2004). Storm disturbance is typically associated with punctuated increases of local rainfall and pulses of freshwater inflow, such that the physical and water quality impacts of a storm event can be simultaneous. However, it is the connectivity created by such events that facilitates fish movements into and out of such coastal systems.

Freshwater flows are also important in permanently open estuaries where they can drive immigration among marine species. Riverine plumes trigger migrations of early life stages of marine species from spawning grounds to settlement areas within estuaries (Sullivan et al. 2006, James et al. 2007, Vinagre et al. 2007). For example, in South Africa, ichthyoplankton density and abundance of larval marine species were greatest in estuaries with greater freshwater inputs despite similar mouth morphologies, channel width and depth (Whitfield 1994a). Moreover, freshwater inflow can trigger breeding migrations in both marine migrants and estuarine-resident species (Walsh et al. 2013). The ecological structure of estuaries may also be altered by freshwater inflow. Nutrient-rich inflow stimulates planktonic production and facilitates pelagic food webs in estuaries with moderate inflow and sufficient water residence time, whereas morphologically similar estuaries with low inflow rely more on benthic productivity by macrophytes and therefore tend to favour more demersal marine fishes (Schlacher & Wooldridge 1996).

Tidal range also influences connectivity between estuaries and marine ecosystems and varies among estuaries depending on hydrodynamic responses to astronomic tidal forcing (from micro- to meso- and macrotidal, respectively <2m, 2–4m, >4m) resulting in major differences in hydrodynamics of estuarine systems (Tweedley et al. 2016). Secondly, tidal height in an estuarine system generally varies cyclically, i.e. daily (or approximately daily) and also within the synodic month, and influences movement of fish and habitat use within estuaries (Vinagre et al. 2006, Næsje et al. 2012).

Freshwater inflow is often considered the principal cause of seasonal variation in estuarine physicochemical conditions (Montagna & Kalke 1992, Sklar & Browder 1998, Kimmerer 2002b). In temperate estuaries, water temperature is a strong contributor (Able et al. 2022). Among other estuarine conditions such as dissolved oxygen, nutrient availability and turbidity, salinity is strongly influenced by this freshwater flow, which becomes a key driver of biotic and abiotic structuring, and thus of species composition and richness of fish assemblages within estuaries (Loneragan 1999, Garcia et al. 2001, Whitfield 2005a).

Changes to freshwater flow may be either natural, e.g. seasonal variations in catchment run-off due to precipitation, or artificial, e.g. through the construction of upstream dams or water diversions. Moreover, freshwater inflow tends to vary among estuaries as a function of catchment size and climate. Natural influences on freshwater flow include extreme episodic events such as floods or droughts and are thus likely to be influenced by climate change. Low flows associated with droughts can accelerate formation or prolong persistence of physical barriers such as mouth-restricting sandbars, reducing connectivity with marine habitats (Cyrus et al. 2011). Prolonged mouth closures may cause marine migrant species to decline, particularly juvenile age classes, whose absence persists until opening of the mouth re-establishes connectivity with spawners or larvae from marine habitats (Bennett et al. 1985, Martin et al. 1992, Cyrus et al. 2011). In some cases, mouth closures may cause hypersalinity due to evaporation, leading to dominance of species with high salinity tolerances (Cyrus & Vivier 2006, Potter et al. 2010).

Water residency times and flushing rates may influence estuarine productivity and water quality with flow on effects on ecological health and community structures. In particular, poor flushing of water bodies and long water residence times can lead to stratification of the water column and accumulation of nutrients driving algal blooms that can increase turbidity and decrease availability of dissolved oxygen (Vitousek et al. 1997, Dettmann 2001). Freshwater inflows (advection) have greater capacity to flush estuarine water masses than tidal influence (dispersion), with residence times decreasing in relation to flow rate (Miller & McPherson 1991). Overall, inflows and tidal ranges in estuaries are governed by seasonal and lunar cycles.

However, climate change may alter residence times and flushing rates, e.g. through changes in timing and intensity of rainfall affecting inflows or more severe and frequent storm surges affecting tidal height.

Estuarine morphology or anthropogenic constructions may also limit upstream connectivity during drought periods. For instance, in the normally closed Culham Inlet in southwestern Australia, falling water levels caused hypersaline conditions and exposed a rock bar, thus preventing movement to less saline upstream areas by black bream *Acanthopagrus butcheri* and ultimately causing a mass mortality of this species (Hoeksema et al. 2006). Similarly, construction of tidal barrages and reduced river inflows into the Murray Estuary of southern Australia prevented freshwater mixing, resulting in hypersalinity, dominance of marine species, inhibited recruitment of catadromous species and reductions in species richness and diversity (Zampatti et al. 2010). Even where estuaries remain open, drought conditions may cause an upstream shift in salinity levels through evaporation and/or seawater intrusion, resulting in increased abundance of marine species at the expense of freshwater species (Martinho et al. 2007).

In contrast to droughts, flood events are often beneficial to estuarine species richness since they deliver large nutrient inputs that stimulate productivity and promote connectivity between estuaries and adjacent habitats, e.g. by enhancing access through estuarine entrances and by inundating wetlands. However, extreme flood events may affect fish assemblages negatively, e.g. by flushing food out from estuarine habitats that can reduce carrying capacity and favour species with competitive advantages. This may be particularly true for small estuaries where floods reduce or temporarily eliminate seawater penetration, lead to very short water residency times and limit development of planktonic food sources, thus becoming less suitable for marine fishes (Wooldridge 1999, Whitfield & Paterson 2003). Floods can likewise alter recruitment processes in estuaries by creating physical barriers in the form of strong flows that obstruct movement or influencing habitat use by changing salinity and/or turbidity regimes (Loneragan 1999). Climatic oscillations may also influence freshwater flows and thus estuarine fish communities. For example, in wet years, the Tagus Estuary in Portugal was dominated by freshwater species, whereas dry years led to dominance by marine migrants and catadromous species (Costa et al. 2007). Similarly, high rainfalls associated with El Niño oscillations led to dominance of freshwater species in the Patos Lagoon estuary in Brazil (Garcia et al. 2003).

The main sources of artificial alteration to freshwater flows are manmade constructions such as dams and flood mitigation or irrigation works that limit or remove natural flows from upstream catchments. Diversion or retention of flows can alter seasonal flow patterns such that they may no longer coincide with breeding cycles or their effect as reproductive cues becomes limited due to decreased magnitude or frequency. Reduced flows associated with upstream water storage or abstraction may also lead to reductions in the area or quality of estuarine habitats, particularly shallow marshlands that often function as important nursery areas for marine fish species, and may thus lead to reduced recruitment in species reliant on such areas (Limburg & Waldman 2009). Alternatively, regulated releases of water from upstream dams may shift or alter the area of estuarine habitats over time. For example, in the Guadalquivir Estuary in Spain, releases of water from a dam 110 km upstream caused expansion and contraction of estuarine nursery habitats used by marine species as salinity gradients shifted up- or downstream (Fernández-Delgado et al. 2007). Similarly, construction of large dams and reduced freshwater flows have led to salinity gradients shifting upstream and colonisation by marine species of areas previously dominated by freshwater species (Chícharo et al. 2006). Additionally, construction of barriers can block migration pathways for diadromous species unless carefully designed to incorporate effective fishways or similar devices (Drinkwater & Frank 1994, Costa et al. 2007). Such barriers to reproductive migrations may even lead to extirpations of diadromous species cut off from spawning areas, removing them from local fish assemblages (Limburg & Waldman 2009).

Given this suite of potential negative effects on estuarine fish assemblages caused by river regulation, consideration should be given to setting aside water allowances for use in managed ecological flows that stimulate productivity or reproductive events in downstream estuaries, i.e. compensation flows required to

maintain the ecology of estuaries (Chícharo et al. 2006, Richter & Thomas 2007). In some regions, legislative requirements to include estuarine health in decisions regarding environmental flow allocations has prompted a range of strategies to assess inflow impacts, including fishery productivity (Robins et al. 2005, Adams 2014). However, a lack of knowledge about the full range of impacts of reduced flow hinders this decision-making, particularly in the face of increased drought events in arid systems.

In addition to salinity, river flow and connectivity with the adjacent marine and freshwater ecosystems (discussed above), other factors may play lesser roles in influencing estuarine fish assemblage composition and richness at local scales, e.g. estuary size, shape, depth, sediment type, dissolved oxygen, temperature, food availability and turbidity (Elliott & Quintino 2007, Blaber 2008). Ecosystem size is also an important predictor of species richness in estuaries at the regional scale (Nicolas et al. 2010), whereby larger systems may provide more scope for heterogeneous habitats and thus more ecological niches and species that fill them (Wootton 1990). Estuary size has also been linked with greater contributions to fish assemblages by marine species (Henriques et al. 2017b). Estuaries with larger area also tend to have greater (and more stable) connectivity with the marine ecosystem and larger drainage basins, and the effects of these different factors are linked.

At very fine scales, fish assemblages are influenced by within-estuary distribution patterns affected by species-specific habitat preferences. The main estuarine habitat types involved at this level of biotic structuring are mudflats, seagrass beds and saltmarshes in temperate estuaries along with mangroves in tropical estuaries. These vegetated habitat types are particularly associated with juvenile age classes as nursery habitats (Orth et al. 1984, Costa & Bruxelas 1989). The main value of seagrass beds and mangroves as nursery habitats appears to be the shelter and high survivorship they offer; growth rates are faster around reefs, yet the former attract greater densities of juveniles (Nagelkerken 2009). Despite the emphasis on these habitat types, estuarine habitats are more diverse and some are frequently unrecognised (see Able et al. 2022).

Since marine migrant species that often form the major contribution to estuarine fish assemblages are particularly drawn to estuaries due to their provision of nursery habitats, any degradation thereto may impact abundance or richness of marine species and is likely to alter fish assemblages. In addition to nursery functions, habitat type can affect food availability and thus species distributions. For example, in a number of European estuaries surveyed, percentage of mud substrate was an important predictor of total fish abundance, indicating a preference by many species for mudflat habitats as foraging grounds (Selleslagh et al. 2009). For instance, Sheaves and Johnston (2009) found that mangrove area and sediment type were the most important estuary-level predictors of fish assemblages in tropical estuaries, with occurrence of mangrove-associated species increasing with mangrove area and benthic feeders favouring coarse substrates.

Finally, other factors also influence richness and composition of fish communities in estuaries. For example, in a survey of temperate and subtropical estuaries, Harrison and Whitfield (2006) found that, along with salinity, temperature was a primary determinant of fish assemblages with turbidity and depth also affecting species composition, while relative abundances of various species were also affected by turbidity (mostly positively) and dissolved oxygen (negatively). It can thus be seen that while salinity (influenced by freshwater flow and connectivity with the marine ecosystem) is the main predictor of estuarine fish assemblages at the local scale, a suite of other factors contribute to environmental filtering and biotic interactions that ultimately determine species composition within estuaries.

## 7.2.3 Predictors of fish functional diversity at global, regional and local scales

Whereas taxonomic diversity refers to the range of organisms that occur in an ecosystem, functional diversity is the range of traits contributed by different species influencing how an ecosystem functions (Tilman 2001). Species with similar traits form functional groups whose members are likely to affect ecosystem functioning

in similar ways (Blondel 2003). Examples of such traits include diet (e.g. herbivores, omnivores, invertebrate feeders, piscivores etc.), salinity preference (freshwater, brackish and marine) and body size.

Biogeography is the major driver of global patterns of taxonomic diversity (species composition and richness) in estuaries (Vasconcelos et al. 2015, Henriques et al. 2017b), but also explains a large proportion of worldwide variation in functional diversity, together with other drivers that have important and complex influences (Figure 7.4) (Henriques et al. 2017a). Functional diversity of fish assemblages in estuaries is primarily regulated by biogeography followed by two main environmental gradients: firstly, a temperature-latitude gradient and secondly an estuary size/hydrology gradient related to the connectivity of estuaries with the adjacent marine ecosystem (Henriques et al. 2017a). Both gradients influence species richness and composition (Vasconcelos et al. 2015, Henriques et al. 2017b) and these patterns seem to be determined by



**Figure 7.4** Summary of drivers of functional trait diversity in estuarine fish assemblages. Traits considered were fish salinity preference, diet, body size, depth of distribution and life span. Drivers considered included biogeography (each dot is an estuary in the study), sea surface temperature (at estuary mouth), catchment terrestrial productivity, width of adjacent continental shelf, marine productivity (chlorophyll *a*), tidal range, estuary type (open versus temporarily closed), estuary size and salinity type (regular to hypersaline). The grey arrows represent the sense of the trend and should be read in the following manner: as an example – fish body size, depth of distribution and life span tend to be higher in estuaries with lower temperature than in estuaries with higher temperature (Henriques et al. 2017b).

the functional role of species within ecosystems given the stronger influence of environmental drivers on trait patterns than on taxonomic patterns (Henriques et al. 2017a).

The primary effect of biogeography on trait patterns is that areas with similar geographical features (evolutionary history) and environmental features (i.e. climatic conditions) tend to support species with similar life strategies and adaptations (e.g. body size, longevity, fecundity) (Bender et al. 2013, Luiz et al. 2013). Similarly, temperature-latitude and estuary size/hydrology gradients likely stem from species traits (adaptations) and latitudinal distributions of estuaries with specific features (e.g. tidal range, estuary area and type – open versus temporarily closed). For instance, smaller estuaries are rare in cooler regions and tend to have limited and less enduring connectivity with marine ecosystems. In fact, trait patterns and trait-environmental relationships are likely generated by multiple processes linked to physiological constraints (i.e. associated with temperature and salinity) in combination with habitat availability (i.e. area and geographical distribution) and connectivity with marine ecosystems (Henriques et al. 2017a).

At the global scale, marine species dominate fish assemblages in estuaries and their proportions increase in open estuaries with large estuary area and high connectivity to adjacent marine areas. In contrast, proportions of freshwater species decrease with system size and connectivity, and these and brackish species increase with temperature (i.e. increase along the north-south temperature gradient), while diadromous species decrease (Henriques et al. 2017a).

In terms of diet, invertebrate feeders dominate estuarine fish assemblages globally in response to the prevalence of small detrivorous crustaceans in estuarine food webs that link allochtonous inputs to in situ productivity accessible to fishes (Costa & Elliott 1991). The proportions of detrivores, omnivores and herbivores tends to increase towards the equator and decrease with estuary size and connectivity, while macrocarnivores (i.e. those that feed on large invertebrates and fish) and planktivores tend to show the opposite pattern, perhaps due to the greater influence of marine species these drivers attract (Henriques et al. 2017a). Similarly, these drivers lead to increases in body size and depth distribution towards the poles and in estuaries with larger connectivity and area (again suggesting a positive link with marine species), whereas most estuaries are dominated by species with small-to-medium body sizes and shallow-to-medium depth distributions. Lifespans of estuarine fishes are dominated by medium- to long-lived species with hypersaline estuaries favouring very long-lived species. These body size patterns (and related traits of depth distribution, lifespan) probably result from multiple causal processes linked with physiological constraints. For example, at higher temperatures, oxygen concentration is lower and implies higher respiration rate and energy loss, therefore smaller species have a physiological advantage due to their shorter diffusion paths and lower energy requirements (Huston & Wolverton 2011, Edeline et al. 2013). Likewise, the similarity between patterns of diet (i.e. macrocarnivores favoured in cooler regions versus omnivores, herbivores and detritivores in warmer regions) and body size (i.e. increase of body size towards the poles) can be due to metabolic, physiological and ecological constraints. Metabolic rates decrease with temperature and plants and detritus are digested more efficiently in warmer waters (i.e. better enzymatic performance) suggesting that detrivores, omnivores and herbivores might have difficulty meeting metabolic demands at cooler temperatures (Kulbicki et al. 2015, Henriques et al. 2017a). Finally, relative to other estuaries, hypersaline estuaries have greater proportions of species with preference for brackish ecosystems as well as omnivores (both more adaptable to changing and extreme environmental conditions).

Species richness and the representation of functional traits in faunal communities can impart resilience and stability on ecosystems, buffering them against stress, although the resistance and resilience of constituent species and biotic interactions also contribute (Mouillot et al. 2013). Understanding the drivers behind taxonomic diversity and functional structure of ecosystems and predicting how changes are likely to affect them are therefore valuable to conservation management. By incorporating such knowledge into management planning, it may be possible to enhance the resilience of ecosystems to climate change and associated stressors such as altered flow regimes, or to attempt to offset the impacts of particular stressors by reducing the impacts of others that are more easily controlled. Conservation management should therefore take a tiered approach to estuary management that is alert to the different drivers of biodiversity and functional structure at different spatial scales and which, if addressed appropriately, may offer assistance in maintaining the resilience of estuarine habitats in the face of human-induced change such as climate change.

## 7.3 Potential impacts of environmental/climate stressors on estuarine fish

Estuaries are naturally complex, dynamic and unstable environments as a result of the mixture of fresh and marine waters, with physico-chemical conditions oscillating naturally at daily, weekly, lunar, seasonal, equinoctial and yearly scales due to tidal cycles and changes in freshwater input, even linked to large-scale climate oscillations such as El Niño/La Niña and the North Atlantic Oscillation. Effects of climate change over time are expected to lead to shifts in the magnitude of natural oscillation patterns and to deviations in long-term mean conditions. In particular, the effects of climate change on estuaries will likely stem from changes to freshwater flow, salinity intrusion, temperature, dissolved oxygen, as well as modifications to entrance channel opening, and spreading of disease (Gillanders et al. 2011a, Robins et al. 2016). In addition, relative sea level rise, which can lead to coastal squeeze, will eliminate habitats such as salt marshes (Crosby et al. 2016, Roman 2017, Schuerch et al. 2018) especially in micro-tidal estuaries (Kearney & Turner 2016). Therefore, climate change has not only the potential to alter directly and indirectly the hydro-geomorphology but also the physico-chemical conditions of estuarine systems beyond the tolerance thresholds of their current estuarine biota, with expected major impacts on ecological processes, and ecosystem functions and services. Overall, the potential impacts of climate change on estuarine fishes are diverse and span all levels of organisation from individuals to fish assemblages, and are compounded by the diversity of processes over which climate change can directly and indirectly impact estuaries (Scavia et al. 2002, Roessig et al. 2004, Gillanders et al. 2011a, Cloern et al. 2016, Hallett et al. 2017) (Figure 7.5).

If greenhouse gas emissions continue unabated, impacts of climate change on estuarine systems are likely to continue to grow in the future, concomitant with the predicted increase in the intensity and frequency of extreme climatic events (including, floods, droughts, heatwaves) (Stott 2016). These are major threats as extreme events can exacerbate climate change disturbances in estuaries and trigger potential turning points in ecosystem structure and functioning (Martinho et al. 2007, Wetz & Yoskowitz 2013, Possamai et al. 2018). Ultimately, it is paramount to understand how future conditions are likely to disrupt estuarine environments, and identify the drivers and implications of future changes to estuarine fishes. Doing so will underpin future management strategies as well as the development of legal policies that can be applied to increase the resilience of estuarine systems (Saul et al. 2016; Blaber et al. 2022).

#### 7.3.1 Salinity and freshwater flow impacts

Salinity is a key factor determining fish species distribution within an estuary, so much so that one of the essential classification schemes for fishes in estuaries is based on species tolerance to salinity (Elliott et al. 2007). At the individual level, salinity influences osmoregulation, impacts metabolism, and triggers physiological and behavioural responses in fish (Smyth & Elliott 2016). Therefore, abrupt changes in salinity (due for instance to rapid changes in freshwater flow, or long-term evaporative processes) may lead to



**Figure 7.5** Potential links between climatic variability and anthropogenic influences on estuarine environments (adapted from Gillanders et al. 2011a).

relocation, but also to mortality if the magnitude of change and osmoregulatory stress surpasses physiological tolerance levels, and fish are unable to move out of an area quickly enough. For example, massive mortalities of fish in south-western Australia (estimated at 1.3 million black bream *A. butcheri*) occurred when salinity spiked and low water levels prevented fish from relocating (Hoeksema et al. 2006). Ultimately, species persistence and survival under changing conditions in estuaries is conditional on species relocating to suitable conditions, with individuals moving throughout an estuary in response to changing conditions (Garcia et al. 2003, Childs et al. 2008). Thus, the most likely impacts of climate change on estuarine fish will be correlated with the stability or continued availability of suitable habitats, and maintenance of appropriate conditions for the reproductive success of fish belonging to the estuarine, marine migrant and diadromous guilds.

In estuaries located in biogeographic regions expected to become warmer and drier, rising salinities expectedly will place oligohaline and mesohaline fishes under greater threat, and displace these species and other freshwater stragglers further upstream (where possible, as other factors such as lack of suitable habitat, dams or other anthropogenic barriers will play a key role, and are intertwined with species persistence in estuarine environments) (Gillanders et al. 2011a, Vasconcelos et al. 2011, Robins et al. 2016, Hallett et al. 2017). Freshwater migrants that rely on estuaries to complete their life cycles will likely also be affected and their occurrence will be linked among others to freshwater input (Sheaves & Johnston 2008, Morrongiello et al. 2011, James et al. 2013).

In contrast, euryhaline and marine stragglers will probably be able to penetrate farther into the estuaries, with several studies worldwide documenting the depletion of freshwater fish and surge in marine affinity species in temperate estuaries associated with decreases in freshwater flow and increased salinity, particularly under drought conditions (Whitfield 2005a, Martinho et al. 2007, Valesini et al. 2013, Garcia-Seoane et al. 2016, Potter et al. 2016). However, a major threat to estuaries with warmer and drier climate is the occurrence of hypersaline conditions as a consequence of reduced freshwater inputs and increased evaporation (Hoeksema et al. 2006, Gillanders et al. 2011a, James et al. 2013, Hallett et al. 2017, Tweedley et al. 2019). Hypersalinity is overwhelmingly associated with negative impacts on estuarine fish communities, with dramatic decreases in community structure and composition, potentially leading to species extirpation, as very few species can tolerate the harsh osmoregulatory conditions (Young & Potter 2002, Zampatti et al. 2010, Gillanders et al. 2011a, Wedderburn et al. 2016, Wooldridge et al. 2016, Hallett et al. 2017).

Whilst an increase in species richness may be expected with estuarine 'marinisation' due mainly to the increased presence and dispersion of marine stragglers, increases in salinity have also led to decreases in estuarine biodiversity linked to reductions in diadromous and freshwater species, as well as associated with the establishment of hypersaline conditions (Zampatti et al. 2010, James et al. 2013, Hallett et al. 2017). Several studies worldwide, including over the long term, have demonstrated how fish guilds along the estuarine salinity gradient (e.g. oligo-, meso-, poli-, euryhaline) respond to different features of climate change that impact river flow and salinity gradients (e.g. Elliott et al. 1990, James et al. 2013, Baptista et al. 2015, Feyrer et al. 2015, Hallett et al. 2017, Possamai et al. 2018), and select case studies by biogeographic region are explored in further detail below (see Section 7.5 for more details).

Variations in freshwater flow in both directions (i.e. increases and decreases) are expected throughout different global biogeographic regions due to the effects of climate change (e.g. linked to changes in precipitation, evapotranspiration or ground recharge/storage capacity). Freshwater inflows to estuaries, from both surface (i.e. rivers, tributaries) as well as groundwater sources, are an integrated product of the hydroand meteorological processes acting upon a catchment. Combined with other interacting forcing factors (e.g. entrance channel modifications and sea level rise), variations in freshwater flow will primarily affect the estuarine salinity gradient (but also turbidity, stoichiometry, dissolved oxygen, or estuarine and coastal productivity or upwelling), and probably affect species and trait composition (Kimmerer 2002a, Garcia et al. 2003, Martinho et al. 2007, Vasconcelos et al. 2011, Jaureguizar et al. 2016) and functional changes in fish assemblages according to species (and different life stages) salinity tolerances (Baptista et al. 2015, Feyrer et al. 2015, Vasconcelos et al. 2015).

A key area of concern regarding the impacts of river flow and salinity relates to the alterations in estuarine fish population dynamics and assemblages, brought about by extreme weather conditions (e.g. droughts, floods) which likely represent future conditions under a changing climate (Martinho et al. 2007, Williams et al. 2017, Possamai et al. 2018). These extreme events can cause dramatic changes in estuarine systems, both in the short and long terms (Wetz & Yoskowitz 2013). Different species within the same estuaries are likely to present varying degrees of sensitivity to drought or increased flow conditions or responses to increasing salinities associated with sea level rise as a result of direct effects on physiology and fitness but also due to indirect effects on ecosystem changes, environmental displacement or habitat degradation. Climate change may also alter residence times and flushing rates through changes in the timing and intensity of rainfall that impact freshwater inflows, as well through the occurrence of more severe and frequent storm surges that affect tidal height potentially flooding diverse habitats and changing estuarine shape (Gillanders et al. 2011a, Robins et al. 2016). In the long term, though with some associated physiological costs, mobile species that can use wide-ranging habitats throughout estuaries will likely fare better than species with high site fidelity and reduced physiological tolerances, or species that are already at their physiological limits of distribution (Sakabe & Lyle 2010, Morrongiello et al. 2011, Williams et al. 2017).

Several studies have demonstrated how changes in river flow over short and long periods (for instance, associated with variations in long-term climatic patterns such as the North Atlantic Oscillation [NAO] or the El Niño-Southern Oscillation [ENSO]) impact the distribution, movements and abundance of different ecological functional fish groups in estuaries, as well as the connectivity between estuarine and coastal fish populations with segregated life history stages (e.g. Kimmerer 2002a, Garcia et al. 2003, Whitfield 2005a,

Martinho et al. 2007, Feyrer et al. 2015, Jaureguizar et al. 2016, Lund-Hansen et al. 2018). For instance, increased precipitation associated with extreme El Niño events coincided with a range expansion and increase in the number of freshwater fish species inside an estuary and triggered changes in the community that lasted over a year (Garcia et al. 2003, Garcia et al. 2019).

For larvae and juveniles of marine migrant species, freshwater is an important cue driving recruitment to estuarine nurseries (Champalbert & Koutsikopoulos 2009, Duffy-Anderson et al. 2014). Both high and low flows may impose physical and chemical barriers that hinder the ingress of larval and juvenile stages, and therefore reduce estuarine colonisation and connectivity. For instance, whilst decrease in flow may reduce larval attraction, or create temperature barriers that inhibit estuarine colonisation, increases in flow, linked to precipitation, may increase osmoregulatory stress, as well as the risk of larval and juvenile stages being flushed out to sea, likely affecting their survival (Vinagre et al. 2009, Gillanders et al. 2011b, Vasconcelos et al. 2011). Moreover, variations in flow and estuarine salinity gradient may limit the extent of suitable habitat for juveniles and prompt area avoidance or fish emigration. Ultimately, changes in flow may result in habitat compression, in particular along mesohaline habitats, which are widely recognised as key nursery areas in estuaries, and thus disrupt the provision of crucial ecosystem functions.

Worldwide, numerous estuaries are temporarily closed off from the sea by a sandbar that forms at the mouth. In temporarily open/closed estuaries, river flow is directly linked to mouth state and fish community richness, diversity and abundance (James et al. 2007, James et al. 2008, Chuwen et al. 2009). In the context of climate change, reduced river flows may lead to longer mouth closures, which will reduce immigration and emigration between the sea and the estuary, resulting in decreases in richness of marine affinity species within estuaries (Gillanders et al. 2011a, James et al. 2013, Hallett et al. 2017). These and other impacts of climate change associated with river flow and entrance channel modifications are discussed in further detail in Sections 7.3.1 and 7.3.6.

As changes to hydrological factors will necessarily affect habitats and food sources within estuaries, they are likely to have a range of indirect effects on estuarine fish. Variations in river flow may either disrupt or promote spatial connectivity in estuarine food webs, with the importance of autochthonous and allochthonous sources likely to change under climate change, and estuarine communities may rely more heavily on non-terrestrial sources of carbon in food webs during drought conditions (Abrantes & Sheaves 2010, Garcia et al. 2019). Moreover, it is likely that changes to freshwater discharges from estuarine plumes will also influence the provision of continental derived matter to coastal food webs (Hoffman et al. 2014, Garcia et al. 2017, Vinagre et al. 2019).

Changes to estuarine productivity associated with variations in freshwater input are also expected, and though extreme events such as floods and droughts can affect estuarine water quality and system productivity, the direction and degree to which this occurs may be related to local conditions (Wetz & Yoskowitz 2013) and between years (Snickars et al. 2015). In general, estuarine productivity is linked to increased river flow, but extreme flood events may lead to reduced food availability due to flush-out of accumulated organic matter as well as the disruption of phyto- and zooplankton cycles (Whitfield & Harrison 2003, Gillanders et al. 2011a). In contrast, less freshwater inflow generally equates to reduced allochthonous organic matter and weakened phytoplankton productivity. However, these conditions are not always the case and the relationship can be somewhat more complicated (see Wetz & Yoskowitz 2013). Overall, estuaries are complex and dynamic systems, where variations in river flow and salinity trigger a wide array of physicochemical and biological responses. In particular, changes in flow have modulating effects on flushing times, turbidity, water quality and stratification, for which determining broadly applicable direct causal links between climate change and impacts on fishes is difficult, and for many of the aforementioned relationships we need further empirical data to establish comparisons among systems.

#### 7.3.2 Temperature impacts

Climate change effects on fishes in estuaries are often interpreted relative to temperature change, which is seen as an important driver (Able & Fahay 2010). Temperature affects traits such as metabolic rates, spawning, migration and development (Costa 1990, Sunday et al. 2014) and is regarded as the most important factor controlling the distribution and climate vulnerability of marine fish (Walther et al. 2002, Portner & Knust 2007, Dahlke et al. 2020). Temperature-dependent processes vary over a species latitudinal distribution, such that fish populations living at the edge of their species distribution (Martinho et al. 2012). Rijnsdorp et al. (2009) predicted that temperate fish species at the limit of their distribution will decrease in abundance and tropical species will increase in abundance. There has been a change in the proportion of tropical and temperate transition zone, with the proportion of tropical species in the catches increasing over time, the overall abundance of temperate species has not decreased (Mbande et al. 2005, James et al. 2013). Temperate juvenile fish should have some capacity to adapt to changing thermal conditions (Munday et al. 2008, Eme et al. 2011), primarily because fish in estuaries are eurythermal and have adapted to wide seasonal changes in temperature (Costa 1990, Schulte 2014).

Specific examples of estuarine fish responses and range shifts are becoming more evident (e.g. Nicola et al. 2018, Peck et al. 2020). Fodrie et al. (2010) reported on range shifts in seagrass fish assemblages in the northern Gulf of Mexico between the 1970s and 2006–2007. They recorded numerous tropical/sub-tropical species in 2006–2007 that were absent from catches in the 1970s, as well as an increase in abundance of tropical species that were present in the 1970s. Similarly, James et al. (2008) documented an increase in occurrence of tropical species annually in a temperate South African estuary over an 11-year study, resulting in an increase in species richness in the estuary. For tropical species, winter survival is often the bottleneck in the establishment of populations in temperate areas (Figueira & Booth 2010). Elevated winter temperatures associated with climate change may allow fish to overwinter and become established in these systems (Hurst 2007). However, in a 26-year (1990–2015) study of larval fish ingress into a US east coast temperate estuary, 5 of the 5 species with northern affinities declined while 18 of 21 species with southern affinities increased over time (Morson et al. 2019).

Climate variability and the frequency and magnitude of extreme events may have the greatest impact on fish populations, rather than a gradual increase in mean temperatures (McCarty 2001). Though estuaries may experience temperature changes on a daily and tidal basis, in temperate and subtropical estuaries, extreme cold events have caused mass mortalities of tropical species (Hurst 2007), potentially maintaining the dominance of temperate species in these systems and slowing the poleward migration of tropical species (Boucek et al. 2016). Low temperatures are often more lethal than high temperatures (Costa 1990) and during extreme cold spells, temperatures decrease below the physiological limits of many tropical species (Boucek et al. 2016). Boucek and Rehage (2014) documented the response of estuarine fish assemblages to an episodic cold front in subtropical South Florida. The cold front decreased water temperatures below the physiological limits of tropical species for less than 2 weeks, resulting in a 70% decrease in abundance of tropical species (Boucek & Rehage 2014). In the subtropics, extreme events are predicted to increase in frequency, intensity and duration as a result of climate change (Boucek et al. 2016).

Species which have evolved in tropical regions are predicted to be most at risk to climate warming because they have evolved within an aseasonal thermal environment and have narrow thermal windows (Sunday et al. 2012). This may not, however, be the case for juvenile fishes inhabiting shallow water estuarine nursery habitats. Bennett (2010) found that approximately 70% of juvenile fishes (44 species) inhabiting shallow water nursery areas (mangrove, seagrass and tide pools) around Hoga Island in Indonesia were able to tolerate temperatures above 40 °C. Juvenile squaretail mullet (*Liza vagiensis*) and crescent terapon (*Terapon jarbua*) from seagrass beds around Hoga Island were exceptionally tolerant of high temperatures, with mean Critical Thermal Maxima values between 44 and 45 °C (Eme et al. 2011). As such, Eme et al. (2011) suggested that water temperatures in shallow water nursery habitats would have to be considerably higher to negatively affect juvenile fishes in the tropical Indo-Pacific region.

Overall, thermal tolerance is dependent on phylogeny, geographic origin and ontogeny but tolerance ranges of spawning adults and embryos are consistently narrower than other life stages (Dahlke et al. 2020). Despite the tolerance of juvenile fishes in estuaries to extreme temperatures, the distribution and abundance of estuary-associated species is also strongly linked to coastal water temperatures as opposed to estuarine water temperatures – as was found for mugilids (James et al. 2016). This is likely a result of the narrower thermal windows in the early life stages (Portner & Peck 2010) which occur in the marine environment, and can originate feedback effects on fish populations that rely on estuaries as nurseries. Predicting how estuarine fish communities will change in response to climate change is further complicated as each species responds differently to warming and assemblages are unlikely to shift their distribution as a unit (James et al. 2013).

Increasing water temperatures may also have positive impacts (such as increased growth, maturity, feeding rates, overwinter survival) on certain fish species, provided it occurs within their tolerance range (Murawski 1993). Faster growth may improve an individual's chance of survival through reduced susceptibility to predation during the shortened juvenile life stages (Drinkwater et al. 2010, Gillanders et al. 2011a). Conversely, warmer water temperatures may also increase the metabolic demands of predators resulting in an increased predation risk (Hurst 2007). Multiple stressors, such as salinity and dissolved oxygen, may also interact with temperature in complex ways (Schulte 2014). For example, hypoxia tolerance declines as temperature increases (Schulte 2014), while death from low temperatures may result from an inability to maintain osmotic balance as temperatures decrease (Hurst 2007).

The toxicity of various substances also varies with temperature (Costa 1990). For example, when reviewing the effects of ammonia on estuarine fishes, Eddy (2005) found that ammonia was more toxic to larva and juvenile fish during periods of elevated temperature and pH. Fishing causes truncation of the age structure (by removal of larger individuals) of exploited populations and loss of spatial heterogeneity (distribution may shrink), which makes exploited species more vulnerable to changing temperatures (Potts et al. 2015).

## 7.3.3 Dissolved oxygen impacts

Dissolved oxygen is a critical parameter that can vary on multiple spatial and temporal scales and manifest impacts on fishes from individuals to populations to ecosystems (Wu 2002, Breitburg et al. 2009b) from the tropics to temperate regions (Roessig et al. 2004). Estuaries, in particular, can experience significant fluctuations in dissolved oxygen concentrations as a result of numerous simultaneous processes (Gillanders et al. 2011a, Dubuc et al. 2019). These include daily fluctuations in photosynthetic activity, seasonal variation in net ecosystem metabolism, periodic and aperiodic inflow events that deliver nutrients to fuel primary production as well as promote circulation with neighbouring oceans, and algal blooms that consume oxygen upon decomposition (Diaz & Rosenberg 2008, Howarth et al. 2011, Johnson et al. 2019).

The solubility of oxygen in water is mediated by temperature, and thus elevated temperatures can contribute to declines in oxygen storage capacity in estuarine waters (Doney et al. 2012). The interactive or synergistic effects of these parameters has direct impacts on fish survival or can act as water quality barriers for estuarine fish species (Pomfret et al. 1991, Roman et al. 2019). In general, the loss of oxygen from



**Figure 7.6** (a) Oxygen solubility at different salinity and temperature conditions, with boxes indicating the range of salinity and temperature during periods of seasonal deoxygenation for seven coastal and estuarine bodies of water around the world (from Roman et al. 2019); and (b) a diagram illustrating the conceptual idea of the effects of hypoxia on the standard metabolic rate (SMR), routine metabolic rate (RMR), maximal metabolic rate (MMR) and aerobic scope (AS) of an oxyregulator (from Rogers et al. 2016), establishing the relationship between an organism's respiration rate (oxygen consumption) and the environmental oxygen partial pressure. Standard metabolic rate (SMR) reflects the oxygen consumption rate at a minimum or defined level of activity, whilst maximum metabolic rate (MMR) reflects the highest rate of oxygen consumption possible by a fish. The aerobic scope (AS) reflects the energy available for activity above minimum requirements, whilst  $P_{crit}$  is the critical point at which oxygen supply no longer meets metabolic requirements.  $Pc_{max}$  is the critical external oxygen partial pressure at which oxygen supply no longer meets the maximum demand for oxygen.

aquatic systems is referred to as 'deoxygenation', whereas waters with oxygen concentrations below specified thresholds are known as 'hypoxic' (Breitburg et al. 2018). Hypoxia is a permanent condition of some estuaries, particularly those with limited circulation and deep areas that may not be physically mixed and reoxygenated, such as the Baltic Sea (Carstensen et al. 2014) (Figure 7.6). In the context of climate change, because water temperature controls oxygen solubility but also the metabolic demand of organisms, variations in temperature across ecosystems will drive the severity of hypoxia impacts on estuarine fish communities. Ultimately, differences in temperature, across ecosystems, or over time within an ecosystem, will dictate the severity of the impacts associated to hypoxia, even at similar dissolved oxygen concentrations (Roman et al. 2019).

Operationally, dissolved oxygen values are typically considered hypoxic when concentrations are below 2.0 mgl<sup>-1</sup>. Hypoxia is typically defined based on generalised organismal responses that are functionally impaired below the 2.0 mgl<sup>-1</sup> threshold. However species-specific oxygen requirements and tolerances can vary substantially (Wannamaker & Rice 2000, Bell & Eggleston 2004) and susceptibility to low oxygen can vary within a species due to age, size or acclimation history (Rogers et al. 2016, Roman et al. 2019). Thus, while the 2.0 mgl<sup>-1</sup> is a common delimiter of hypoxic waters, not all organisms may respond uniformly to that threshold. Some species may be sensitive to waters with dissolved oxygen in the 2–4 mgl<sup>-1</sup> range and suffer physiological or behavioural consequences at these intermediate values. Others may tolerate dissolved oxygen levels below 2.0 mg l<sup>-1</sup> with varying magnitudes of sublethal effects, and these more tolerant species may not always exhibit significant reductions in growth performance (Altenritter & Walther 2019). The

range of dissolved oxygen sensitivities across species means species assemblages may shift at intermediate dissolved oxygen levels and become relatively more dominated by hypoxia-tolerant species.

One metric that is often used to identify low oxygen tolerance is the critical  $O_2$  tension, or  $P_{crit}$ , which is defined as the oxygen concentration at which organisms switch from oxyregulating (i.e. able to maintain stable oxygen uptake necessary for metabolic requirements) to oxyconforming (i.e. oxygen uptake declines linearly with declining ambient DO availability) (Figure 7.6). Although the concept and experimental procedures used to measure  $P_{crit}$  have been criticised for methodological, definitional and theoretical inconsistencies (Wood 2018), it is still frequently used to evaluate the relative sensitivity of fishes to low oxygen levels and help predict tolerant and sensitive species.

A recent meta-analysis of in marine and freshwater fishes worldwide by Rogers et al. (2016) found wide ranges in  $P_{\rm crit}$  values, but the authors noted these values interacted with additional biotic and abiotic variables such as salinity, temperature, body mass and routine metabolic rate. Sensitivity to low dissolved oxygen can also be influenced by water pH, diet quality, consumption rates, dissolved oxygen acclimation histories and toxic metal exposure. Clearly, some or all of these factors should be considered in combination with  $P_{\rm crit}$  when attempting to accurately predict species-specific responses to dissolved oxygen reductions. Additional methods to assess dissolved oxygen sensitivity include loss of equilibrium measurements and quantifying lactate accumulation to indicate metabolic depression (Wood 2018). Regardless, the central point that individual species may not respond uniformly to a specific concentration of dissolved oxygen remains, and this issue will be critical for accurate forecasting of estuarine responses to future dissolved oxygen conditions.

The most immediate impact of low oxygen is mortality of sensitive organisms. Mortality events can be punctuated and severe, particularly when a low oxygen event is associated with a short-term change in the system such as increased eutrophication from a flood event that prompts algal blooms and oxygen depletion. These can lead to 'fish kills' where large abundances of fishes experience mass mortality (Figure 7.7). Thronson and Quigg (2008) conducted a meta-analysis of a 55-year time series of fish kills leading to mortality of over 383 million fish along the coast of Texas in the north-western Gulf of Mexico. They found that 57% of fish kill events were attributed to low dissolved oxygen driven by various factors, including algal respiration, eutrophication and reduced inflow. The impact of low oxygen varied by location,



**Figure 7.7** Fish kill event due to hypoxia in (a) Nangudga Lake Estuary, Australia (Photo: courtesy of Melinda Coleman) and (b) Mdloti Estuary, South Africa (Photo: courtesy of Nicolette Forbes).

with 78% of fish kills in Galveston Bay alone attributed to low oxygen. However, the distribution of mortality was unequal across species. For Galveston Bay fish kills, Gulf menhaden *Brevoortia patronus* and finescale menhaden *Brevoortia gunteri* were the leading species affected, likely because of their physiological sensitivity, schooling behaviour and attraction to algal blooms for feeding. This case study illustrates the severe impact punctuated oxygen loss can have on differential mortality of estuarine fisheries.

Fundamentally, with oxygen required to sustain aerobically respiring organisms, minimum concentrations of dissolved oxygen are essential for the sustainable functioning of estuarine-dependent fish populations beyond impacts of climate-related mass mortality events alone. Physiological processes including metabolism, development and growth can all change in response to reduced oxygen, and many of these responses are additionally temperature-dependent (Marcek et al. 2019, Targett et al. 2019). Beyond those processes directly linked to metabolism, fish may be impaired in other fitness-relevant traits such as fecundity (Hassell et al. 2008). For instance, hypoxia exposure has been shown to interfere with normal reproductive development in estuarine-dependent fishes (Thomas et al. 2007). Low DO can disrupt hormone function, ovary development, oocyte maturation, and ultimately egg production by females. Male gametogenesis and sperm motility can also be impaired by low dissolved oxygen (Thomas & Rahman 2010). In extreme cases, this hormone impairment can lead to the simultaneous development of spermatocytes and oocytes within an individual female fish, and the prevalence of intersex individuals can be as high as 19% in the estuarine-dependent Atlantic croaker *Micropogonias undulatus* inhabiting coastal waters of the northern Gulf of Mexico (Thomas & Rahman 2012).

Reductions in gamete development, output and fertilization success could lead to significant reductions in long-term population growth. In an individually based modelling effort that integrated lethal and sublethal effects of hypoxia on physiology, reproduction and behaviour of *M. undulatus*, Rose et al. (2018) found that the sublethal reduction in fecundity was more important than reduced growth resulting in lower population sizes projected 100 years into the future. This highlights how the relative impacts of dissolved oxygen on specific fitness-related factors may have variable carry-over effects for future generations, and not all factors may be equally important for population-level processes (Rose et al. 2009).

Because they are mobile, estuarine fish may avoid or move outside hypoxic waters, dependant on the rate of environmental change in oxygen levels. Yet, if climate change leads to widespread areas or habitats becoming permanently or suddenly hypoxic, then both increases in fish mortality or ecosystem-level changes are expected (Gillanders et al. 2011a). Understanding the physiological and ecological consequences of changes in estuarine habitat use associated with hypoxia is critical at both individual (e.g. survival, energy demand, diet shifts, condition and overall fitness) or estuary-level (e.g. assemblages, food web shifts, ecosystem functions).

In the context of climate change, the effects of hypoxia on ecosystem-level productivity may be difficult to identify unambiguously given the numerous co-varying stressors that may be simultaneously operating on a given system. In addition to water temperature, hypoxia is often fuelled by large inputs of terrigenous nutrients, which may at least temporarily increase primary and secondary productivity of the system (eutrophication). The true impact of hypoxia on estuarine fishes may, therefore, be complex and a result of the net trade-offs of increased productivity, physiological stress, and growth and reproductive impairments (Breitburg 2002). Species assemblages may shift as a result of different hypoxia tolerances, whereby the standing stock of biomass could remain similar before and after a hypoxic event but the community composition could shift dramatically. Overall, low oxygen episodes can create barriers to different areas and habitats, or change the spatial distribution of estuarine fishes but will be tightly coupled to prevailing temperatures and species-specific metabolic requirements and tolerances (Pomfret et al. 1991, Ludsin et al. 2009, Roman et al. 2019).

The effects of hypoxia on food web interactions can be significant for many organisms (Levin et al. 2009, Zhang et al. 2010). In addition to direct mortality, a range of sublethal and indirect effects can cause altered interactions with diverse outcomes (Breitburg et al. 1997). For consumers, a notable effect is displacement from hypoxic waters, both laterally and vertically due to behavioural avoidance of hypoxic waters (Ludsin et al. 2009, Zhang et al. 2009). Displacement may lead to reduced growth and condition if consumers are forced into sub-optimal habitats with energetically inferior prey or due to density-dependent interactions among conspecifics or competitors (Essington & Paulsen 2010). If benthic prey that remain in hypoxic zones exhibit increased tolerance to low DO compared to consumers, hypoxic zones act as a refuge that reduces predation rates on benthic food webs (Sagasti et al. 2001). Predation pressure would instead shift to alternative food webs, such as pelagic prey.

Alternatively, hypoxia may cause benthic infauna to move closer to the sediment-water interface where they are more readily available to mobile demersal predators that preferentially target stressed or moribund prey items (Pihl et al. 1992, Long & Seitz 2008). If predators exhibit greater hypoxia tolerance than prey, hypoxia does not provide refuge and instead enhances consumer consumption rates. Hypoxic displacement often leads to aggregations of consumers at the edges of suboxic waters (Craig & Crowder 2005, Craig 2012), presumably to allow periodic foraging forays into hypoxic zones to target preferred prey (Mohan & Walther 2016). Thus, predators may experience a net benefit from periodic hypoxic events through increased foraging efficiency that may offset energetic expenditures induced by hypoxic exposure stress. In this case, food web interactions between demersal predators and benthic prey may remain robust despite hypoxia exposure, particularly for predators with greater hypoxia tolerance.

## 7.3.4 Impacts of elevated CO,

Carbon dioxide emissions have been increasing rapidly since the industrial revolution. Associated with increasing atmospheric CO<sub>2</sub> is the dissolution of this gas into surface waters forming bicarbonate (HCO<sub>3</sub><sup>-</sup>) and hydrogen (H<sup>+</sup>) ions. Increased oceanic CO<sub>2</sub> leads to increased CO<sub>2</sub> partial pressure which results in a drop in ocean pH. To date, a decrease of ~0.1 pH units has occurred, but without abatement of CO<sub>2</sub> emissions, pH is expected to decrease by a further 0.3–0.4 units by the end of the century (Allmon & Esbaugh 2017, Cattano et al. 2018). Estuarine systems are naturally more susceptible to ocean acidification than marine systems since eutrophication leading to increased microbial respiration can also lead to changes in pH (Allmon & Esbaugh 2017). Estuarine systems being shallower, less saline, and having lower alkalinity and more freshwater input than marine waters may also make them more vulnerable to ocean acidification (Miller et al. 2009). Moreover, there are also regular diel and seasonal shifts in CO<sub>2</sub> levels associated with estuaries.

Estuarine systems experience substantial changes in  $O_2$  and  $CO_2$  between day and night. This variation can lead to high levels of oxygen during the day associated with photosynthesis and low levels at night associated with respiration, referred to as diel-cycling hypoxia (Miller et al. 2016, Lifavi et al. 2017). The diel cycles in oxygen also cause considerable changes in  $CO_2$  leading to increased p $CO_2$  and decreased pH (Miller et al. 2016, Lifavi et al. 2017). Eutrophication and ocean acidification exacerbate these patterns. In experiments combining diel cycles of both dissolved oxygen and pH/p $CO_2$ , growth of juvenile weakfish (*Cynoscion regalis*) was not impacted by moderate diel cycles; however, at extreme diel-cycling there was 100% mortality within 10 days (Lifavi et al. 2017). Overall, *C. regalis* were tolerant of diel cycles of oxygenation and acidification encountered in estuarine habitats. Acidification may, however, make fish more sensitive to hypoxia which can impact all life stages (Miller et al. 2016).

Much research has focussed on behavioural and ecological effects of ocean acidification on marine species, and there has been far less research on estuarine species (Lonthair et al. 2017, Lauchlan & Nagelkerken 2019). Many of the consequences of ocean acidification will affect the ecology, behaviour and physiology of fishes, with effects likely to vary by life stage and species (Cattano et al. 2018). Many of the effects of  $CO_2$  on fish are related to compensation for respiratory acidosis where acid equivalents are transported into the environment and base equivalents are transported into the blood plasma via gill ionocytes (Allmon & Esbaugh 2017).

A recent review has investigated the effects of increased CO<sub>2</sub> (mid-century and end-of-century emission scenarios) on eco-physiological and behavioural responses of fish (Cattano et al. 2018). Briefly, this metaanalysis involved 64 studies and 42 species covering all oceanic regions. It included ontogenetic stages as a factor in the analyses, as larval fish are thought to be more susceptible to ocean acidification than juvenile or adult fish due to inefficiencies in ionic regulation (Baumann et al. 2011). Physiological types of stenohaline or euryhaline fish, climate zones and habitats were also included as factors in the meta-analysis. Depending on where fish have evolved, differences in the effects of ocean acidification may be expected. For example, fish living in environments, climate zones or habitats that fluctuate very little in terms of environmental conditions may be expected to be more sensitive. Fish living in estuarine systems which often fluctuate greatly in terms of freshwater input (and therefore salinity) may be expected to be less sensitive (but see Pistevos et al. 2017). Despite these predictions, the meta-analysis found a higher mortality of embryos and larvae exposed to high pCO<sub>2</sub> for euryhaline rather than stenohaline fish (Cattano et al. 2018). These authors found numerous CO,-dose-dependent effects on a range of measured traits, particularly for larvae, suggesting ecological consequences for fish populations. They noted that reproduction, development and habitat choice have been poorly studied and that there is a need for multi-generational, multistressor experiments (Cattano et al. 2018). In addition, whilst community-level effects are likely, species interactions and ecosystem-level changes, particularly for estuarine fish, are also poorly understood. Ultimately, most studies to date report negative effects of acidification based on short-term experimental manipulations of pH, and therefore it is pivotal we address this issue over multiple generations, and investigate transgenerational plasticity. This may be a powerful mechanism by which fish populations may potentially adapt and ameliorate the negative impacts of acidification (Munday 2014, Cattano et al. 2018).

Changes in  $CO_2$  are likely to also involve changes in other environmental variables. As indicated in the recent meta-analysis, there is a need for more multi-stressor experiments that involve ocean acidification, since most experiments involve temperature and ocean acidification. In estuarine systems, however, changes in salinity or oxygen are also likely to occur. Few studies have combined ocean acidification with other stressors such as hypoxia (but see DePasquale et al. 2015, Miller et al. 2016, Lifavi et al. 2017, Targett et al. 2019) or salinity (Pistevos et al. 2017) to investigate effects on fish.

Behavioural experiments suggest that ocean acidification may affect sensory function of fish, thereby potentially affecting a fish's ability to orientate including detection of estuarine cues (Leis 2018) [but see Clark et al. (2020)]. Interestingly, it was suggested that fish may then conform to a passive-dispersal paradigm and that less self-recruitment, increased dispersal differences and decreased settlement rates may occur (Leis 2018). Only a single study has demonstrated that preference for different temperature and salinity cues (associated with estuarine waters) may change associated with ocean acidification (Pistevos et al. 2017). Altered sensory responses are likely associated with changes in neurological processing and coordination. If fish recruiting to estuaries cannot sense the temperature and salinity gradients associated with an estuary, this is likely to impact population replenishment and alter connectivity patterns. Moreover, acidified conditions can alter otolith development, size and relative density, impacting a fish's auditory sensitivity, with potential deleterious effects on survival and recruitment; however, such effects have not yet been shown for fish that use estuaries as nurseries (Munday et al. 2010, Simpson et al. 2011, Bignami et al. 2013).

Fish with short generation times may respond to ocean acidification via evolutionary adaptation. If this is not possible, as might be the case for long-lived species, then genotypes within the population that are resilient to ocean acidification may persist. In addition, phenotypic plasticity may also allow fish to persist. Estuarine fish species are also thought to be resilient to elevated CO<sub>2</sub> since they naturally experience changes in acidification and other environmental parameters. Experimental work suggests that, a fast-growing estuarine species, red drum (*Sciaenops ocellatus*) may exhibit tolerance to high pCO<sub>2</sub> in terms of survival and behaviour (Lonthair et al. 2017).

#### 7.3.5 Sea level rise

Sea level rise will play a dominant role in shaping estuarine environments because not only will it change seawater intrusion and drive changes to tidal prism but it will also be the main climatic factor dictating increases to estuarine depth and thus intertidal habitat loss. Sea level rise will modify estuarine topography and, as water level rises, marine water will naturally penetrate farther into estuaries (Chua & Xu 2014, Yang et al. 2015). However, the direct impact of sea level rise on estuarine fish is hard to quantify. Overall, numerous physical and chemical mechanisms in estuaries, including erosion/deposition cycles, sediment composition, changes to the extent of the photic zone and biogeochemical cycles will be affected (Wolanski & Elliott 2016). This makes it difficult to isolate the effects of sea level rise in any given estuary. Moreover, within estuaries, the impacts of rising sea levels will vary depending on geomorphology, as well as the level of development and regularisation of marginal habitats and estuarine boundaries, and thus individual estuaries may be affected differently (Zhong et al. 2008, Robins et al. 2016, Hallett et al. 2017, Borchert et al. 2018). A further confounding factor is that sea level rise is not spatially uniform and there are accompanying vertical land movements (e.g. isostatic rebound, land sinking/subsidence) (Day et al. 2011) and thus sea level rise is often termed relative sea level rise to allow for the effects of local isostatic adjustments. For example, in east coast US estuaries between Cape Cod and Cape Hatteras, there is extensive postglacial subsidence such that the region has the highest sea level rise rate, approximately double the global average (Miller et al. 2013).

Several studies have documented changes in estuarine geomorphology, salinity intrusion, habitat loss and changes in animal and plant communities in estuaries linked to sea level rise and associated coastal squeeze, with several reporting chronic decreases in productivity and loss of nursery habitat though others show increases in seagrass habitat and nursery function due to advancing marinisation, or improved productivity and habitat quality linked to greater tidal flushing (Day et al. 2011, Gillanders et al. 2011a, Cloern et al. 2016, Hallett et al. 2017, Borchert et al. 2018).

For fishes in estuaries, the impact of sea level rise will be underpinned by the capability of estuarine systems to continue to deliver key ecosystem functions. One key concern regarding sea level rise in estuaries is the potential for loss of critical habitats for fishes (such as wetlands, mangroves, saltmarshes, seagrasses, oyster reefs, and other subtidal and intertidal nursery, feeding or spawning grounds). This is especially true in systems where coastal development and urbanisation impede the natural relocation of habitats in parallel with the displacement of optimum or suitable conditions along the estuarine gradient (i.e. coastal squeeze – where estuarine and coastal habitats are blocked from relocating up the estuary and are squeezed between the encroaching ocean and human built environment). An increase in sea level can also compromise wetland and intertidal communities by increasing inundation frequency and duration, or inducing stronger wave energy (Day et al. 2011, James et al. 2013). In many cases, the rate of sea level rise will exceed that of sediment accretion, and may outpace the ability of habitats to relocate and establish new subtidal, intertidal or wetland habitats (Roessig et al. 2004, Janousek & Mayo 2013, Rodriguez et al. 2017). As an example, in the Mid-Atlantic portion of the US east coast, Spartina marshes are being lost to sea level rise related erosion at the marsh edge and in the interior as marsh pools turn into lakes with subsequent loss of marsh vegetation (Kirwan & Megonigal 2013, Able 2021). On the other hand, in the Gulf of Carpentaria, unrestrained by direct human impacts or urbanisation, mangroves have expanded landward as a result of the combined effects of sea level rise and the prolonged periods of tidal and freshwater inundation on coastal lowlands (Asbridge et al. 2016), yet how mangroves will adjust to sea level rise will vary according to local oceanographic conditions, topography and sediment availability (Woodroffe 2018). Put simply, essential estuarine habitats for fishes will only persist if they can (i) accrete or grow vertically at a rate that matches that of sea level rise, or (ii) swiftly relocate to alternative, adequate upslope or landward areas along the land-sea interface.

Most estuaries are preferred human settlement areas, and thus sea level rise impacts in estuaries worldwide are aggravated by urban development, bank regulation and other built structures (e.g. dams, weirs, roads) which impede the displacement of estuarine biota along with the shifts in abiotic conditions, slowly squeezing out suitable areas available for key fish habitats (Day et al. 2011, Kirwan & Megonigal 2013, Robins et al. 2016, Borchert et al. 2018). Ultimately, the concerted impacts of sea level rise, together with other local and regional scale factors, will have the greatest impacts on estuarine fishes via the potential loss, reduction or impairment of essential habitats.

## 7.3.6 Estuary entrance channel openings and fish access

Fish responses to climate change are highly context-dependent, as they are influenced at large biogeographical scales but also by local estuarine conditions, the fish guilds (Elliott et al. 2007) or even the species being considered, and will differ depending on estuarine hydrogeomorphology (Gillanders et al. 2011a, Valesini et al. 2013, Vasconcelos et al. 2015). Estuaries can be generically classified as permanently or temporarily open. In permanently open estuaries, access to the marine, estuarine and freshwater environments is continuous. However, variations in physicochemical conditions have the potential to curb or enhance the ingress, permanence or egress of different species and life stages, linked primarily to river flow, habitat availability and species affinities to environmental conditions (e.g. salinity and others), as discussed previously. These responses can be dwarfed by new, permanently open inlets when induced by storms as has occurred in the western North Atlantic (Olin et al. 2019).

The responses of fishes to climate-induced changes in temporarily open/closed estuaries will be less predictable, and will depend on the timing and duration of the bar openings, and the effects of altered river flows on salinities (Gillanders et al. 2011a, James et al. 2013, Hallett et al. 2017). Changes to rainfall patterns, as well as sea level rise (e.g. via stronger wave actions, increased storm surges) will regulate sand-bar formation and the frequency, timing and duration of mouth opening. For instance, periodically open estuaries are particularly common in southern Africa and Australia, where declining rainfall in some regions, allied to changes in erosion/deposition cycles, will likely stimulate the closure of some systems or promote extended closed periods between mouth openings (Gillanders et al. 2011a, James et al. 2013, Hallett et al. 2017) (Figure 7.8). The prospect of such events will in turn increase the likelihood of dramatic increases in salinity, temperature, algal blooms or water column stratification, among others, which will significantly impact estuarine fish (Hodgkin & Hesp 1998, Hoeksema et al. 2006, Wetz & Yoskowitz 2013, Human et al. 2016). Over time, species persistence, and individual biological performance or survival will be inextricably mediated by homeostasis and stress tolerance to changing conditions.

The state of the mouth has a pivotal role in estuarine connectivity and determining species richness, composition, diversity and abundance of fishes within temporarily open/closed estuaries (Whitfield 2005a,



**Figure 7.8** Artificial opening of (a) Kianga Lake Estuary (Australia) and (b) Mdloti Estuary (South Africa). Opening of intermittently closed and open estuaries, lakes or coastal lagoons is managed by various authorities to allow fish access, and mitigate or reduce the impacts of hypoxia, increases in salinity, temperature, algal blooms or water column stratification, among others (Photos: courtesy of Melinda Coleman and Nicolette Forbes).

Valesini et al. 2013, Vasconcelos et al. 2015, Hallett et al. 2017). Together with fish species that complete their life cycles within estuaries, or that can tolerate sudden changes (a common feature), one of the largest group of fishes in temporarily open/closed estuaries are juveniles of taxa that breed at sea and use coastal environments as nurseries (i.e. marine migrants) (Elliott et al. 2007, James et al. 2007, Taddese et al. 2018). The continued ability for larvae to recruit to temporarily open/closed estuaries is inextricably linked to the timing, duration and associated flow conditions of mouth opening periods. In the same manner, subadults will require periodical openings to join adult populations and later spawn in the marine environment.

A similar prerequisite exists for diadromous species to perform their migratory runs and complete their life cycles. Because of their dependence on freshwater-marine transitions to complete their life cycle, diadromous species face an uncertain future worldwide as climate change compounds a long-lasting litany of anthropogenic threats (Lassalle & Rochard 2009, Limburg & Waldman 2009). Overall, increases in entrance closures will in general restrict connectivity and fish movement between fresh, estuarine and marine environments, contributing to greater proportions of resident species, and to declines in the presence of marine, marine migrant and diadromous taxa, associated in part with reduced larval recruitment and loss of estuarine nursery function.

Nevertheless, there is increasing awareness that partial migration is widespread in fish populations. This complex migratory behaviour implies the existence of different co-existing groups with distinct migratory strategies within the same population (e.g. migratory versus resident, distinct seasonal and lifetime migration patterns) (Kerr et al. 2009, Chapman et al. 2012, Gahagan et al. 2015). The same applies to different reproductive cohorts of the same species (Neuman & Able 2003). In the context of climate change, and in particular for estuarine fishes, understanding these sources of variation is particularly pertinent, as distinct strategies may be more favourable in some conditions than others (Gillanders et al. 2015, Lennox et al. 2019). Therefore, understanding whether long-lived individuals migrate under some conditions but not others, as a result of individual-level plasticity and behavioural responses, or associated with distinct individual genetic makeup and physiological thresholds to environmental conditions, will underpin how species and fish populations may be affected by changing conditions in estuaries, including modifications to the estuarine mouth.

#### 7.3.7 Disease

In addition to influencing growth, behaviour, reproduction and the distribution of fish, altered environmental variables associated with climate change are expected to influence the impacts of pathogens and parasites (Harvell et al. 1999, Marcogliese 2001). For example, warming may cause increases in transmission rates, shifts in host ranges, increased virulence and increased susceptibility in fish populations leading to general expansion in the distributions and abundance of pathogens (Harvell et al. 2002).

Temperature is likely to have the greatest effect on disease and parasites in fish due to its effects on distributions and physiology. Such effects may also be exerted on estuarine fish populations by other factors associated with changing climate, e.g. changes in sea level and eutrophication. Links between warming El Niño southern oscillations and outbreaks of marine diseases are well established (Marcogliese 2008). Long-term monitoring and field studies also indicate the proliferation of harmful pathogens under warming regimes. Field studies over 40 years have demonstrated harmful bacteria of the genus Vibrio in estuarine and coastal regions are most abundant during periods of seasonal warming (Pruzzo et al. 2005). During long-term monitoring off the mouth of the Rhine Estuary from 1961 to 2005, Vibrio spp., which prefer warm temperatures and low salinities, were in greatest abundance during warm years (Vezzulli et al. 2013). In the Mediterranean, Vibrio abundance rose sharply when temperatures exceeded 22 °C and triggered mass benthic invertebrate mortality events (Vezzulli et al. 2010). Warming has also been linked with changing distributions of pathogens in aquatic environments. The bacterial pathogen Aeromonas salmonicida, which typically infects salmonids and other freshwater-estuarine species with ulcerative furunculosis, made its first appearance in the James Bay region of northern Canada when water temperatures exceeded 15°C in the 1990s (Tam et al. 2011). However, influences of warming are unlikely to be uniform. In cooler regions, warming may lead to influxes of new species and increased rates of growth and transmission; however, in warm regions, further warming may lead to decreases in host lifespan and biodiversity leading to reductions in pathogen and parasite communities (Lafferty 2009).

Effects of temperature are likely to operate in conjunction with other changes in environmental variables associated with climate change. Changes to precipitation and riverine flows may cause stratification that can exacerbate effects of temperature and alter salinity regimes. Sea level rise will also lead to changes in estuarine salinity regimes, with potential for saltwater intrusion. Salinity can play a key role in the occurrence of aquatic pathogens. For example, *Streptococcus* infection in Nile tilapia *Oreochromis niloticus* increases at warm temperatures with high salinities (Chang & Plumb 1996). The bacteria *Neoparamoeba pemaquidensis*, which causes Amoebic Gill Disease (a costly and widespread disease affecting wild capture and aquaculture fisheries), is associated with warm temperatures but has species-specific effects in relation to salinity, occurring in turbot *Scophthalmus maximus* at lower salinities (~22) than in salmonids (~32) (Munday et al. 2001). Such species-specific effects of pathogens in relation to multiple environmental variables will add to the complexity of interactions likely to affect estuarine fish communities in response to a changing climate. Additionally, changing salinity regimes will likely see increases in saltwater parasites in estuaries in response to saltwater intrusion with sea level rise (Marcogliese 2008).

Eutrophication, though caused by elevated nutrient inputs, may be exacerbated by environmental variables, e.g. reduced precipitation may result in low riverine flows and thus limited mixing and promoting eutrophication. Eutrophication can also promote the spread of pathogens and parasites (Johnson & Carpenter 2008). In basic terms, this occurs where elevated nutrients increase food availability resulting in population growth; this provides an increase in hosts and population density that is conducive to the spread of disease (Lafferty & Holt 2003). Ectoparasites and generalist parasites with simple life cycles are especially likely to exploit elevated eutrophic conditions, while macroparasite populations may decline due to reductions in intermediate hosts or mortality in free-living stages (Zander & Reimer 2002). When eutrophication exceeds certain limits resulting in algal proliferation and hypoxia, it becomes a stressor that increases fish susceptibility to infection. For example, in Chesapeake Bay, infection of mummichogs *Fundulus heteroclitus* by nematodes of the genus *Eustrongylides* was highest in eutrophic areas of the system and more prevalent as salinity decreased (Weisberg et al. 1986). This further illustrates the often complex interplay between environmental variables that are likely to be influenced by climate change and the capacity of disease to exploit favourable conditions that may arise in estuarine fish communities.

## 7.4 Climate change and fisheries in estuaries

Requirements for food security increase with human population growth and drive associated demands on fisheries production that are expected to rise by up to 50% by 2050 (Rice & Garcia 2011). Estuaries are key contributors to fisheries production, e.g. around half of fisheries catches landed in the USA depend on estuarine habitats (Houde & Rutherford 1993, Lellis-Dibble et al. 2008), while in Australia the figure may reach 75% (Creighton et al. 2015). Estuaries provide habitat not only for estuarine-resident species, but play key roles in the life cycles of other fish as spawning-, nursery- or transitory habitats. These include many species that spend most of their lives in other habitats, including estuarine spawners (e.g. winter flounder *Pseudopleuronectes americanus* and numerous sub-stocks of herrings *Clupea* spp.); marine spawners (e.g. eels *Anguilla* spp., European flounder *Platichthys flesus* and Atlantic menhaden *Brevoortia tyrannus*) and freshwater spawners (e.g. salmon *SalmolOnchoryncus* spp. and barramundi *Lates calcarifer*). In addition to providing habitat, estuaries also provide resources that drive fisheries catches, e.g. estuarine discharge is frequently a key nutrient source and driver of productivity in coastal fisheries (Drinkwater & Frank 1994, Grimes 2001). Climate effects on estuaries therefore have potential to affect not just estuarine fisheries, but broader coastal fisheries.

#### 7.4.1 Links to fisheries catches

Climate change will likely impact both yield and composition of fisheries catches (increases in some species and declines in others) via effects on physiology, connectivity and productivity, potentially resulting from changes in precipitation, temperature, sea-level and carbon dioxide levels (Figure 7.9). Changes to intensity and timing of precipitation are predicted to be key effects of climate change. A shift to severe individual storms and fewer weak storms is predicted with greater contrasts between wet and dry regions and seasons, though with considerable spatial variation, i.e. precipitation increases at high- and moist midlatitudes and decreases at dry mid-latitudes and in subtropical and semi-arid regions (IPCC: Collins et al. 2013).

Changes to catchment precipitation will influence riverine flow, which drives nutrient-enrichment of food webs, changes in habitat quality/availability and alteration of fish behaviour in estuaries (see: Gillson 2011), and are likely to influence fisheries catches by affecting abundance, productivity and catchability (Robins et al. 2005, Jaureguizar et al. 2016). Nutrient transport from upstream catchments is a key driver of estuarine and coastal productivity (see Section 7.2.1) and enhances recruitment (the largest contributor to fisheries catches), survival and growth (Grimes 2001, Darnaude et al. 2004, Breitburg et al. 2009a). Conversely, periods of low riverine flow have seen reductions in landings of target species in fisheries ranging from tropical estuaries (Gillson et al. 2009) to temperate coastal zones (Erzini 2005). Increased flows in some regions may therefore result in improved catches by stimulating recruitment and growth, while decreased flows in other regions may reduce catches.



Figure 7.9 Links between potential climate change effects and fisheries catches. Dashed link indicates changes unless thresholds are exceeded.

Timing of riverine flows may also affect productivity and fisheries catches, e.g. unseasonal summer storm flows delivering high levels of nutrients may cause algal blooms, eutrophication and losses of benthic vegetation that deplete oxygen, reduce habitat and food, and can cause fish kills (Bricker et al. 1999, Cloern 2001). Adverse effects of nutrient enrichment are likely to have greater impact in estuaries with long residence times, e.g. coastal lagoons and systems periodically closed by mouth sedimentation (Bricker et al. 2008). Reduced flows can prolong closures and isolation from marine habitats, reducing connectivity and recruitment, and influencing population dynamics, e.g. favouring freshwater and estuarine-breeding species (Sheaves & Johnston 2009, Vivier et al. 2010).

Low flows and rising temperatures lead to evaporation, causing falling water levels and altered salinity regimes and further affecting species distributions, connectivity and survival (Hoeksema et al. 2006, Zampatti et al. 2010, Gillanders et al. 2011a). Conversely, surge flows, as may be associated with more frequent extreme weather events, may provide important cues for reproductive activity and migrations in fishes that enhance catchability in fisheries (Loneragan 1999). Surge flows also increase turbidity that may favour recruitment in some species by providing refuge from predation (Blaber & Blaber 1980), while also causing reductions in habitat and recruitment in other species by driving oxygen availability below species-specific tolerances (González-Ortegón et al. 2010). Effects of salinity or turbidity (and other variables discussed below) may be greater where they coincide with spawning- or hatching seasons; eggs and larvae are particularly vulnerable due to lack of motility that limits their ability to seek refuge from adverse conditions.

In addition to precipitation changes, warming temperatures are a significant effect associated with climate change, yet impacts of warming on fisheries catches will likely vary among species and systems. Warmer temperatures are likely to drive increased growth rates in many species unless thermal tolerances are exceeded and so long as food is not limiting (Rombough 1997). This could result in increased growth, survival and recruitment. For example, incubation at warmer temperatures (31 °C) favoured muscle differentiation and growth in *Lates calcarifer* hatchlings compared to cooler temperatures (26 and 29 °C) (Carey et al. 2009). Survival also increased with temperature in Atlantic salmon leading to greater returns to North American estuaries (Friedland 1993, Friedland et al. 2003). However, warming could also lead to decreased recruitment if faster growth drives larval or juvenile energy demands that cannot be met and lead to starvation (Gehrke et al. 2011). This may be particularly true of slow-growing species, where energy consumption and feeding will likely increase with temperature, but growth may be slowed by high energy costs associated with increased respiration (Dijkstra et al. 2013).

Although warming may increase growth and thus catches of temperature-tolerant species, catches may be negatively impacted if physiological thresholds are exceeded. High temperatures cause oxidative stress in fish with potential to drive cellular malfunction (Lushchak & Bagnyukova 2006, Madeira et al. 2013). Such effects appear to be species-specific and most likely to occur when thermal optima are exceeded (Vinagre et al. 2012). As ectotherms, most estuarine fishes are incapable of regulating body temperature and thus rely on moving to thermal refuges when temperatures approach upper limits. Where topography allows, warming is likely to drive shifting distributions with fish movement toward cooler, potentially deeper, parts of estuaries and may disproportionately affect smaller species with faster life cycles (Perry et al. 2005). Importantly, thermal tolerance can be decreased by exposure to contaminants (Patra et al. 2007, Sokolova & Lannig 2008) or high salinity (Jian et al. 2003). Flounder Platichthys flesus, from the heavily contaminated Seine Estuary had lower heat tolerance and poorer condition when exposed to elevated temperatures than flounder from a moderately contaminated estuary (Lavergne et al. 2015). In addition, higher temperatures can modify contaminant chemistry and enhance toxicity while also increasing uptake through increased ventilation rates in fish (Kennedy & Walsh 1997, Schiedek et al. 2007). Therefore, fish in estuaries with large amounts of contamination or hypersalinity may be particularly vulnerable to temperature effects.

In addition to direct physiological effects, indirect effects of warming on estuarine ecology may affect fisheries catches. Loss of benthic vegetation in estuaries removes food and habitat for immature age classes of many fish species and is likely to reduce recruitment to fisheries (Costa et al. 2002). Warming may cause such losses, particularly at the limits of species' ranges, e.g. large diebacks of eelgrass *Zostera marina* occurred in Chesapeake Bay near the southern limit of its range due to increased water temperatures (Moore & Jarvis 2008). Warming can also enhance eutrophication, i.e. elevated nutrient loads, stimulating macrophytic growth whose decomposition depletes oxygen levels and may alter food web dynamics, depleting abundance of larger piscivores, increasing abundance of small omnivores and leading to removal of grazing zooplankton that control macrophytic growth (Gyllström et al. 2005, Teixeira-de Mello et al. 2009).

Warm temperatures also stimulate growth and reproduction of cyanobacteria that can cause hypoxia and toxic blooms (Robson & Hamilton 2003, Paerl & Huisman 2009). Moreover, warming can itself release nutrients from sediments (e.g. phosphorous) by increasing mineralisation rates and stimulating microbial activity (Jensen & Andersen 1992). Hypoxia can affect physiological processes influencing survival, reproduction and growth (Cech et al. 1984, Roussel 2007), while hypoxic zones in estuaries may also act as barriers to connectivity, interrupting reproductive migrations and limiting recruitment or even causing local extirpations of fisheries species (Weisberg et al. 1996, Jones 2006, Maes et al. 2007).

Increases in sea level and atmospheric  $CO_2$  may also affect estuarine-related fisheries and exacerbate physiological effects of warming by altering salinity and pH. Estuaries provide favourable growing conditions for fish because fish growth is strongly linked to salinity, with moderate salinities (8–20) often resulting in optimal growth, and estuarine salinity gradients provide opportunities for fish to seek areas within optimal ranges (Boeuf & Payan 2001). Salinity affects physiology via a trade-off whereby oxygen consumption increases when osmoregulatory efficiency is exceeded (Swanson 1998, Sardella & Brauner 2007). For example, metabolic activity in Mozambique tilapia *Oreochromis mossambicus* increased up to 20% following a shift from iso-osmotic conditions (12%) to 75% seawater (Morgan et al. 1997).

Sea level increases that inundate estuaries may also cause direct physiological effects that impact fisheries catches by affecting fish distributions within estuaries, possibly favouring marine species, and limiting growth due to increased allocation of metabolic energy to osmoregulation. Sea level may also influence catches indirectly by affecting estuarine wetlands, e.g. saltmarshes, that provide important ecosystem services, e.g. food and shelter for larval fish, as well as waste processing, e.g. nitrogen removal (Costanza et al. 1997). Projected sea level rises of 0.3–1 m this century (IPCC: Meehl et al. 2007) are expected to submerge 20–60% of coastal wetlands, causing reductions in ecosystem services (Nicholls et al. 2007) and potential loss of critical habitat (Drinkwater et al. 2010). Losses of wetland habitats, ecosystem services and societal goods and benefits are expected to be most extreme at lower and upper salinity ranges of estuaries (Craft et al. 2009), though rapid accretion may partially offset losses where local geomorphology and shoreline development allows inland migration of wetlands (Kirwan et al. 2010). Alternatively, estuaries with high human population densities and associated construction of structures designed to contain estuarine areas typically prevent inland migration (i.e. coastal squeeze) (Kirwan et al. 2010).

Marine waters have absorbed around one-third of anthropogenic CO<sub>2</sub> emissions (Sabine et al. 2004); however, water chemistry is altered in the process, reducing water pH and saturation of carbonate ions with a range of potential effects on aquatic fauna. In addition to atmospheric CO<sub>2</sub>, estuaries may face increased risks of acidity where catchments are subject to drainage of acid sulphate soils (Sammut et al. 1996). Effects of acidification on fisheries catches are difficult to predict due to the specificity of effects and complexity of ecological interactions involved (Le Quesne & Pinnegar 2012). While reduced access to carbonate ions may be harmful to calcifying organisms, e.g. fish prey items including many molluscs and plankton, low pH may directly affect fish physiology (Fabry et al. 2008, Heuer & Grosell 2014) and behaviour (Munday et al. 2009, Nilsson et al. 2012). Such effects include changes to metabolism (Munday et al. 2010), larval brain function and swimming performance (Domenici et al. 2012), and olfactory recognition of suitable habitat (Munday et al. 2009), prey (Cripps et al. 2011), conspecifics and predators (Dixson et al. 2010). Given the complexity of measuring impacts of these effects at the population and community levels, effects of low pH on fisheries catches remain unclear. Impacts on recruitment appear possible due to auditory and olfactory deficits; however, as in the case of other climate effects, speciesspecific tolerances may lead to more resilient species profiting, thus altering catch compositions.

Predicting effects of climate change on fisheries catches can thus be seen to face complexities, e.g. in accounting for future climatic conditions and interactions among them, physiological tolerances of numerous species, population dynamics (e.g. predation and competition), effects of range shifts and invasive species and physico-chemical variation among locations. Despite these challenges, many models suggest fisheries catches may tend initially to increase towards the poles and decrease at lower latitudes with thermally driven range shifts (e.g. Cheung et al. 2010, Blanchard et al. 2012, Barange et al. 2014), though more regionally mosaic variations in catches have also been predicted (Free et al. 2019). There is potential that declines in fisheries species with narrow physiological tolerances may be partially compensated by growing stocks of more resilient species, e.g. due to reductions in competitors and/or predators (Lenoir et al. 2011,

Serpetti et al. 2017). Given these likely changes to yields and composition (and thus value) of fisheries catches, it is prudent to consider their potential socio-economic effects and management implications for fisheries-based communities.

#### 7.4.2 Socio-economic effects and management implications

Global fisheries production was around 171 million tons in 2016, of which over 90 million tons derived from wild capture fisheries, providing over 20% of daily protein intake for over 3.2 billion people (with the highest proportions of fish consumption in developing countries: FAO 2018). Given the contribution of estuaries to both estuarine- and coastal fisheries (Drinkwater & Frank 1994, Grimes 2001, Lellis-Dibble et al. 2008, Blaber et al. 2022), climate effects on estuaries have the potential to heavily affect food security and fishery-related incomes on a global scale (Peck et al. 2020). These impacts are likely to be greatest in developing areas, e.g. Africa, Latin America and the Indo-Pacific, where per capita GDP is low or reliance on fish for protein is greatest (Allison et al. 2009). Communities most sensitive to socio-economic impacts will likely be characterised by high dependence on fishing-related industries, low levels of education, and low per capita income or asset bases that limit adaptive capacity to overcome periods of scarcity or diversify into other industries (Robinson et al. 2013). In addition, developing regions most dependent on fisheries often lack structural diversity in their economies, being highly dependent on a small range of industries. For example, recreational fishing and diving are important components of tourism income in many developing countries but are also susceptible to the same climate effects as fisheries through loss of habitat (e.g. coral bleaching), changes in biodiversity and access to iconic species whose distributions may change (Uvarra et al. 2005, Brander et al. 2007, Ruckelshaus et al. 2013). In these regions, management should focus not only on mitigating climate effects on fisheries catches but also on strategies to help vulnerable communities adapt to changing fisheries landscapes. For example, enhancing adaptive capacity by improving education, organising collective resources to augment physical capital, and providing assistance to diversify income sources may help reduce socio-economic vulnerability to climate effects on fisheries (Islam et al. 2013). Resettlement or migration may be unavoidable in certain circumstances, e.g. in the face of rising sea levels. However, this brings with it the risk of breaking community bonds that could decrease adaptive capacity in vulnerable communities (De Silva & Yamao 2007).

Mechanisms by which fisheries may be affected by climate change primarily involve changes to yield and composition (value) of catches due to gain, loss or shifting of suitable conditions for fisheries species (Roessig et al. 2004). Both range shifts and changing the species abundances could lead to opportunities as well as threats. Where opportunities arise, changes in targeted species may occur. In these cases, access to new gears or financial resources for their purchase should be planned for and could be assisted by subsidies or insurance schemes to help cover costs of re-equipping vessels (Broad et al. 1999, Badjeck et al. 2010). Catch diversification could spread risk over a wider number of techniques and target species, thus relieving pressure on heavily targeted or highly vulnerable stocks. More frequent extreme weather events have potential to reduce fishing days as well as damage vessels, equipment and infrastructure used in the landing, processing and transport of catches (Westlund 2007, Badjeck et al. 2010). Adaptive management measures, e.g. flood mitigation infrastructure, have the potential to impact catches by altering riverine flows that drive productivity (Sklar & Browder 1998), impede connectivity (Halls 1998, Boys & Williams 2012) or degrade fringing vegetation (Pollard & Hannan 1994).

Direct management should consist of measures that mitigate climate effects on fisheries catches. Riverine flows should be maintained where possible by limiting abstraction or diversion of water in upstream catchments. Flood mitigation works should be engineered so as to allow natural flows and minimise obstruction

to migrating fish as far as possible. Where barriers to connectivity exist, measures should be considered to restore fish movement, e.g. fishways. Critical habitats, e.g. nursery areas, should be conserved as the foundation of fisheries productivity and enhanced where possible. Fringing vegetation in particular (e.g. salt marshes and mangroves) should be conserved with priority (e.g. by avoiding injudicious shoreline development) due to its function as nursery habitats, stabilising agents against erosion and sea level rise, and waste removal sinks. Factors with potential to exacerbate climate effects, e.g. contaminant inputs that compound physiological stress or nutrient loads that drive eutrophication, should also be addressed to minimise cumulative or synergistic effects. The need for foresight and thoughtful planning for future management is thus clear and should be reinforced by further study on the impacts of climate effects on estuarinedependent fisheries to better understand and implement responses to climate change (see Elliott et al. 2022, on management of estuarine fish and fisheries).

# 7.5 Case studies

The impacts of global change in any given estuary will vary linked to overarching biogeographic realms, and how large-scale patterns drive changes to local physical, chemical and ecological processes. Just as environmental filtering governs the patterns of taxonomic diversity in estuarine fish communities (Henriques et al. 2017b), the relative impacts of global changes on estuarine fishes will not be identical or equally distributed worldwide. Moreover, large-scale, regional and local features and processes can impact what may be similar estuaries differently. But one key aspect to consider is that the responses of local fish communities will be determined by the sensitivity/tolerances of the species and estuarine habitats that occur (see Able et al. 2022), as well as their adaptability and scope (or opportunity) to respond and persist within each estuary.

# 7.5.1 Arctic

Arctic estuaries are unique in that for part of every year they are covered by ice sheets extending onto the shelf that creates a fresh-seawater mixing zone in the Arctic Ocean, which itself has been labelled a vast estuary with freshwater input second only to the Amazon delta (Macdonald 2000). Seasonal oscillations between freezes and thaws cause large fluxes in salinity; salinity increases with the export of sea ice in some areas to produce hypersaline brine and in other areas decreases with melting ice of terrestrial origin. The latter derives to a large extent from the catchments of Arctic rivers flowing into estuaries whose characteristics are heavily influenced by the unique polar climate. The Mackenzie River is Canada's longest river, and second in length only to the Mississippi in North America, flowing 1738 km in a north-westerly direction from Great Slave Lake through boreal forest and tundra before discharging into the Arctic Ocean via a large delta complex. With the Ob, Lena and Yenisei, the MacKenzie is one of the largest arctic estuaries globally and thus provides a useful subject for case study.

In addition to the delta, the nearshore area of the Beaufort Sea into which the MacKenzie discharges has long been referred to as the MacKenzie Estuary due to the substantial freshwater input there (Paulic & Papst 2013). As with other Arctic estuaries, the MacKenzie has very limited winter flows (when its estuary is covered by 2m of ice) and high summer flows; around 20% of annual flow occurs in June alone (Macdonald 2000). These flows occur through easily eroded landscapes, delivering high levels of suspended sediment causing high turbidity and deposition processes that shape the estuarine delta complex. A combination of high turbidity and ice cover limits light penetration, giving rise to low phytoplankton

productivity (Bodaly et al. 1989). Nevertheless, despite typically arid Arctic conditions, sub-surface permafrost allows the occurrence of extensive summer wetlands that dominate the MacKenzie delta (Ford & Bedford 1987, Prowse et al. 2006) and contribute to its relatively high productivity compared to surrounding landscapes.

As noted in preceding sections (7.2.2), estuarine fish species richness tends to decline towards the poles compared to regions at lower latitudes (Blaber 2008); however, delta estuaries such as the MacKenzie provide some of the highest productivity and greatest species diversity among Arctic habitats (Prowse et al. 2006). The large seasonal variability in flow leads to large longitudinal shifts in the salinity gradient of the MacKenzie Estuary. During summer months, when flows peak, freshwater discharge pushes out onto the Beaufort shelf in plumes of low-salinity riverine discharge that are important drivers of larval fish assemblages, e.g. coastal species dominated by Pacific herring *Clupea pallasii* are associated with the intense plume, while estuarine species dominated by Arctic cod *Boreogadus saida* prefer more diffuse sections of the plume (Paulic & Papst 2013).

Periodically, fish biomass of the MacKenzie Estuary is also dominated by salmonids (Bodaly et al. 1989). As with other Arctic rivers, salmonids and other diadromous fishes spend much of their life cycles in marine habitats, from bouts of summer foraging to longer periods of extended growth and maturation, and thus provide a key ecological linkage between estuaries and adjacent freshwater and marine habitats (e.g. nutrient transfer). The estuary is home to anadromous salmonids, e.g. Arctic char *Salvelinus alpinus*, Arctic cisco *Coregonus autumnalis* and lake whitefish *Coregonus clupeaformis*, that migrate upstream to spawn prior to freeze up before moving downstream to the delta to overwinter (Bodaly et al. 1989). Larvae are likely flushed to sea by the flows of the spring thaw (Bond & Erickson 1985). This prevalence of migratory species leads to large spatial and temporal variations in abundance of fish biomass, which in turn causes large fluxes in nutrient transfer and periods of increased vulnerability to stressors, e.g. overfishing.

The unique characteristics of Arctic estuaries, e.g. high dependence on seasonally low flows to supplement low nutrient inputs and low productivity, and their important role as hotspots of species richness in Arctic landscapes should be assessed when considering management or alteration of Arctic river systems. Potential climate change effects include shorter periods of ice coverage and increased precipitation in Arctic areas (Macdonald 2000), which are likely to alter salinity and flow regimes and may see significant temporal and spatial shifts in the distributions of fish species in Arctic estuaries.

In addition to changing climate, construction of upstream hydroelectric dams presents potential risks for the MacKenzie Estuary. A proposed dam construction project was cancelled in 2010; however, such proposals continue to recur periodically. Dams have potential to reduce already limited flows and alter their timings, leading to changes in estuarine productivity (see Section 7.2.1). Given the already low productivity of the MacKenzie Estuary, any such developments should be made with careful consideration given to allocating flows to maintain natural ecological functioning. These considerations hold true not just for the MacKenzie Estuary but for Arctic estuaries more generally.

#### 7.5.2 Temperate northern Atlantic

The north-eastern Atlantic from the British Isles to the Iberian Peninsula encompasses a wide temperature gradient, from cool-temperate to warm-temperate biogeographic regions, with areas in the southern Iberian Peninsula presenting a semi-arid, Mediterranean-type climate with reduced precipitation and hot summers. Throughout the region, sea shelf warming is expected to continue to rise, with climate projections predicting changes in regional hydrometeorological processes, such as increasingly clustered rainfall events in the UK, or increasing drought frequency in the Iberian Peninsula, modulated by variations in

large scale climatic patterns such as the North Atlantic Oscillation (e.g. Trigo et al. 2004, Vicente-Serrano et al. 2014, Robins et al. 2016).

Over recent decades, rising water temperatures have led to surging occurrences of subtropical ichthyofauna along the latitudinal gradient of the North Atlantic coast. As fish species overcome overwintering bottlenecks (Figueira & Booth 2010), estuarine communities are mirroring marine environments and gradually showing increased numbers of traditionally lower latitude species, whilst other cool-water species are displaced poleward (e.g. Cabral et al. 2001, Henderson 2007, Heath et al. 2012, Baptista et al. 2015, Robins et al. 2016, Peck et al. 2020). Evidence for a broad range climate-induced biogeographical shift in estuarine fish communities along the north-eastern Atlantic coast from Portugal to Scotland was provided by Nicolas et al. (2011). Comparisons of mean latitudinal species distributions supported the northward migration of estuarine species over 30 years (1970s–2010s) (e.g. common eel Anguilla anguilla), with trends further corroborated by the occurrence of numerous subtropical species above historical latitudinal limits (e.g. round sardinella Sardinella aurita). This increase in warm-temperate, tropical species, and retreating cold-temperate species has been evident along the Portuguese coast for species that use estuaries opportunistically (e.g. sprat Sprattus sprattus – decreasing), as well as among estuarine-residents and species that use estuaries as nursery areas (e.g. Senegalese sole Solea senegalensis - increasing) (Cabral et al. 2001, Vinagre et al. 2009). It is also seen for salmon along the Spanish Atlantic coast (Nicola et al. 2018). In waters surrounding the British Isles, Peck et al. (2020) highlighted climate-related temperature effects in the poleward shifts of estuarine nursery fish species such as pollock Pollachius pollachius and fivebeard rockling Ciliata mustela.

In addition to temperature, changes to river flow will play a pivotal role in shaping fish assemblages in temperate Atlantic estuaries as in other regions around the globe. For the Gironde Estuary in France, Nicolas et al. (2010) highlighted how shifts in environmental conditions significantly affected the abundance of a variety of species, with enhanced presence of marine stragglers in periods of reduced river flow. In contrast, in the outer Severn Estuary in the UK, total fish species declined with increasing salinity (Henderson 2007). Overall, numerous studies have shown how the occurrence and abundance of fishes in estuaries are related to environmental variables, including river flow and regional scale climatic patterns (Chícharo et al. 2006, Costa et al. 2007, Henderson 2007, Nicolas et al. 2010, Pasquaud et al. 2012). These studies provide a clear indication of the potential responses of North Atlantic estuarine fish assemblages to environmental change, but whilst general trends follow Henriques et al. (2017a), these trends are still subject to species- and site-specific responses according to physiological scope, estuarine conditions, habitat availability or estuarine geomorphology (Le Pape et al. 2003, Dolbeth et al. 2010, Vasconcelos et al. 2011). Ultimately, recruitment and species persistence can differ among estuaries and are dependent on the interplay between influencing factors.

A growing concern is how climate change, and in particular alterations brought by extreme events (e.g. droughts, floods), may affect nursery-role function and the contribution of European estuaries to coastal areas, where many key commercially important marine species are sustained by juveniles deriving from estuarine habitats (e.g. Reis-Santos et al. 2013, Tanner et al. 2013). In the Iberian Peninsula, a key issue is the expected increase in frequency and severity of drought (Vicente-Serrano et al. 2014). Studies in numerous estuaries have shown how drought and reductions in river flow impact estuarine fish assemblages, and can compromise estuarine nursery role via a decrease in the abundance of juveniles (Chicharo et al. 2006, Costa et al. 2007, Martinho et al. 2007, Baptista et al. 2010, Dolbeth et al. 2010). This reduction is often linked to reduced river plumes, and cues for larval recruitment, as well as sub-optimal environmental conditions or food availability (Vinagre et al. 2009).

Overall, production in drought-affected estuaries decreased with decreasing river flow but patterns were not always consistent (Dolbeth et al. 2010). In the Mondego Estuary, severe drought reduced juvenile densities, and thus nursery role, but as generally expected overall species richness increased. However, in the Tagus Estuary, Costa et al. (2007) observed increased densities of estuarine species as well as of those that use the estuary as a nursery in drier years. This resilience is likely due to the size and geomorphology of the Tagus, compared with the Mondego, where species can move elsewhere in the estuary, but we cannot exclude differences in the severity of the conditions between studies as they do not overlap temporally. Similarly, in the Gironde (a large estuarine system in south-western France), a combination of temperature-induced shifts in species distributions, associated with decreases in river flow and raised salinities, increased juvenile abundances, and therefore Pasquaud et al. (2012) argued that climate change may positively influence this estuary's nursery role.

In a recent study, Robins et al. (2016), summarised the main climatic drivers, and expected impacts of climate change in UK estuaries. The anticipated increase in river flows in winter may enhance high turbidity and hypoxia events, potentially deterring larval recruitment or leading to flushing out of fishes from within these estuaries. Linked to changes in river flow, and consequent variations in the inputs of allochthonous materials and indirect effects on prey species, estuarine food webs that can play determinant roles in shaping the persistence of many fishes within estuaries are also likely to be affected, together with a myriad of other factors previously discussed throughout this chapter. One key issue in estuaries around the UK, and other regions of the North Atlantic, is the concomitant effects of climate change-related sea rise with land sinking, which result in a greater relative sea level rise in the region, and therefore increased estuarine flood risk and changes to geomorphology and consequently habitat availability, whilst in others (e.g. Scotland, Sweden) isostatic rebound (land mass rise) dampens this effect (Ducrotov et al. 2000, Shennan & Horton 2002, Robins et al. 2016). Overall, one major take-home message of this study was that we should take into consideration the complexities of local systems before applying general climate predictions. Yet, as baseline and historical knowledge remains lacking for many estuarine systems, we should strive to build long-term monitoring strategies that focus on the river-estuary-seascape as a whole, to support risk prediction and mitigatory strategies in the near future.

#### 7.5.3 Temperate northern Pacific

The 12652km shoreline of the Pacific Coast of the conterminous USA (California, Oregon, and Washington) is home to at least 440 estuaries encompassing over 735000 ha of surface area (Emmett et al. 2000, Brophy et al. 2019). The diverse topography and geology of the landscape interact with prevailing regional climate patterns to produce many functional types of estuaries, including large and small bays, fjords, river deltas, coastal creek mouths, and closed (bar built) and open lagoons (Emmett et al. 2000, Heady et al. 2014, Brophy et al. 2019, Clark & O'Connor 2019). Climate varies from Mediterranean-type (seasonally relatively cool and wet in winter versus relatively warm and dry in summer) in the south, to temperate (predominately cool and wet) in the north. Physico-chemical properties of the estuaries are driven primarily by a combination of marine inputs via mixed semidiurnal tides (high and low tides of different sizes each lunar day) and freshwater inputs from local catchments.

The spatio-temporal dynamism of Pacific coast estuaries provides a variety of habitat functions for fishes, ranging from temporary habitat for transient species to permanent habitat for resident species (Monaco et al. 1990, Allen et al. 2006, Hughes et al. 2014). The seagrass *Zostera marina* is a notable component of the seaward portions of these estuaries and provides important nursery habitat functions for fishes and invertebrates (see Able et al. 2022).

Pacific coast estuaries respond to both inland and oceanic climate variability (Reum et al. 2011, Feyrer et al. 2015). There is strong regional variability associated with south-north oceanic and climatic gradients


**Figure 7.10** (a) Fishing boat in San Francisco Bay Estuary (Photo: courtesy of Matthew J. Young) and (b) aerial image of the upper estuary in Suisun Bay, California (Photo: courtesy of Carson Jeffres).

because most of the catchments are relatively small (Hickey & Banas 2003). The three largest estuaries – Columbia, San Francisco and Puget Sound estuaries – have large inland catchments that encompass and integrate a broad expression of climate patterns. There is also extensive human alteration and degradation of the estuaries and their catchments, which can force systems into novel ecological regimes and impact their capacity to respond naturally to climate variability or climate change (Borde et al. 2003, Mac Nally et al. 2010, Thomson et al. 2010, Mahardja et al. 2017, Toft et al. 2018).

The San Francisco Estuary (together with its upstream Sacramento-San Joaquin Delta) is the largest (by historical surface area) and most well-studied of the estuaries (Cloern & Jassby 2012, Brophy et al. 2019) (Figure 7.10). It is located in central California, which represents the southern range limit of multiple marine and anadromous species that use the estuary. The system has been highly altered by human activities associated with agricultural and municipal development. Among them include the reclamation of over 90% of historic tidal wetland habitat and the extraction of freshwater from the upper estuary to support California's growing economy (fifth largest in the world) and population (> 25 million) (Whipple et al. 2012). It is generally considered the most highly altered and invaded estuary on the planet (Conomos 1979, Nichols et al. 1986, Cohen & Carlton 1995). Native fish populations have plummeted as these changes have occurred (for details see Moyle & Stompe 2022), resulting in 14 migratory or resident fishes protected under state or federal endangered species laws. These declines have stimulated rigorous scientific investigations to understand the underlying mechanisms, including the effects of climate (Sommer et al. 2007, Cloern et al. 2011, Cloern & Jassby 2012). Notably, effects of climate variability on San Francisco Bay Estuary fishes and primary productivity are apparent despite the presence of such strong local internal drivers and perturbations that could confound them (Lehman 2000, 2004, Cloern & Dufford 2005, Cloern et al. 2010, Cloern & Jassby 2012, Feyrer et al. 2015).

Projections from down-scaled global climate models suggest that the San Francisco Estuary will likely experience increased water temperature, sea level rise, salinity intrusion, changes in the amount and timing of freshwater inputs, and increased average annual water-year outflows (Knowles & Cayan 2002, 2004, Cloern et al. 2011, Healey et al. 2016, Thorne et al. 2018). These changes will potentially affect fishes in many indirect and direct ways. Anticipated rates of sea level rise will likely outpace the rate at which extant wetlands can adjust and may contribute to further loss of wetland habitat to support fishes (Thorne et al. 2018). Changes in timing and magnitude of freshwater inflows can affect inundation of floodplains

important to reproduction of some fishes (Cloern et al. 2011). Changes in water temperature may change suitability of habitat, timing of critical life history stages (phenology) and fish behaviour (Brown et al. 2013, Brown et al. 2016, Davis et al. 2019). Generally, these potential broad-scale changes to the system are likely to compress in time and space the types of physico-chemical habitat favourable for supporting native fish species while promoting conditions that favour non-natives (Moyle 2008, Feyrer et al. 2010, Cloern et al. 2011, Cloern & Jassby 2012, Brown et al. 2013, Brown et al. 2016). Implementing effective climate change adaptation strategies to conserve and manage fishes and fisheries, while also addressing human water needs, in the already highly altered San Francisco Estuary will be a considerable challenge for resource managers.

## 7.5.4 Tropical Atlantic

Extreme weather events in South America have become more intense and more frequent in the last 50 years. In north-eastern Brazil, 16 out of 25 years between 1991 and 2016 experienced drastic reductions in precipitation (Marengo et al. 2018). The meteorological and oceanic mechanisms that lead to circulation and rainfall changes responsible for drought in north-eastern Brazil include the occurrence of El Niño, and/or a latitudinal shift of the Intertropical Convergence Zone (ITCZ) driven by major forces from the Northern Hemisphere climate (Utida et al. 2019).

Since 2010, the semiarid portion of north-eastern Brazil has been experiencing one of the most protracted and intense droughts in decades (Pereira et al. 2014, Erfanian et al. 2017). In this region, the climate-physiographic properties of river basins vary greatly from estuarine and adjacent coastal zones to their headwaters. Lower parts of basins are dominated by a humid climate with tropical forest ecosystems and habitats, while upper basins are characterized by semiarid climates and habitats (Costa et al. 2016). For example, two-thirds of the Mamanguape River basin (in the state of Paraíba, North-Eastern Brazil) is under the influence of a semiarid climate, which causes an intermittent flow regime in most of the basin; but the 25 km long tidal estuary is perennial and influenced by smaller tributaries from wetter climate areas (da Silva et al. 2018). Overall, the functioning of Mamanguape Estuary is strongly influenced by the magnitude and timing of freshwater run-off reaching the estuary, and the freshwater run-off largely determines the salinity distribution within the system. Estuarine conditions during the dry season in 1997 and 2000 indicated that salinity means in the Mamanguape Estuary were consistently lower than those observed in more recent surveys conducted in 2011 and 2014 during an extended drought period (Oliveira & Pessanha 2014, da Silva et al. 2018). The authors inferred that for this reason, many fish species moved across the estuarine systems in response to salinity changes, associated with a decrease in both precipitation intensity and frequency. Therefore, combined with other environmental variables, expected increases in salinity could be an important predictor of habitat selection by fish assemblages in estuaries in northern Brazil under future drier climate conditions.

Relationships between fish abundance and freshwater flow in Mamanguape Estuary have also been found. For example, a large rainfall event in 2011 contributed to the 'estuarisation' of the adjacent beaches as a result of increased river discharges (Oliveira & Pessanha 2014). The resultant expansion of the estuarine zone to coastal areas promoted a shift in coastal fish assemblages. During this period, the lowest abundance of resident fishes typically found near sandy beaches (e.g. *Polydactylus virginicus, Lycengraulis grossidens* and *Trachinotus carolinus*) was recorded, whilst the abundance of Clupeiformes increased significantly (*Rhinosardinia bahiensis, Lile piquitinga, Amblygaster clupeoides, Anchoviella lepidentostole* and *L. grossidens*). In contrast, in 2012 low precipitation was not sufficient to reduce salinity in the lower reaches of the estuary or expand the estuarine area to the coastal area, and thus a higher abundance and

intrusion of marine species (e.g. *Caranx hippos*, *P. virginicus*, *T. carolinus*, *T. falcatus* and mugilids) was found. The increase in marine species was explained by the more uniform salinity conditions in 2012 between the coastal zone and estuary, i.e. 'marinization', a phenomenon that occurs in numerous estuaries globally (Oliveira & Pessanha 2014).

The strong presence and distribution of estuarine-dependent and marine species throughout the Mamanguape Estuary may result from seawater intrusion and tidal influence (Dolbeth et al. 2016, da Silva Lima et al. 2020). Tidal influence is fundamental in driving the transport of eggs and larvae into the upper regions of the estuary. Species guild dominance in this area is associated with river flow, particularly during the dry season. In 2015 and 2016, rainfall was reduced and concomitantly the transport of marine larval fish into the upper estuarine region occurred. These findings also show that *Elops saurus* (leptocephalus larvae), Engraulidae larvae and *Rhinosardinia bahiensis* used the upper reaches of the estuary only during their early life stages. Greater representation of pre-flexion larvae of families of marine origin, such as Sciaenidae, Carangidae and Bleniidae, in the upstream areas due to the tidal stream transport, suggested that the larvae moved vertically within the water column during the flood tide and were transported by convection through the salt wedge to the upper reaches of the estuary (De Silva & Yamao 2007, Dolbeth et al. 2010).

Reductions in precipitation also resulted in increased fish size in the lower regions of the Mamanguape Estuary, with largest individuals found during the dry season (e.g. *Diapterus auratus, Lycengraulis grossidens* and *Rhinosardinia bahiensis*). This evidence reflects the ontogenetic habitat shifts hypothesis, in which small individuals, such as larvae and juveniles, obtain an environmental refuge during the rainy season and appear to migrate to adjacent areas as they grow throughout the dry season (Clark & Pessanha 2015, Medeiros et al. 2018). This migration is particularly beneficial because fish can grow and develop rapidly, becoming less vulnerable to predation. The earliest evidence of this theory was provided by Clark & Pessanha (2015), who found large schools of *R. bahiensis* juveniles selecting sheltered sites in tidal creek habitats, while adults were found on intertidal mudflats. Araújo et al. (2016) also documented a greater proportion of juvenile mojarras (Gerreidae) in shallow waters than in deeper waters, which was interpreted as a size-dependent refuge response to increasing predation pressure.

Overall, the influence of rainfall on salinity and its effects on other environmental variables regulate the composition and distribution of the fish assemblages in this tropical estuary. This emphasizes the importance of seasonal changes in fresh water discharge for larvae and juvenile fishes, with salinity acting as the main environmental filter. Predicted changes in global climate for the region, and the effects of long-term droughts on recruitment processes and the distribution of fish populations, have the potential to affect both fish assemblages directly, as well as indirectly via the loss of estuarine habitats, which may ultimately lead to a decrease in estuary-associated species, which are highly prized by local fishers (Dolbeth et al. 2016).

## 7.5.5 Indo-Pacific

Estuaries form significant ecological zones for many fish species in the Indo-Pacific region. Rivers flowing through these zones are characterised by high monsoonal rainfall followed by extended dry seasons with limited outflows (Dudgeon 2005). These systems thus tend to have high monsoonal sediment loads which give rise to extensive delta systems (Estelles et al. 2002). The delta systems provide a rich transition from freshwater to seawater and leading to productive fisheries with a diverse array of life history strategies (Nguyen et al. 2002). Furthermore, the majority of these systems occur in catchments where fish form a significant component of regional diets, contributing both protein and essential micronutrients for millions of people (Hortle & Suntornratana 2008). Thus, the connection between seawater and freshwater is

extremely important ecologically and socially and, interestingly, the most significant fisheries in the region are dominated by anadromous and catadromous species (Honda et al. 2016, Hossain et al. 2019).

Climate change is expected to affect south-east Asia in several ways. These include temperature increases of 2–4 °C, more intense rainfall events (1–12% increase in precipitation) including cyclones, longer drought periods, and sea level rise predictions of 0.2–0.6 m (IPPC 2007). For instance, Myanmar is widely considered one of the most vulnerable countries in the world to the impacts of climate change for people and biodiversity (Horton et al. 2017). Climate change is expected to exacerbate existing threats to anadromous fish in Myanmar due to direct threats such as overexploitation of fish stocks and habitat loss, and indirectly through climate change impacts on humans and their reliance on intact ecosystems (Rao et al. 2013). By 2100, Myanmar could experience a 0.2–0.5 m sea level rise (Rao et al. 2013). A 0.5 m sea level rise would result in the shoreline along the Ayeyarwady Delta advancing inland by up to 10 km. In addition, irrigated rice inland in Myanmar is expected to also be severely impacted by changes in rainfall and temperature (Deb et al. 2014).

Supported under current government policy, polders and sluices may proliferate to combat sea-level rise and protect cropping areas, which in turn will impede anadromous fish being able to enter freshwater habitats to complete their life cycles (Conallin et al. 2019). For example, the Hilsa shad (*Tenualosa ilisha*) is an important species that supports capture fisheries throughout Myanmar (Hartmann & Ingthamjitr 2007) (Figure 7.11). The Hilsa is an important part of Burmese culture and an important feature in the catches of fisheries lease holders and market stalls (Figure 7.11). After maturation at sea, adults can migrate several thousands of kilometres upstream to reach the gravel beds which form important spawning grounds (Rahman & Cowx 2006). Hilsa have experienced a significant decline over the past two decades with the proliferation of tidal barrages constructed to provide a more stable source of freshwater to support rice crops blocking access to spawning grounds (Dubois et al. 2019). In addition, in inland rivers, especially within the dry zone, it is likely that there will be an increase in water abstraction during the dry season for cropping and an increase in infrastructure to support abstraction such as dams and weirs. The alteration of flow due to damming and the loss of water in the rivers due to abstraction could also have severe consequences for anadromous fish.

There is a paucity of information on migratory pathways of fish species that are important to livelihoods. If unplanned river and wetland infrastructure occurs, it is expected that migratory species may decline in the next decade as development continues. Studies have shown that if corridors are blocked through infrastructure such as hydropower dams, poor flow management, or irrigation infrastructure on rivers and wetlands, species richness and abundance will decline to unsustainable levels (Baumgartner & Wibowo 2018). Dam development is in a 'boom' period in south-east Asia, threatening biodiversity and local community livelihoods. Without adequate consideration of either environmental data, ecology or migratory corridors, policy formation lacks key knowledge on which areas are best suited for expansion of development, which areas should be protected, and which areas may be best to implement remediation of existing infrastructure.

The Mekong River is an iconic river of the Indo-Pacific, flowing over 2500 km to the sea, and is one of the top 10 river systems around the world in terms of discharge (Baran & Cain 2001). The Mekong has remarkable fisheries productivity (Hogan et al. 2004). The first sale value of the fishery has been estimated at \$US17B per year, and 2% of the global fisheries catch (including marine) is harvested from the Mekong each year (Hortle 2007). As a monsoonal river, it also has a high silt load during the rainy season and flows into the South China Sea through an intricate series of canals in an elaborate delta system. One of the economically important species among the 1200 reported for the Mekong is the anadromous Krempfi catfish, *Pangasius krempfi*, often referred to as the Mekong salmon (Hogan et al. 2007) (Figure 7.11). Each



**Figure 7.11** (a) Hilsa shad *Tenualosa ilisha* displayed in a market in Pathein, Myanmar, and (b) krempfi catfish *Pangasius krempfii* in a market at Pakse, Lao PDR. Also shown, (c) bag nets and (d) trawl fishing in the Mekong Delta used to catch *P. krempfii*. (Photos: courtesy of An Vu).

year, thousands of adult fish migrate from the South China Sea some 600 km upstream to spawn. Adults that perform spawning migrations often exceed 1 m in length, and fetch high market prices so these migrations are actively targeted by fishermen with bag nets or by trawling (Cacot 2004) (Figure 7.11).

The main threat to the Krempfi catfish is hydropower development (Dugan et al. 2010). The Lao government recently approved the Don Sahong Hydropower Project; and for many years, the Cambodian government has been considering the completion of Sambor Dam. Both projects exist on the mainstem and will ultimately delay or completely block the Krempfi catfish spawning migration if adequate mitigation measures are not put in place. Provision of fish passage in both directions is an important consideration, both to allow upstream adult migrations to occur and for larvae to reach the sea and mature. Failure to adequately cater for the entire life cycle will ultimately threaten long-term persistence of this important large-bodied species. From a climate change perspective, long-term changes to hydrology, catchment runoff or seawater levels may alter the fundamental dynamics of migration routes for these catfish, with resultant extensive impacts on the livelihoods and incomes of communities that depend on them. Therefore, management strategies to protect Krempfi catfish migrations should consider future climate change scenarios in addition to the obvious challenges posed by river infrastructure. These examples highlight challenges in the response to climate change and maintaining connectivity between freshwater and estuarine regions within the Indo-Pacific region. There is presently an unprecedented boom in hydropower, irrigation and barrage construction which is seeking to provide water security for over 60 million people. However, it is important that these river development projects adequately consider potential impacts on fisheries resources and how development may interact with impacts derived from climate change. There are many more diadromous species potentially threatened by such development and there is a limited window to ensure fish with essential estuarine life history stages are considered in development projects (Mekong River Commission 2017).

#### 7.5.6 Temperate South America

The extensive  $(3.5 \times 10^4 \text{km}^2)$  and shallow (average depth <10m) estuary (35°S) of the Río de la Plata is the largest estuarine system in the temperate South American ecoregion (Barletta et al. 2010). The estuary, extending 320 km inland varies in width from 20 km (head) to 220 km (mouth), is characterized by microtidal variability (<1m), a semi-permanent salt-wedge regime (100 km of length with a strong vertical salinity gradient up to 10 units per meter), and by its high susceptibility to atmospheric forcing (Guerrero et al. 1997, Simionato et al. 2004, Meccia et al. 2013). The Río de la Plata basin hydrologic cycle, ruled by the South Atlantic anticyclone, is typically defined by a warm season (October–April) with an average rainfall of 5.5 mmd<sup>-1</sup> and a cold season (May–September) with average rainfall less than 2mmd<sup>-1</sup> (Caffera & Berbery 2006).

At interannual time scales, large changes in basin rainfall (Robertson & Mechoso 1998), predominant wind pattern (Meccia et al. 2013), intensity of salinity gradient (Acha et al. 2012), position of salinity and turbidity front (Nagy et al. 2002, Jaureguizar et al. 2007, Jaureguizar et al. 2016, Brugnoli et al. 2018) and sea level (Saraceno et al. 2014) are observed in the estuary, and all have been associated with El Niño/La Niña events. During El Niño events, an excess of spring rain increases run-off, with flows reaching up to  $87 \text{ m}^3 \text{ s}^{-1}$ , while under drought conditions associated with La Niña, run-off can fall as low as  $7.8 \text{ m}^3 \text{ s}^{-1}$  (Jaureguizar et al. 2016). Therefore, it has been suggested that in the south-western Atlantic region, altered basin rainfall and river flows associated with climate change may be more important factors in shaping estuarine environmental conditions than increases in temperature.

Although future scenarios contain some uncertainty, global climate models for the next 50 years predict a net increase of ~15% in rainfall and 2°C in atmospheric temperature (CICPCP 2017). Furthermore, the poleward expansion of the subtropical South Atlantic anticyclone belt (Barros et al. 2008) will increase the frequency of easterly and north-easterly winds (Simionato et al. 2004, Meccia et al. 2013). This scenario would see changes in freshwater discharge and wind patterns favouring downriver displacement of the salinity fronts, stronger salinity gradients, and a southward extension of the Río de la Plata plume with a northern retraction of the outer salinity front. In fact, run-off events over the past decade have already provided a snapshot of the expected predominant conditions that will occur in the future (see Jaureguizar et al. 2016). Biological studies during those events allow us to predict that climate change will affect fish recruitment, the spatial distribution of species, along with fisheries yields of commercial fisheries that operate in estuarine and coastal neighbouring waters.

The whitemouth croaker (*Micropogonias furnieri*), is a size-structured estuarine species of high economic and ecological importance within the estuary (Jaureguizar et al. 2003, 2016). Juveniles inhabit shallow waters (<10m) with low bottom salinity (<18), matching the maximum suspended particulate matter concentration (>50 mgl<sup>-1</sup>) (Jaureguizar et al. 2003, Camiolo et al. 2019), and are more abundant at intermediate run-off of ~26 m<sup>3</sup> s<sup>-1</sup> than at high (>28 m<sup>3</sup> s<sup>-1</sup>) or low run-off rates (<22 m<sup>3</sup> s<sup>-1</sup>). Adults occupy outer estuarine waters and migrate towards the inner estuary during the spring-summer reproductive season following



**Figure 7.12** Spatial distribution of life stages of *Micropogonias furnieri* and their relationship with the environmental variables in the Río de la Plata during spring: (a) distribution areas per sampling stations groups (inshore; middle; offshore) showing their age classes structure (Based on Simionato et al. 2008, Braverman et al. 2009, Acha et al. 2012), (b) turbidity spatial distribution (%WR, percentage of water reflectance; redrawn from Jaureguizar et al. 2003), (c) environmental preference per sex (M, male; F, female) and maturity stages (1, immature; 2, developing and partially spent; 3, gravid [with hydrated oocytes] or running; 4, spent; 5, resting) (redrawn from Jaureguizar et al. 2008) and (d) vertical diagram of hypothetical spawning ground (redrawn from Bava et al. 1996). PE, Punta del Este; PP, Punta It.

the bottom salinity signature (Macchi et al. 1996, Jaureguizar et al. 2003, 2008). Mature females concentrate to spawn at the bottom salinity front (Macchi et al. 1996, Jaureguizar et al. 2008), an area retentive of eggs and larvae (Simionato et al. 2008, Braverman et al. 2009, Acha et al. 2012) (Figure 7.12).

Research suggests that planktonic egg retention is associated with *Micropogonias furnieri* spawning location and run-off conditions, and the highest retention resulted from a combination of upstream spawning and low run-off (Acha et al. 2012). Therefore, future scenarios of higher freshwater run-off that promote a down-river shift of the salinity front are likely to impact the movements of spawning females

and increase the advection of spawned eggs and larvae to adjacent marine coastal areas, both negatively affecting reproductive success (Acha et al. 2012). Also, a greater spatial distribution of size assemblages will result in lower catchability of adults and increased juvenile catchability in the small-scale or artisanal fisheries of Samborombom Bay (Camiolo et al. 2019) and along the northern coast of the Río de la Plata estuary (Norbis et al. 1992). Similarly, climate change will affect other species that use the inner salinity front for spawning, such as other sciaenids (*Macrodon ancylodon* and *Pogonias cromis*) and other species, e.g. *Brevoortia aurea* (Acha & Macchi 2000) and *Ramnogaster arcuata* (Rodrigues et al. 2008).

The increasing E-NE wind frequency will intensify the surface/bottom estuary recirculation (Simionato et al. 2004, Meccia et al. 2013), producing a higher outflow of brackish (<28) surface waters (up to 10 m depth) and a corresponding higher inflow of saltier (>30) bottom waters upstream. It has been demonstrated that vertical stratification of the water column determines the degree of intrusion of freshwater and marine species into the estuary (Jaureguizar et al. 2003, Jaureguizar et al. 2016); higher stratification will allow a higher intrusion of freshwater (e.g. *Pimelodus maculatus* and Loricariidae species) and marine taxa (e.g. *Cynoscion guatucupa*). At the same time, the E-NE wind prevalence will increase southward forcing of surface estuarine waters along the Argentine coast, driving greater intrusion of bottom marine waters into the estuary, increasing bottom salinities and affecting the southern extension of the salinity tongue and location of the outer salinity front (Jaureguizar et al. 2007, Jaureguizar et al. 2016). The outer edge of the salinity plume is an important mating/spawning area and key juvenile ground for many marine species (Jaureguizar et al. 2016). It also restricts the seasonal (e.g. Trichiurus lepturus in summer and Discopyge tschudii in winter: Jaureguizar et al. 2004) and interannual intrusion [(e.g. Prionotus nudigula: Jaureguizar et al. 2007) and (e.g. Dicospyge tschudii: Cortés et al. 2011)] of marine species associated with shelf waters within the estuary (Jaureguizar et al. 2006).

In general, it is expected that future oceanographic conditions (i.e. changes in salinity and, temperature gradients, wind fields and hydrological cycles at annual to decadal time scales) will affect fish assemblage composition in the region through the relocation of mating areas of chondrychthyans (e.g. *Mustelus schmitti, Squatina guggenheim*), and spawning areas of marine bony fishes (e.g. *Umbrina canosai, Cynoscion guatucupa, Percophis brasiliensis, Paralichthys patagonicus, Parona signata*), while changes to environmental conditions in areas used by juveniles may influence food availability and refuge from predators, thus influencing recruitment and growth (Jaureguizar et al. 2016). Survival and growth of juvenile age classes are among the most important influences on small shark species population growth (e.g. *M. schmitti*, Cortés et al. 2010); consequently, changing outer environmental conditions are likely to negatively impact shark populations through restriction of suitable habitats for growth and reproduction.

Overall, variations in species distributions are likely to diminish small-scale fishery profits by reducing availability of targeted fish. For example, a seasonally important small-scale gillnet fishery (Jaureguizar et al. 2015) with varying interannual fishery yield (Jaureguizar et al. 2016, De Wysiecki et al. 2017) exists along the southern estuarine boundary of the Río de la Plata that is strongly affected by the strength of the intrusion of estuarine waters. The best catches of some marine species (*C. guatucupa, P. signata*) occur before the reproductive period, whereas for other species good catches appear to be associated with favourable environmental conditions during cold months (*Squalus acanthias, Callorhinchus callorhynchus, Galeorhinus galeus*) or warm months (*T. lepturus*) (Jaureguizar et al. 2015). Therefore, the upper region of the Río de la Plata will affect the mix of species available on a seasonal basis, and reduce catches of marine species that migrate into nearshore waters during the reproductive season.

Despite the scarcity of long-term data, reported interannual (2002–2008) variability in Sea Surface Temperature (SST) is very small over most of Río de la Plata Estuary (<1%; Simionato et al. 2010), yet over the last few years there is an increasing presence of tropical and subtropical fish species (De Wysiecki et al. 2018,

Milessi et al. 2018, 2019) in the marine water adjacent to the Río de la Plata during the warm season (November–February), and coincident with a SST increase (0.5 °C in 2002–2018, (Milessi et al. 2018). Similar accounts and associations relating to ocean warming have been observed in the Uruguayan fishery landings (1973–2017, (Gianelli et al. 2019). Therefore, it is expected that higher SST scenarios in the RdIP will bring higher occurrences of fauna from warmer bioregions, change species movements and reproductive migrations associated with altered oceanographic conditions, and affect the CPUE and fishing calendar for the small-scale fisheries settled along the coast of Río de la Plata.

### 7.5.7 Temperate southern Africa

The temperate coastline of southern Africa extends approximately 3800km from Namibe (15.2°S) in Angola to the south-east coast (31.9°S) of South Africa (Potts et al. 2015). Coastal rainfall is very low along the cool-temperate Angolan and Namibian coastline (<20 mm per annum), with only one permanent estuary in this region (Potts et al. 2015). Whilst van Niekerk et al. (2020) identified 33 estuaries along the South African cool-temperate west coast, this number rises to 124 along the warm-temperate south and south-east coast, the majority (105) being temporarily open/closed systems. Estuaries in southern Africa generally experience higher flow variability than those elsewhere, with floods being unpredictable and relatively large compared to base flows (Gillson 2011) (Figure 7.13).

Harrison & Whitfield (2006) found that water temperature, along with salinity, was a major determinant of the distribution and abundance of fish species in southern African estuaries. Warm-temperate estuaries are dominated by warm-water endemic taxa. Widespread species, cool-water endemics, temperate species and eurythermal tropical species also extend their distribution into these estuaries. The abundance of tropical Indo-Pacific species decreases substantially southwest of Algoa Bay, where the warm Agulhas Current diverges from the coast (Maree et al. 2000) and summer upwelling can cause inshore waters to be at least 5°C cooler than the surrounding shelf water (Lutjeharms et al. 2001). Far fewer fish species occur in cool-temperate compared to subtropical estuaries, with cool-water endemic and widespread species dominating the fish assemblage in the former region (Harrison & Whitfield 2006). Day (1974) suggested that summer temperatures at the mouths of cool-temperate estuaries prevent the entry of marine fish larvae into these systems for all but the hardiest of species. The surface water temperatures of the west coast average between 13 and 15°C, even during summer as a result of pronounced summer upwelling (Harrison 2002).

Fishes in southern African estuaries will be influenced by changing marine conditions associated with climate change as well as altered estuarine conditions and an increase in the frequency of extreme events (Whitfield et al. 2016). The main climate change-related variables likely to impact estuarine fish assemblages in southern Africa were summarised in James et al. (2013) (Figure 7.14). Changes in the distribution patterns of estuarine-associated fish species associated with warming coastal and estuarine temperatures have been recorded in southern Africa's temperate region (James et al. 2013). The occurrence and/or abundance of tropical species has increased in some warm-temperate estuaries, without a corresponding decrease in abundance (or numbers) of temperate species, resulting in an increase in species richness over time (James et al. 2008, 2013, Whitfield et al. 2016). A recent paper (Duncan et al. 2019), however, showed that upwelling intensity has increased along southern Africa's south coast. An increase in the frequency and intensity of upwelling associated with climate change may ultimately limit the southward penetration of tropical species into temperate areas (James et al. 2013, Whitfield et al. 2016).

Climate change is predicted to cause an increase in the frequency and intensity of extreme events such as droughts, sea storms and river floods along the southern African coastline (Engelbrecht et al. 2013, James et al. 2013). An increase in floods will increase sediment delivery to estuaries and



**Figure 7.13** Flow variability at the mouth of the East and West Kleinemonde estuaries, open in (a) 2001, (b) 2003 and (c) during a sea storm surge in September 2008. Also shown, (d) upper Kowie Estuary catchment during flood conditions and (e) floodwater leaving the Kowie Estuary and entering the sea (Photos: courtesy of Paul Cowley and Alan Whitfield).

associated intertidal habitats, e.g. mangroves and salt marshes, and also increase within-estuary sediment instability affecting subtidal habitats (James et al. 2020). Studies in temperate South African estuaries show that the effects of river flooding on the fish fauna are dependent on the type of estuary, species composition and the intensity and duration of the flood event (James et al. 2017, Nodo et al. 2017, 2018, James et al. 2020).

In permanently open estuaries with narrow intertidal areas and limited habitat variability, flood events resulted in declines in fish abundance and richness and changes in species composition (Nodo



**Figure 7.14** Major climate change-related variables in (a) freshwater, (b) terrestrial and (c) marine environments that are likely to directly or indirectly influence estuary-associated fish assemblages in the future, through changes in estuarine water temperature, dissolved oxygen, turbidity, nutrient levels and general estuarine dynamics, e.g. changes in salinity and depth, and resulting in eutrophication and habitat loss (James et al. 2013).

et al. 2017, James et al. in press). In contrast, in a large permanently open estuary with substantial intertidal areas (and habitat availability), flooding had limited impact on the fish fauna (James et al. 2017). Recovery of the fish assemblage to pre-flood conditions (in terms of species composition and abundance) was fairly rapid in all systems studied and it was only when consecutive flood events occurred that recovery was less rapid. The response of the fish fauna in estuaries to floods (in terms of resilience) is thus likely to be affected by the frequency and magnitude of consecutive events (James et al. 2017).

In the permanently open Kariega Estuary, freshwater abstraction and prolonged droughts has led to the estuary becoming increasingly hypersaline in the upper reaches and a decrease in the abundance of estuaryassociated marine species (e.g. Whitfield 2004). In this system a major episodic flood event, the largest recorded in the estuary, resulted in the establishment of typical estuarine conditions (estuarine salinity gradient, high nutrients, etc.) for at least a year post-flood and was beneficial to the fish fauna, with an increase in the abundance of juvenile estuarine-dependent marine species (Nodo et al. 2018).

The fish fauna inhabiting intermittently open estuaries may be less resilient to the effects of droughts and floods than those inhabiting permanently open estuaries. In the intermittently open East Kleinemonde Estuary, situated in the warm-temperate region, analysis of a 20-year dataset indicated that the abundance of most marine species declined in recent years. This was attributed mainly to greater climatic variability (an increase in the intensity and duration of floods and droughts), which affected the availability of subtidal habitat for fishes (James et al. 2018).

#### 7.5.8 Temperate Australia

Australia is a large, mostly flat continent, spanning from tropical to temperate climatic biogeographic areas. It is the driest inhabited land mass on Earth, with rainfall sporadic, highly variable and irregular over much of the continent and among years. Climate change is evident in Australia, affecting freshwater and estuarine fish (Koehn et al. 2011, Morrongiello et al. 2011, Hallett et al. 2017). Moreover, there is a sense of urgency in tackling this issue as further impacts are inevitable, with projections reporting sustained rises in temperature and a decline in rainfall, and with past and ongoing events such as prolonged drought or enhanced water extraction shedding a grim light on the future of many Australian estuarine systems under climate change (Gillanders et al. 2011a, Hobday & Lough 2011, Hallett et al. 2017).

In the northern and subtropical areas, the timing and magnitude of rainfall and consequent variations to river flow between the wet and dry seasons are the most likely drivers of change to estuarine conditions. However, in southern temperate Australia and its Mediterranean-type climate regions, the impacts of reducing precipitation and dwindling river flow will be strongly associated with, and exacerbated by, increased temperature and evaporation processes. These Mediterranean-type weather regions (i.e. southern and south-western Australia) are markedly seasonal, with cooler, wetter winters but very hot, dry summers, and estuaries here are likely to be some of the most affected by climate change in Australia (Klausmeyer & Shaw 2009, Pittock & Finlayson 2011, Hallett et al. 2017). Warming and higher salinities are expected in southern Australian estuaries, associated with progressive 'marinisation' of estuarine conditions and fish communities (Valesini et al. 2013). However, because many estuarine systems in this region are also small sized, shallow, and have only temporarily open connections with the ocean, it is likely that many will experience lengthier closure periods, and an increase in the frequency and severity of hypersalinity, water column stratification and hypoxia conditions, among others (Gillanders et al. 2011a, Hallett et al. 2017). Some of these stresses are now self-evident, with recent work bringing many of these issues to management attention, regarding both major as well smaller temporarily open estuarine systems (e.g. Gillanders et al. 2011a, Koehn et al. 2011, Hallett et al. 2017).

The Coorong and Murray mouth are the terminus of Australia's largest riverine system – the Murray-Darling basin. This is one of the largest catchments in the world (flowing over 3750 km from headwaters to sea), but dramatic declines in run-off and river flow over time, together with chronic and widespread anthropogenic impacts, including rising water extraction rates, associated with the construction of five barrages in the 1930s to prevent the ingression of saltwater into the lower lakes, have led to closures of the Murray estuary mouth and severe hypersalinity conditions in the Coorong (Figure 7.15a–d). Today, the mouth of the Murray is continually dredged to keep the connection to sea open. Across the basin, evaporation is four times higher than rainfall, and only a fraction of the rainfall recharges groundwater or streamflow as higher air temperature boosts evaporation and reduce soil moisture; thus inflow to the estuary has been reduced by over 70%, in comparison with 1890s levels (Perkins & Pitman 2008, Kingsford et al. 2011, Pittock & Finlayson 2011).

The Coorong and Murray Estuary mouth are sites of international importance under the Ramsar convention, and renown nursery areas for numerous fish species. However, during the 2000s a severe drought provided a stark example of the potential long-term effects of climate change on estuarine fish communities, with many of the impacts and ecosystem shifts enduring long after the extreme drought period ceased (Gillanders et al. 2011a, Wedderburn et al. 2014). Reductions in freshwater flow to the Coorong resulted in large expanses of highly hypersaline conditions (>4× seawater, salinity >130), which severely impacted the diversity, distribution and abundance of fishes, and concurrently decreased productivity, prey availability and food web complexity. In the most hypersaline areas, the community became dominated by brine



**Figure 7.15** Aerial images of the Coorong in (a) 1983, (b) 1988 and (c) 2019 and (d) of one of the tidal barrages that regulate the flow between the lower lakes and the Murray mouth and Coorong. Also shown, temporarily closed off to sea estuaries of the (e) Western River and (f) Middle River both on Kangaroo Island, South Australia (Photos: courtesy Murray Darling Basin Authority, Geoff Gallasch, Luke Mosley, and Patrick Reis-Santos).

shrimp and a single species of hardy, salt-tolerant fish (*Atherinosoma microstoma*), with these circumstances still remaining today (Zampatti et al. 2010, Gillanders et al. 2011a, Brookes et al. 2015). Concomitantly, the reductions in flow impacted the lower salinity and freshwater areas of the estuary, resulting in a substantial decline in many freshwater and diadromous fishes, with multiple species impacted by changes in habitat availability, structure and loss of connectivity between critical habitats (Zampatti et al. 2010, Wedderburn et al. 2014). Ultimately, increased salinity and reduced flow promoted by drought resulted in the Coorong and Murray Estuary becoming less productive, with reduced nutrient concentrations, and lower chlorophyll and primary productivity, with evident water quality problems and decreases in biodiversity and fish abundance (Brookes et al. 2015). Towards the future, reduced freshwater flow will continue to reshape the environmental stressors fishes face within the system, and will continue to do so at increasing rates if the factors driving climate change continue unabated. An additional major issue in the Murray, as in other systems worldwide, is reductions in river flow cannot be seen in isolation from other exacerbating anthropogenic stressors such as water extraction (Koehn et al. 2011, Pittock & Finlayson 2011, Brookes et al. 2015).

In contrast to the large Murray-Darling basin, small, shallow, microtidal estuaries are ubiquitous throughout Southern Australia, with many of these systems temporarily closed off from the sea (Figure 7.15e, f). Hallett et al. (2017) made a comprehensive review of the observed and predicted effects of climate change in estuaries in southwestern Australia. These observations apply throughout most of the southern Australian coast but also reflect the future of estuaries in Mediterranean-type weather regions around the globe. Rising salinities will contribute to marine stragglers progressing farther into these estuaries and dominating the estuarine communities (Valesini et al. 2013, Potter et al. 2016, Hallett et al. 2017) whilst the maintenance of freshwater affinity species is jeopardized (Beatty et al. 2011, Wedderburn et al. 2014). Increasing temperatures pose a major physiological threat to fishes within these systems, and impacts will be particularly marked if estuaries are disconnected from the sea for long periods of time (Koehn et al. 2011, Killen et al. 2013). Decreasing precipitation and inflow associated with increasing evaporation will prolong the closure of periodically open estuaries, and potentially lead to the permanent closure of some systems (e.g. Chuwen et al. 2009). Overall, the responses of fish communities in temporarily open estuaries of southern Australia will be tightly linked to the duration and timing of aperture to the ocean, and such changes will ultimately impact species occurrence, both by limiting survival conditions within the estuary in periods of closure, or by stymieing the entry of new individuals via migratory movements, estuarine spawning or larval recruitment from coastal areas (Gillanders et al. 2011a, Hallett et al. 2017). Extended closure periods increase the susceptibility of extreme water temperatures, hypersalinity and/or hypoxia, which will contribute to increased physiological stress, impaired fish condition, and lead to reductions in the richness of fish communities, drastic simplifications of food webs, mass mortality events and local extirpation of fishes (Hoeksema et al. 2006, Chuwen et al. 2009, Cottingham et al. 2018, Wong et al. 2018, Tweedley et al. 2019). Overall, a wide suite of interacting environmental drivers and stressors can profoundly impact the estuarine ecology of estuaries in southern temperate regions of Australia and may potentially compromise the functions and ecosystem services they support.

# 7.6 Gaps in knowledge and future research directions

The ever-increasing prospect of severe climate change impacts on coastal and estuarine systems presents a clear call for global action. Given the pace of climate change, its influence on geomorphologic and bio-physical processes in estuaries, the risks it poses to the biodiversity of estuarine fish communities, as well

as to the ecosystem functions and services they provide, it is pivotal that we develop research strategies that support the resilience of estuarine environments and respond to the conservation, adaptation, management and governance challenges ahead (Sheaves et al. 2016, Williams et al. 2020). Yet, many efforts are still crippled by the lack of baseline knowledge on a myriad of issues ranging from species biology and life history characteristics, to effectively quantifying effects of climate stressor on estuarine fishes at individual or regional scales. In this section, we highlight some critical gaps in knowledge and priorities for future research directions regarding climate change and its effects on estuarine fishes.

As seen throughout the case studies, the risks of climate change to estuarine fish are not distributed equally, i.e. they vary among estuaries and associated habitats, processes therein, as well as across biogeographic regions. Yet, one common aspect is the lack of baseline understanding of the biology of many estuarine species, including their environmental sensitivities and tolerances (Gillanders et al. 2011a, Hallett et al. 2017). This holds true for both estuarine residents or species that use estuaries as nurseries. And because life history characteristics and vital rates (e.g. recruitment, growth, survival, see Houde et al. 2022) will govern the fish population's ability to respond or adapt to climate change-related impacts, this lack of baseline knowledge limits our ability to predict ecological responses, hindering the development of tailored adaptation and mitigation measures for estuarine fishes or habitats. Moreover, fishes in estuaries will be under a threatening combination of interactive stressors (e.g. increased temperature, decreased dissolved oxygen, acidification, sea level rise) that can act synergistically, additively or antagonistically, rendering evaluations of fish responses to individual stressors of climate change insufficient. Hence, integrative and holistic frameworks are warranted, and should include ecological responses, physiological impacts, genomics, metabolomics and ecological modelling studies, aiming at ascertaining how individual-level effects may impact estuarine fish populations.

Future research should also focus on how anthropogenic impacts in estuaries (e.g. contamination, habitat destruction, water extraction and flow restrictions for human use) will interact and compound climaterelated impacts (Koehn et al. 2011, Mach et al. 2015, Hallett et al. 2017, Cabral et al. 2019). The cumulative effects of such impacts will likely increase the vulnerability of fish species, communities and estuarine ecosystems to climate change. Harnessing the power of modelling approaches to anticipate the impacts of global and regional climate change on estuarine environments at ecologically relevant scales will be vital. Ultimately, filling these gaps will provide key reference information to any acclimation or adaptation predictions, and underpin the provision of successful management and mitigation strategies, including restoration or ecological engineering approaches that safeguard or restore natural processes and habitats.

Many estuarine fishes are characterised by wide environmental tolerances, and will have some capacity to tolerate or adapt to changing conditions (Elliott et al. 2007, Gillanders et al. 2011a). However, the range, magnitude and speed of changing conditions will play a key role, and will be most detrimental if significantly faster than the adaptative capacities of the estuarine species (Morrongiello et al. 2011). One key area of research, beyond short-term experiments characterising responses of fishes in estuaries to projected effects of climate change, is to understand how transgenerational acclimation may drive adaptive responses, and if these processes can ameliorate the climate-related impacts of warming, acidification and hypoxia on fishes over time (Munday 2014). Yet, irrespective of their ability to endure changing conditions, species- and assemblage-level responses will also be intricately related to habitat availability and trophic interactions. If key habitats are affected or physically squeezed out, it is likely species survival will be undermined. Therefore, it is essential when addressing the potential resilience and adaptation of fishes in estuaries under climate change to consider the implications on the ecosystem.

Moving towards safeguarding the resilience of estuarine systems and fish communities to climate change requires an a priori understanding of what is a healthy ecosystem (Tett et al. 2013, Sheaves

et al. 2016, Cabral et al. 2022). Therefore, knowledge on the abundance and composition of fish assemblages will contribute primarily to understanding key ecological processes and their interactions, including trophic and functional biodiversity, competitive relationships and trophic web dynamics. Such knowledge will underpin our ability to monitor, evaluate and manage estuarine ecosystems under change. These factors preclude an understanding of changes to interdependent and among species relations that may result from changes in habitat conditions, habitat availability, system productivity and consequent cascading effects on food web dynamics and trophic interactions. Ultimately, in the face of the continual and substantial threat of climate change, and the suite of complex impacts that affect estuaries both directly, or indirectly via changes to adjacent freshwater and marine environments, continued research on ecosystem function is essential to future-proofing the ecological value of estuarine systems (Sheaves et al. 2016, Hallett et al. 2017, Henriques et al. 2017b, Williams et al. 2020).

Given that species interactions underpin the structure of many estuarine and coastal communities, some of the most far-reaching impacts of climate change are likely to be driven by the decoupling of phenological relationships (Edwards & Richardson 2004, Pankhurst & Munday 2011, Staudinger et al. 2019). Changes in the timing of biotic interactions, in particular mismatches in life cycle events that disrupt trophic relationships (e.g. predator–prey interactions), will have significant implications at species, assemblage and community levels in estuaries, and can ultimately impact estuarine nursery roles and other ecosystem function or services (Chevillot et al. 2017). Therefore, we should strive to develop an integrated understanding of how climate change is affecting phenology, ontogeny and the timing of species trophic interactions in estuaries, which cannot be accomplished by investigating any given species in isolation (Yang & Rudolf 2010). Pivotal stepping stones include evaluating synchronicity between offshore spawning and the oceanographic conditions that promote estuarine settlement, or the temporal synchronicity between juvenile fish and zooplankton in estuaries, as well as other processes that may impact fish population connectivity and estuarine nursery role.

Regarding connectivity, it is crucial we enhance our understanding of how climate change may impact the recruitment of larvae and early life stages into estuaries, species persistence and movement between segregated life stages (e.g. juveniles in estuaries, and adults offshore). For early life stages (i.e. larvae and juveniles) of marine migrant species, changes to geomorphological and biophysical conditions within and outside the estuary will influence physiological impacts of climate change. Coupling biophysical models with larval movement will assist in determining potential changes to estuarine recruitment. Yet it is critical that we strive to include biological parameters (e.g. hatching success, pelagic larval duration, larval behaviour, survival and their variations with changing abiotic conditions) and information on available habitats for recruitment. Successful larval recruitment to estuarine nurseries is a multi-causal process, with a strong oceanographic component, modulated by abiotic conditions, and individual life history characteristics and fitness playing major roles. This combined area of research is yet underexplored and will significantly enhance the power of models, contribute to identifying the key areas for larval recruitment and boost our predictions of how changing oceanographic conditions may lead to changing recruitment patterns, or even community changes and loss of key species (Lett et al. 2010, Cetina-Heredia et al. 2015, Lacroix et al. 2018, Barbut et al. 2019). Connectivity and ecotone conservation, ensuring the link between key habitats along the estuarine-seascape continuum will be essential to safeguard the recruitment and persistence of marine migrant species, and indispensable to functioning and resilient estuarine systems.

Ultimately, a fundamental knowledge gap is long-term datasets. Long-term data allow us to decipher the past and establish baseline conditions to anticipate and mitigate future responses to environmental changes. Yet there is a general lack of biological data series in estuaries globally, while those that exist are frequently limited in scope. Developing monitoring efforts, and building long-term data series will be key assets to evaluate change, and it is never too late to start collecting data. In some cases, long-term datasets might be attainable by sharing data from different researchers, or collating data within an organisation. In the absence of long-term data, sclerochronological approaches can provide a robust, reliable and powerful look into the past and allow for confident predictions on the possible consequences of climate change (Doubleday et al. 2015, Black et al. 2016, Izzo et al. 2016). Another approach that can be taken is to use variations in oscillatory climate signals to predict future changes or impacts on estuarine biological communities. These signals include changes in climate or weather patterns that vary across a variety of temporal scales (from inter-decadal, to daily or weekly), e.g. variations in the El Niño Southern Oscillations, extreme heatwaves, droughts or floods (e.g. Garcia et al. 2003, Martinho et al. 2007, Hallett et al. 2017). A limitation of this approach is the assumption that biological responses to shorter signals in climate will match those over longer time scales, so it is important that all evaluations, management and adaptation options are robust to this potential uncertainty (Koehn et al. 2011). Nonetheless, these approaches may provide unique opportunities to explore future consequences of climate change, and examine some of the most likely future scenarios of climate change in real-world settings, before these are fully established and consolidated in estuarine environments.

## 7.7 Acknowledgements

This work had the support of the Fundação para a Ciência e Tecnologia (FCT) through the Strategic Project Grant UIDB/04292/2020 to MARE, project LIFELINE (PTDC/BIA-ECO/29261/2017), the post-doctoral grant SFRH/BPD/95784/2013 to PRS, and the researcher contract DL57/2016/CP1479/CT0024 to SH. RPV was funded by IPMA via the EU Data Collection Framework (namely for Portuguese fisher-ies data – Programa Nacional de Amostragem Biológica). BDW was funded by the United States National Science Foundation (NSF-OCE-1924011).

## 7.8 References

- Able, K.W. 2021. From cedar cemeteries to marsh lakes: a case study of sea-level rise and habitat change in a northeastern US salt marsh. Estuaries and Coasts 44, 1649–1657.
- Able, K.W. & Fahay, M.P. 2010. Ecology of Estuarine Fishes: Temperate Waters of the Western North Atlantic. The Johns Hopkins University Press, Baltimore.
- Able, K.W., Simenstad, C.A., Strydom, N.A., et al. 2022. Chapter 4. Habitat use and connectivity. In: Fish and Fisheries in Estuaries: A Global Perspective (ed., Whitfield, A.K., Able, K.W., Blaber, S.J.M., et al.), pp. 188–254. Wiley Blackwell, Oxford, UK.
- Abrantes, K.G. & Sheaves, M. 2010. Importance of freshwater flow in terrestrial-aquatic energetic connectivity in intermittently connected estuaries of tropical Australia. Marine Biology 157, 2071–2086.
- Acha, E.M. & Macchi, G.J. 2000. Spawning of Brazilian menhaden, *Brevoortia aurea*, in the Rio de la Plata estuary off Argentina and Uruguay. Fishery Bulletin 98, 227–235.
- Acha, E.M., Simionato, C.G., Carozza, C., et al. 2012. Climate-induced year-class fluctuations of whitemouth croaker *Micropogonias furnieri* (Pisces, Sciaenidae) in the Río de la Plata estuary, Argentina-Uruguay. Fisheries Oceanography 21, 58–77.
- Adams, J.B. 2014. A review of methods and frameworks used to determine the environmental water requirements of estuaries. Hydrological Sciences Journal 59, 451–465.

- Allen, L.G., Yoklavich, M.M., Cailliet, G.M., et al. 2006. Bays and estuaries. In: The Ecology of Marine Fishes: California and Adjacent Waters (eds., Allen, L.G., Pondella, D.J. & Horn, M.H.), pp. 119–148. University of California Press, Berkeley.
- Allison, E.H., Perry, A.L., Badjeck, M.-C., et al. 2009. Vulnerability of national economies to the impacts of climate change on fisheries. Fish and Fisheries 10, 173–196.
- Allmon, E.B. & Esbaugh, A.J. 2017. Carbon dioxide induced plasticity of branchial acid-base pathways in an estuarine teleost. Scientific Reports 7, 45680.
- Altenritter, M.E. & Walther, B.D. 2019. The legacy of hypoxia: tracking carryover effects of low oxygen exposure in a demersal fish using geochemical tracers. Transactions of the American Fisheries Society 148, 569–583.
- Anderson, M.J., Ellingsen, K.E. & McArdle, B.H. 2006. Multivariate dispersion as a measure of beta diversity. Ecology Letters 9, 683–693.
- Araújo, A.L.F., Dantas, R.P. & Pessanha, A.L.M. 2016. Feeding ecology of three juvenile mojarras (Gerreidae) in a tropical estuary of northeastern Brazil. Neotropical Ichthyology 14, e150039.
- Asbridge, E., Lucas, R., Ticehurst, C., et al. 2016. Mangrove response to environmental change in Australia's Gulf of Carpentaria. Ecology and Evolution 6, 3523–3539.
- Attrill, M.J. & Rundle, S.D. 2002. Ecotone or ecocline: ecological boundaries in estuaries. Estuarine, Coastal and Shelf Science 55, 929–936.
- Badjeck, M.-C., Allison, E.H., Halls, A.S., et al. 2010. Impacts of climate variability and change on fishery-based livelihoods. Marine Policy 34, 375–383.
- Baptista, J., Martinho, F., Dolbeth, M., et al. 2010. Effects of freshwater flow on the fish assemblage of the Mondego estuary (Portugal): comparison between drought and non-drought years. Marine and Freshwater Research 61, 490–501.
- Baptista, J., Martinho, F., Nyitrai, D., et al. 2015. Long-term functional changes in an estuarine fish assemblage. Marine Pollution Bulletin 97, 125–134.
- Baran, E. & Cain, J. 2001. Published ecological and modelling approach to flood-fish relationships in the Mekong River Basin. In: Proceedings of the National Workshop on Ecological and Environmental Modelling (ed. Koh, H.L. & Abu Hasan, Y.). Universiti Sains Malaysia, Penang, Malaysia.
- Barange, M., Merino, G., Blanchard, J.L., et al. 2014. Impacts of climate change on marine ecosystem production in societies dependent on fisheries. Nature Climate Change 4, 211–216.
- Barbut, L., Groot Crego, C., Delerue-Ricard, S., et al. 2019. How larval traits of six flatfish species impact connectivity. Limnology and Oceanography 64, 1150–1171.
- Barletta, M., Jaureguizar, A.J., Baigun, C., et al. 2010. Fish and aquatic habitat conservation in South America: a continental overview with emphasis on neotropical systems. Journal of Fish Biology 76, 2118–2176.
- Barros, V.R., Doyle, M.E. & Camilloni, I.A. 2008. Precipitation trends in southeastern South America: relationship with ENSO phases and with low-level circulation. Theoretical and Applied Climatology 93, 19–33.
- Barton, P.S., Cunningham, S.A., Manning, A.D., et al. 2013. The spatial scaling of beta diversity. Global Ecology and Biogeography 22, 639–647.
- Basset, A., Barbone, E., Elliott, M., et al. 2013. A unifying approach to understanding transitional waters: fundamental properties emerging from ecotone ecosystems. Estuarine, Coastal and Shelf Science 132, 5–16.
- Baumann, H., Talmage, S.C. & Gobler, C.J. 2011. Reduced early life growth and survival in a fish in direct response to increased carbon dioxide. Nature Climate Change 2, 38–41.
- Baumgartner, L.J. & Wibowo, A. 2018. Addressing fish-passage issues at hydropower and irrigation infrastructure projects in Indonesia. Marine and Freshwater Research 69, 1805–1813.
- Beatty, S.J., Morgan, D.L., Rashnavadi, M., et al. 2011. Salinity tolerances of endemic freshwater fishes of southwestern Australia: implications for conservation in a biodiversity hotspot. Marine and Freshwater Research 62, 91–100.

- Bell, G.W. & Eggleston, D.B. 2004. Species-specific avoidance responses by blue crabs and fish to chronic and episodic hypoxia. Marine Biology 146, 761–770.
- Bender, M.G., Pie, M.R., Rezende, E.L., et al. 2013. Biogeographic, historical and environmental influences on the taxonomic and functional structure of Atlantic reef fish assemblages. Global Ecology and Biogeography 22, 1173–1182.
- Bennett, B., Hamman, K., Branch, G., et al. 1985. Changes in the fish fauna of the Bot River estuary in relation to opening and closure of the estuary mouth. Transactions of the Royal Society of South Africa 45, 459–464.
- Bennett, W.A. 2010. Extreme physiology of intertidal fishes of the Wakatobi. In: Marine Research and Conservation in the Coral Triangle: The Wakatobi National Park (eds., Clifton, J., Unsworth, R.K.F. & Smith, D.J.), pp. 111– 128. Nova Science Publishers, New York.
- Bignami, S., Sponaugle, S. & Cowen, R.K. 2013. Response to ocean acidification in larvae of a large tropical marine fish, Rachycentron canadum. Global Change Biology 19, 996–1006.
- Blaber, S.J. 2008. Tropical Estuarine Fishes: Ecology, Exploitation and Conservation. Blackwell Science Ltd, Oxford.
- Blaber, S.J.M., Able, K.W. & Cowley, P.D. 2022. Chapter 9. Estuarine fisheries. In: Fish and Fisheries in Estuaries: A Global Perspective (ed., Whitfield, A.K., Able, K.W., Blaber, S.J.M., et al.), pp. 553–616. Wiley Blackwell, UK.
- Blaber, S.J.M. & Blaber, T.G. 1980. Factors affecting the distribution of juvenile estuarine and inshore fish. Journal of Fish Biology 17, 143–162.
- Black, B.A., Griffin, D., van der Sleen, P., et al. 2016. The value of crossdating to retain high-frequency variability, climate signals, and extreme events in environmental proxies. Global Change Biology 22, 2582–2595.
- Blanchard, J.L., Jennings, S., Holmes, R., et al. 2012. Potential consequences of climate change for primary production and fish production in large marine ecosystems. Philosophical Transactions of the Royal Society B: Biological Sciences 367, 2979–2989.
- Blondel, J. 2003. Guilds or functional groups: does it matter? Oikos 100, 223-231.
- Bodaly, R., Reist, J., Rosenberg, D., et al. 1989. Fish and fisheries of the Mackenzie and Churchill River basins, northern Canada. In: Proceedings of the International Large River Symposium (ed., Dodge, D.P.) Book 106, pp. 128– 144. Canadian Special Publication of Fisheries and Aquatic Sciences.
- Boeuf, G. & Payan, P. 2001. How should salinity influence fish growth? Comparative Biochemistry and Physiology Part C: Toxicology & Pharmacology 130, 411–423.
- Bond, W. & Erickson, R. 1985. Life history studies of anadromous coregonid fishes in two freshwater lake systems on the Tuktoyaktuk Peninsula, Northwest Territories. Canadian Technical Report of Fisheries and Aquatic Sciences No. 1336, 71 pp.
- Borchert, S.M., Osland, M.J., Enwright, N.M., et al. 2018. Coastal wetland adaptation to sea level rise: quantifying potential for landward migration and coastal squeeze. Journal of Applied Ecology 55, 2876–2887.
- Borde, A.B., Thom, R.M., Rumrill, S., et al. 2003. Geospatial habitat change analysis in Pacific Northwest coastal estuaries. Estuaries 26, 1104–1116.
- Boucek, R.E., Gaiser, E.E., Liu, H., et al. 2016. A review of subtropical community resistance and resilience to extreme cold spells. Ecosphere 7, e01455.
- Boucek, R.E. & Rehage, J.S. 2014. Climate extremes drive changes in functional community structure. Global Change Biology 20, 1821–1831.
- Boys, C.A. & Williams, R.J. 2012. Succession of fish and crustacean assemblages following reinstatement of tidal flow in a temperate coastal wetland. Ecological Engineering 49, 221–232.
- Brander, L.M., Van Beukering, P., Cesar, H.S.J. 2007. The recreational value of coral reefs: a meta-analysis. Ecological Economics 63, 209–218.

- Braverman, M.S., Acha, E.M., Gagliardini, D.A., et al. 2009. Distribution of whitemouth croaker (*Micropogonias furnieri*, Desmarest 1823) larvae in the Río de la Plata estuarine front. Estuarine, Coastal and Shelf Science 82, 557–565.
- Breitburg, D. 2002. Effects of hypoxia, and the balance between hypoxia and enrichment, on coastal fishes and fisheries. Estuaries 25, 767–781.
- Breitburg, D., Levin, L.A., Oschlies, A., et al. 2018. Declining oxygen in the global ocean and coastal waters. Science 359, eeam7240.
- Breitburg, D.L., Craig, J.K., Fulford, R.S., et al. 2009a. Nutrient enrichment and fisheries exploitation: interactive effects on estuarine living resources and their management. Hydrobiologia 629, 31–47.
- Breitburg, D.L., Hondorp, D.W., Davias, L.A., et al. 2009b. Hypoxia, nitrogen, and fisheries: integrating effects across local and global landscapes. Annual Review of Marine Science 1, 329–349.
- Breitburg, D.L., Loher, T., Pacey, C.A., et al. 1997. Varying effects of low dissolved oxygen on trophic interactions in an estuarine food web. Ecological Monographs 67, 489–507.
- Bricker, S.B., Clement, C.G., Pirhalla, D.E., et al. 1999. National estuarine eutrophication assessment: effects of nutrient enrichment in the nation's estuaries. NOAA, National Ocean Service, Special Projects Office and the National Centers for Coastal Ocean Science, Silver Spring, MD.
- Bricker, S.B., Longstaff, B., Dennison, W., et al. 2008. Effects of nutrient enrichment in the nation's estuaries: a decade of change. Harmful Algae 8, 21–32.
- Briggs, J.C. & Bowen, B.W. 2012. A realignment of marine biogeographic provinces with particular reference to fish distributions. Journal of Biogeography 39, 12–30.
- Broad, K., Pfaff, A. & Glantz, M. 1999. Climate information and conflicting goals: El Nino 1997-98 and the Peruvian fishery. Public Philosophy, Environment and Social Justice 21, 1999.
- Brookes, J.D., Aldridge, K.T., Bice, C.M., et al. 2015. Fish productivity in the lower lakes and Coorong, Australia, during severe drought. Transactions of the Royal Society of South Australia 139, 189–215.
- Brophy, L.S., Greene, C.M., Hare, V.C., et al. 2019. Insights into estuary habitat loss in the western United States using a new method for mapping maximum extent of tidal wetlands. PLoS One 14, e0218558.
- Brown, L.R., Bennett, W.A., Wagner, R.W., et al. 2013. Implications for future survival of delta smelt from four climate change scenarios for the Sacramento–San Joaquin Delta, California. Estuaries and Coasts 36, 754–774.
- Brown, L.R., Komoroske, L.M., Wagner, R.W., et al. 2016. Coupled downscaled climate models and ecophysiological metrics forecast habitat compression for an endangered estuarine fish. PLoS One 11, e0146724.
- Brugnoli, E., Verocai, J., Muniz, P., et al. 2018. Weather, hydrological and oceanographic conditions of the Northern Coast of the Río de la Plata estuary during ENSO 2009–2010. In: Estuary (ed., Froneman, W.), pp. 19–38. IntechOpen, London.
- Cabral, H., Fonseca, V., Sousa, T., et al. 2019. Synergistic effects of climate change and marine pollution: an overlooked interaction in coastal and estuarine areas. International Journal of Environmental Research and Public Health 16, 2737.
- Cabral, H.N., Borja, A., Fonseca, V.F., et al. 2022. Chapter 6. Fishes and environmental health. In: Fish and Fisheries in Estuaries: A Global Perspective (ed., Whitfield, A.K., Able, K.W., Blaber, S.J.M., et al.), pp. 332–379. Wiley Blackwell, Oxford, UK.
- Cabral, H.N., Costa, M.J. & Salgado, J.P. 2001. Does the Tagus estuary fish community reflect environmental changes? Climate Research 18, 119–126.
- Cacot, P. 2004. Activity report: domestication of the indigenous Mekong catfish Pangasius krempfi: overview of the fishery in Cambodia and Laos and preliminary study of artificial reproduction above the Khone waterfalls. CIRAD and MRC, Laos.
- Caffera, M. & Berbery, E.H. 2006. Climatologia de la Cuenca del Plata. In: El Cambio climático en la Cuenca del Plata (eds., Barros, V., Clarke, R. & Silva Dias, P.), pp. 19–47. Conicet, Buenos Aires.

- Camiolo, M.D., Cozzolino, E., Carozza, C.R., et al. 2019. Influencia del ambiente en la composición interanual de los ensambles de clases de talla de *Micropogonias furnieri*, especie de interés comercial. Revista Española de Estudios Agrosociales y Pesqueros 252, 59–89.
- Carey, G., Kraft, P., Cramp, R., et al. 2009. Effect of incubation temperature on muscle growth of barramundi *Lates* calcarifer at hatch and post-exogenous feeding. Journal of Fish Biology 74, 77–89.
- Carstensen, J., Andersen, J.H., Gustafsson, B.G., et al. 2014. Deoxygenation of the Baltic Sea during the last century. Proceedings of the National Academy of Sciences of the United States of America 111, 5628–5633.
- Cattano, C., Claudet, J., Domenici, P., et al. 2018. Living in a high CO<sub>2</sub> world: a global meta-analysis shows multiple trait-mediated fish responses to ocean acidification. Ecological Monographs 88, 320–335.
- Cech, J.J., Mitchell, S.J. & Wragg, T.E. 1984. Comparative growth of juvenile white sturgeon and striped bass: effects of temperature and hypoxia. Estuaries 7, 12–18.
- Cetina-Heredia, P., Roughan, M., van Sebille, E., et al. 2015. Strengthened currents override the effect of warming on lobster larval dispersal and survival. Global Change Biology 21, 4377–4386.
- Champalbert, G. & Koutsikopoulos, C. 2009. Behaviour, transport and recruitment of Bay of Biscay Sole (*Solea solea*): laboratory and field studies. Journal of the Marine Biological Association of the United Kingdom 75, 93–108.
- Chang, P. & Plumb, J. 1996. Effects of salinity on *Streptococcus* infection of Nile tilapia, *Oreochromis niloticus*. Journal of Applied Aquaculture 6, 39–45.
- Chapman, B.B., Skov, C., Hulthen, K., et al. 2012. Partial migration in fishes: definitions, methodologies and taxonomic distribution. Journal of Fish Biology 81, 479–499.
- Cheung, W.W., Lam, V.W., Sarmiento, J.L., et al. 2010. Large-scale redistribution of maximum fisheries catch potential in the global ocean under climate change. Global Change Biology 16, 24–35.
- Chevillot, X., Drouineau, H., Lambert, P., et al. 2017. Toward a phenological mismatch in estuarine pelagic food web? PLoS One 12, e0173752.
- Chícharo, M.A., Chícharo, L. & Morais, P. 2006. Inter-annual differences of ichthyofauna structure of the Guadiana estuary and adjacent coastal area (SE Portugal/SW Spain): before and after Alqueva dam construction. Estuarine, Coastal and Shelf Science 70, 39–51.
- Childs, A.R., Cowley, P.D., Næsje, T.F., et al. 2008. Do environmental factors influence the movement of estuarine fish? A case study using acoustic telemetry. Estuarine, Coastal and Shelf Science 78, 227–236.
- Chua, V.P. & Xu, M. 2014. Impacts of sea-level rise on estuarine circulation: an idealized estuary and San Francisco Bay. Journal of Marine Systems 139, 58–67.
- Chuwen, B.M., Hoeksema, S.D. & Potter, I.C. 2009. Factors influencing the characteristics of the fish faunas in offshore, deeper waters of permanently-open, seasonally-open and normally-closed estuaries. Estuarine, Coastal and Shelf Science 81, 279–295.
- CICPCP. 2017. Comité Intergubernamental Coordinador de los Países de la Cuenca del Plata; Strategic Action Program for the La Plata Basin. Estados Unidos : Organización de los Estados Americanos. Ciudad Autónoma de Buenos Aires.
- Clark, F.J.K. & Pessanha, A.L.M. 2015. Diet and ontogenetic shift in habitat use by *Rhinosardinia bahiensis* in a tropical semi-arid estuary, north-eastern Brazil. Journal of the Marine Biological Association of the United Kingdom 95, 175–183.
- Clark, R. & O'Connor, K. 2019. A systematic survey of bar-built estuaries along the California coast. Estuarine, Coastal and Shelf Science 226, 106285.
- Clark, T.D., Raby, G.D., Roche, D.G., et al. 2020. Ocean acidification does not impair the behaviour of coral reef fishes. Nature 577, 370–375.
- Cloern, J.E. 2001. Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology Progress Series 210, 223–253.

- Cloern, J.E., Abreu, P.C., Carstensen, J., et al. 2016. Human activities and climate variability drive fast-paced change across the world's estuarine-coastal ecosystems. Global Change Biology 22, 513–529.
- Cloern, J.E. & Dufford, R. 2005. Phytoplankton community ecology: principles applied in San Francisco Bay. Marine Ecology Progress Series 285, 11–28.
- Cloern, J.E., Hieb, K.A., Jacobson, T., et al. 2010. Biological communities in San Francisco Bay track large-scale climate forcing over the North Pacific. Geophysical Research Letters 37, L21602.
- Cloern, J.E. & Jassby, A.D. 2012. Drivers of change in estuarine-coastal ecosystems: discoveries from four decades of study in San Francisco Bay. Reviews of Geophysics 50, RG4001.
- Cloern, J.E., Knowles, N., Brown, L.R., et al. 2011. Projected evolution of California's San Francisco Bay-Delta-river system in a century of climate change. PLoS One 6, e24465.
- Cohen, A.N. & Carlton, J.T. 1995. Nonindigenous Aquatic Species in a United States Estuary: A Case Study of the Biological Invasions of the San Francisco Bay and Delta. U. S. Fish and Wildlife Service, Washington, DC.
- Collins, M., Knutti, R., Arblaster, J., et al. 2013. Long-term climate change: projections, commitments and irreversibility. In: Climate Change 2013-The Physical Science Basis: Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change (eds., Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M.M.B., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V. & Midgley, P.M.), pp. 1029–1136. Cambridge University Press, New York, USA.
- Conallin, J.C., Baumgartner, L.J., Lunn, Z., et al. 2019. Migratory fishes in Myanmar rivers and wetlands: challenges for sustainable development between irrigation water control infrastructure and sustainable inland capture fisheries. Marine and Freshwater Research 70, 1241–1253.
- Conner, W.H., Day, J.W., Baumann, R.H., et al. 1989. Influence of hurricanes on coastal ecosystems along the northern Gulf of Mexico. Wetlands Ecology and Management 1, 45–56.
- Conomos, T.J. 1979. San Francisco Bay: The Urbanized Estuary, Investigations into the Natural History of San Francisco Bay and Delta with Reference to the Influence of Man. American Association for the Advancement of Science, San Francisco, CA.
- Cortés, F., Jaureguizar, A.J., Guerrero, A.R., et al. 2011. Influence of estuarine and continental shelf water advection on the coastal movements of apron ray *Discopyge tschudii* in the Southwestern Atlantic. Journal of Applied Ichthyology 27, 1278–1285.
- Cortés, F., Jaureguizar, A.J., Menni, R.C., et al. 2010. Ontogenetic habitat preferences of the narrownose smoothhound shark, *Mustelus schmitti*, in two Southwestern Atlantic coastal areas. Hydrobiologia 661, 445–456.
- Costa, D.D., da Silva Pereira, T.A., Fragoso, C.R., et al. 2016. Understanding drought dynamics during dry season in eastern northeast Brazil. Frontiers in Earth Science 4, 69.
- Costa, M. & Elliott, M. 1991. Fish usage and feeding in two industrialised estuaries: the Tagus, Portugal and the Forth, Scotland. In: Estuaries and Coasts: Spatial and Temporal Intercomparisons (eds., Elliott, M. & Ducrotoy, J.-P.), pp. 269–297. Olsen and Olsen, Fredensborg, Denmark.
- Costa, M.J. 1990. Expected effects of temperature changes on estuarine fish populations. In: Expected Effects of Climatic Change on Marine Coastal Ecosystems (eds., Beukema, J.J., Wolff, W.J. & Brouns, J.J.W.M.), pp. 99–103. Kluwer Academic Publishers, Dordrecht.
- Costa, M.J. & Bruxelas, A. 1989. The structure of fish communities in the Tagus Estuary, Portugal, and its role as a nursery for commercial fish species. Scientia Marina 53, 561–566.
- Costa, M.J., Cabral, H.N., Drake, P., et al. 2002. Recruitment and production of commercial species in estuaries. In: Fishes in Estuaries (eds., Elliott, M. & Hemingway, K.), pp. 54–123. Blackwell Publishing Ltd, Oxford.
- Costa, M.J., Vasconcelos, R., Costa, J.L., et al. 2007. River flow influence on the fish community of the Tagus estuary (Portugal). Hydrobiologia 587, 113–123.

- Costanza, R., d'Arge, R., de Groot, R., et al. 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260.
- Cottingham, A., Huang, P., Hipsey, M.R., et al. 2018. Growth, condition, and maturity schedules of an estuarine fish species change in estuaries following increased hypoxia due to climate change. Ecology and Evolution 8, 7111–7130.
- Cowman, P.F. & Bellwood, D.R. 2013. Vicariance across major marine biogeographic barriers: temporal concordance and the relative intensity of hard versus soft barriers. Proceedings of the Royal Society B: Biological Sciences 280, 20131541.
- Craft, C., Clough, J., Ehman, J., et al. 2009. Forecasting the effects of accelerated sea-level rise on tidal marsh ecosystem services. Frontiers in Ecology and the Environment 7, 73–78.
- Craig, J.K. 2012. Aggregation on the edge: effects of hypoxia avoidance on the spatial distribution of brown shrimp and demersal fishes in the northern Gulf of Mexico. Marine Ecology Progress Series 445, 75–95.
- Craig, J.K. & Crowder, L.B. 2005. Hypoxia-induced habitat shifts and energetic consequences in Atlantic croaker and brown shrimp on the Gulf of Mexico shelf. Marine Ecology Progress Series 294, 79–94.
- Creighton, C., Boon, P.I., Brookes, J.D., et al. 2015. Repairing Australia's estuaries for improved fisheries productionwhat benefits, at what cost? Marine and Freshwater Research 66, 493–507.
- Cripps, I.L., Munday, P.L., McCormick, M.I. 2011. Ocean acidification affects prey detection by a predatory reef fish. PLoS One 6, e22736.
- Crosby, S.C., Sax, D.F., Palmer, M.E., et al. 2016. Salt marsh persistence is threatened by predicted sea-level rise. Estuarine, Coastal and Shelf Science 181, 93–99.
- Cyrus, D., Jerling, H., MacKay, F., et al. 2011. Lake St Lucia, Africa's largest estuarine lake in crisis: combined effects of mouth closure, low levels and hypersalinity. South African Journal of Science 107, 01–13.
- Cyrus, D. & Vivier, L. 2006. Status of the estuarine fish fauna in the St Lucia estuarine system, South Africa, after 30 months of mouth closure. African Journal of Aquatic Science 31, 71–81.
- da Silva Lima, C.S., de Araújo Souto Badú, M.L., Pessanha, A.L.M. 2020. Response of estuarine fish assemblages to a typical climatic event in northeastern Brazil. Regional Studies in Marine Science 35, 101121.
- da Silva, R.S., Baeta, A.S.B.V., Pessanha, A.L.M. 2018. Are vegetated areas more attractive for juvenile fish in estuaries? A comparison in a tropical estuary. Environmental Biology of Fishes 101, 1427–1442.
- Dahlke, F.T., Wohlrab, S., Butzin, M., et al. 2020. Thermal bottlenecks in the life cycle define climate vulnerability of fish. Science 369, 65–70.
- Darnaude, A.M., Salen-Picard, C., Polunin, N.V., et al. 2004. Trophodynamic linkage between river runoff and coastal fishery yield elucidated by stable isotope data in the Gulf of Lions (NW Mediterranean). Oecologia 138, 325–332.
- Davis, B.E., Hansen, M.J., Cocherell, D.E., et al. 2019. Consequences of temperature and temperature variability on swimming activity, group structure, and predation of endangered delta smelt. Freshwater Biology 64, 2156–2175.
- Day, J.H. 1974. The ecology of Morrumbene Estuary, MoÇambique. Transactions of the Royal Society of South Africa 41, 43–97.
- Day, J.W., Yáñez-Arancibia, A. & Rybczyk, J.M. 2011. Climate change: effects, causes, consequences. In: Treatise on Estuarine and Coastal Science (eds., Wolanski, E. & McLusky, D.), pp. 303–315. Academic Press, Waltham.
- De Silva, D. & Yamao, M. 2007. Effects of the tsunami on fisheries and coastal livelihood: a case study of tsunamiravaged southern Sri Lanka. Disasters 31, 386–404.
- De Wysiecki, A.M., Bruno, I., Carvalho-Filho, A., et al. 2018. First documented records of tropical morays (Muraenidae) in Argentina and Uruguay. Pan-American Journal of Aquatic Sciences 13, 254–259.
- De Wysiecki, A.M., Jaureguizar, A.J., Cortés, F. 2017. The importance of environmental drivers on the narrownose smoothhound shark (*Mustelus schmitti*) yield in a small-scale gillnet fishery along the southern boundary of the Río de la Plata estuarine area. Fisheries Research 186, 345–355.

- Deb, P., Thin, N.M.M., Shrestha, S. 2014. Assessment of climate change impacts on irrigation water requirement and rice yield for Ngamoeyeik Irrigation Project in Myanmar. Journal of Water and Climate Change 5, 427–442.
- DePasquale, E., Baumann, H., Gobler, C.J. 2015. Vulnerability of early life stage Northwest Atlantic forage fish to ocean acidification and low oxygen. Marine Ecology Progress Series 523, 145–156.
- Dettmann, E.H. 2001. Effect of water residence time on annual export and denitrification of nitrogen in estuaries: a model analysis. Estuaries 24, 481–490.
- Diaz, R.J. & Rosenberg, R. 2008. Spreading dead zones and consequences for marine ecosystems. Science 321, 926–929.
- Dijkstra, J.A., Buckman, K.L., Ward, D., et al. 2013. Experimental and natural warming elevates mercury concentrations in estuarine fish. PLoS One 8, e58401.
- Dixson, D.L., Munday, P.L. & Jones, G.P. 2010. Ocean acidification disrupts the innate ability of fish to detect predator olfactory cues. Ecology Letters 13, 68–75.
- Dolbeth, M., Martinho, F., Freitas, V., et al. 2010. Multi-year comparisons of fish recruitment, growth and production in two drought-affected Iberian estuaries. Marine and Freshwater Research 61, 1399–1415.
- Dolbeth, M., Vendel, A.L., Baeta, A., et al. 2016. Exploring ecosystem functioning in two Brazilian estuaries integrating fish diversity, species traits and food webs. Marine Ecology Progress Series 560, 41–55.
- Domenici, P., Allan, B., McCormick, M.I., et al. 2012. Elevated carbon dioxide affects behavioural lateralization in a coral reef fish. Biology Letters 8, 78–81.
- Doney, S.C., Ruckelshaus, M., Duffy, J.E., et al. 2012. Climate change impacts on marine ecosystems. Annual Review of Marine Science 4, 11–37.
- Doubleday, Z.A., Izzo, C., Haddy, J.A., et al. 2015. Long-term patterns in estuarine fish growth across two climatically divergent regions. Oecologia 179, 1079–1090.
- Drinkwater, K.F., Beaugrand, G., Kaeriyama, M., et al. 2010. On the processes linking climate to ecosystem changes. Journal of Marine Systems 79, 374–388.
- Drinkwater, K.F. & Frank, K.T. 1994. Effects of river regulation and diversion on marine fish and invertebrates. Aquatic Conservation: Marine and Freshwater Ecosystems 4, 135–151.
- Dubois, M.J., Akester, M., Leemans, K., et al. 2019. Integrating fish into irrigation infrastructure projects in Myanmar: rice-fish what if. . .? Marine and Freshwater Research 70, 1229–1240.
- Dubuc, A., Baker, R., Marchand, C., et al. 2019. Hypoxia in mangroves: occurrence and impact on valuable tropical fish habitat. Biogeosciences 16, 3959–3976.
- Ducrotoy, J.-P., Elliott, M., de Jonge, V.N. 2000. The North Sea. Marine Pollution Bulletin 41, 5–23.
- Dudgeon, D. 2005. River rehabilitation for conservation of fish biodiversity in monsoonal Asia. Ecology and Society 10, 15–35.
- Duffy-Anderson, J.T., Bailey, K.M., Cabral, H.N., et al. 2014. The planktonic stages of flatfishes: physical and biological interactions in transport processes. In: Flatfishes: Biology and Exploitation (eds., Gibson, R.N., Nash, R.D., Geffen, A.J., et al.), pp. 132–170. John Wiley & Sons, Ltd, Chichester, West Sussex.
- Dugan, P.J., Barlow, C., Agostinho, A.A., et al. 2010. Fish migration, dams, and loss of ecosystem services in the Mekong basin. Ambio 39, 344–348.
- Duncan, M.I., James, N.C., Bates, A.E., et al. 2019. Localised intermittent upwelling intensity has increased along South Africa's south coast due to El Niño–Southern Oscillation phase state. African Journal of Marine Science 41, 325–330.
- Eddy, F.B. 2005. Ammonia in estuaries and effects on fish. Journal of Fish Biology 67, 1495–1513.
- Edeline, E., Lacroix, G., Delire, C., et al. 2013. Ecological emergence of thermal clines in body size. Global Change Biology 19, 3062–3068.
- Edwards, M. & Richardson, A.J. 2004. Impact of climate change on marine pelagic phenology and trophic mismatch. Nature 430, 881–884.

- Elliott, M., Lonsdale, J.A., Houde, E.D., et al. 2022. Chapter 12. Management of fishes and fisheries in estuaries. In: Fish and Fisheries in Estuaries: A Global Perspective (ed., Whitfield, A.K., Able, K.W., Blaber, S.J.M., et al.), pp. 706–797. Wiley Blackwell, Oxford, UK.
- Elliott, M., O'Reilly, M.G., Taylor, C.J.L. 1990. The Forth estuary: a nursery and overwintering area for North Sea fishes. Hydrobiologia 195, 89–103.
- Elliott, M. & Quintino, V. 2007. The estuarine quality paradox, environmental homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. Marine Pollution Bulletin 54, 640–645.
- Elliott, M., Whitfield, A.K., Potter, I.C., et al. 2007. The guild approach to categorizing estuarine fish assemblages: a global review. Fish and Fisheries 8, 241–268.
- Eme, J., Dabruzzi, T.F., Bennett, W.A. 2011. Thermal responses of juvenile squaretail mullet (*Liza vaigiensis*) and juvenile crescent terapon (*Terapon jarbua*) acclimated at near-lethal temperatures, and the implications for climate change. Journal of Experimental Marine Biology and Ecology 399, 35–38.
- Emmett, R., Llansó, R., Newton, J., et al. 2000. Geographic signatures of North American west coast estuaries. Estuaries 23, 765–792.
- Engelbrecht, C.J., Engelbrecht, F.A. & Dyson, L.L. 2013. High-resolution model-projected changes in midtropospheric closed-lows and extreme rainfall events over southern Africa. International Journal of Climatology 33, 173–187.
- Erfanian, A., Wang, G. & Fomenko, L. 2017. Unprecedented drought over tropical South America in 2016: significantly under-predicted by tropical SST. Scientific Reports 7, 5811–5811.
- Erzini, K. 2005. Trends in NE Atlantic landings (southern Portugal): identifying the relative importance of fisheries and environmental variables. Fisheries Oceanography 14, 195–209.
- Essington, T.E. & Paulsen, C.E. 2010. Quantifying hypoxia impacts on an estuarine demersal community using a hierarchical ensemble approach. Ecosystems 13, 1035–1048.
- Estelles, P., Jensen, H., Sanchez, L., et al. 2002. Sustainable Development in the Mekong Delta. Center for Environmental Studies, University of Aarhus, Denmark.
- Fabry, V.J., Seibel, B.A., Feely, R.A., et al. 2008. Impacts of ocean acidification on marine fauna and ecosystem processes. ICES Journal of Marine Science 65, 414–432.
- FAO. 2018. The State of World Fisheries and Aquaculture 2018-Meeting the sustainable development goals. FAO Rome, Italy
- Fernández-Delgado, C., Baldó, F., Vilas, C., et al. 2007. Effects of the river discharge management on the nursery function of the Guadalquivir river estuary (SW Spain). Hydrobiologia 587, 125–136.
- Feyrer, F., Cloern, J.E., Brown, L.R., et al. 2015. Estuarine fish communities respond to climate variability over both river and ocean basins. Global Change Biology 21, 3608–3619.
- Feyrer, F., Newman, K., Nobriga, M., et al. 2010. Modeling the effects of future outflow on the abiotic habitat of an imperiled estuarine fish. Estuaries and Coasts 34, 120–128.
- Figueira, W.F. & Booth, D.J. 2010. Increasing ocean temperatures allow tropical fishes to survive overwinter in temperate waters. Global Change Biology 16, 506–516.
- Fodrie, F.J., Heck, K.L., Powers, S.P., et al. 2010. Climate-related, decadal-scale assemblage changes of seagrassassociated fishes in the northern Gulf of Mexico. Global Change Biology 16, 48–59.
- Ford, J. & Bedford, B.L. 1987. The hydrology of Alaskan wetlands, U.S.A.: a review. Arctic and Alpine Research 19, 209–229.
- Free, C.M., Thorson, J.T., Pinsky, M.L., et al. 2019. Impacts of historical warming on marine fisheries production. Science 363, 979–983.
- Friedland, K. 1993. Marine survival of North American and European Atlantic salmon: effects of growth and environment. ICES Journal of Marine Science 50, 481–492.

- Friedland, K.D., Reddin, D.G., McMenemy, J.R., et al. 2003. Multidecadal trends in North American Atlantic salmon (*Salmo salar*) stocks and climate trends relevant to juvenile survival. Canadian Journal of Fisheries and Aquatic Sciences 60, 563–583.
- Gahagan, B.I., Fox, D.A. & Secor, D.H. 2015. Partial migration of striped bass: revisiting the contingent hypothesis. Marine Ecology Progress Series 525, 185–197.
- Garcia, A.F.S., Pasquaud, S., Cabral, H., et al. 2019. Assimilation of allochthonous matter by estuarine consumers during the 2015 El Niño event. Estuaries and Coasts 42, 1281–1296.
- Garcia, A.M., Vieira, J. & Winemiller, K. 2001. Dynamics of the shallow-water fish assemblage of the Patos Lagoon estuary (Brazil) during cold and warm ENSO episodes. Journal of Fish Biology 59, 1218–1238.
- Garcia, A.M., Vieira, J. & Winemiller, K. 2003. Effects of 1997–1998 El Niño on the dynamics of the shallow-water fish assemblage of the Patos Lagoon Estuary (Brazil). Estuarine, Coastal and Shelf Science 57, 489–500.
- Garcia, A.M., Winemiller, K.O., Hoeinghaus, D.J., et al. 2017. Hydrologic pulsing promotes spatial connectivity and food web subsidies in a subtropical coastal ecosystem. Marine Ecology Progress Series 567, 17–28.
- Garcia-Seoane, E., Dolbeth, M., Silva, C.L., et al. 2016. Changes in the fish assemblages of a coastal lagoon subjected to gradual salinity increases. Marine Environmental Research 122, 178–187.
- Gaston, K.J. 2000. Global patterns in biodiversity. Nature 405, 220.
- Gehrke, P.C., Sheaves, M.J., Boseto, D.T., et al. 2011. Vulnerability of freshwater and estuarine fisheries in the tropical Pacific to climate change. In: Vulnerability of Tropical Pacific Fisheries and Aquaculture to Climate Change (eds., Bell, J.D., Johnson, J.E. & Hobday, A.J.), pp. 577–646.Secretariat of the Pacific Community, Noumea, New Caledonia.
- Gianelli, I., Ortega, L., Marín, Y., et al. 2019. Evidence of ocean warming in Uruguay's fisheries landings: the mean temperature of the catch approach. Marine Ecology Progress Series 625, 115–125.
- Gillanders, B.M., Elsdon, T.S., Halliday, I.A., et al. 2011a. Potential effects of climate change on Australian estuaries and fish utilising estuaries: a review. Marine and Freshwater Research 62, 1115–1131.
- Gillanders, B.M., Elsdon, T.S., Roughan, M. 2011b. Connectivity of estuaries. In: Treatise on Estuarine and Coastal Science (eds., Wolanski, E. & McLusky, D.), pp. 119–142. Academic Press, Waltham.
- Gillanders, B.M., Izzo, C., Doubleday, Z.A., et al. 2015. Partial migration: growth varies between resident and migratory fish. Biology Letters 11, 20140850.
- Gillson, J. 2011. Freshwater flow and fisheries production in estuarine and coastal systems: where a drop of rain is not lost. Reviews in Fisheries Science 19, 168–186.
- Gillson, J., Scandol, J. & Suthers, I. 2009. Estuarine gillnet fishery catch rates decline during drought in eastern Australia. Fisheries Research 99, 26–37.
- González-Ortegón, E., Subida, M.D., Cuesta, J.A., et al. 2010. The impact of extreme turbidity events on the nursery function of a temperate European estuary with regulated freshwater inflow. Estuarine, Coastal and Shelf Science 87, 311–324.
- Graham, C.H. & Fine, P.V. 2008. Phylogenetic beta diversity: linking ecological and evolutionary processes across space in time. Ecology Letters 11, 1265–1277.
- Grimes, C.B. 2001. Fishery production and the Mississippi River discharge. Fisheries 26, 17–26.
- Guerrero, R.A., Acha, E.M., Framin<sup>a</sup>n, M.B., et al. 1997. Physical oceanography of the Río de la Plata Estuary, Argentina. Continental Shelf Research 17, 727–742.
- Gyllström, M., Hansson, L.A., Jeppesen, E., et al. 2005. The role of climate in shaping zooplankton communities of shallow lakes. Limnology and Oceanography 50, 2008–2021.
- Hallett, C.S., Hobday, A.J., Tweedley, J.R., et al. 2017. Observed and predicted impacts of climate change on the estuaries of south-western Australia, a Mediterranean climate region. Regional Environmental Change 18, 1357–1373.
- Halls, A. 1998. Impact of flood control schemes on river fish migrations and species assemblages in Bangladesh. Journal of Fish Biology 53, 358–380.

- Harrison, T.D. 2002. Preliminary assessment of the biogeography of fishes in South African estuaries. Marine and Freshwater Research 53, 479–490.
- Harrison, T.D. & Whitfield, A.K. 2006. Temperature and salinity as primary determinants influencing the biogeography of fishes in South African estuaries. Estuarine Coastal and Shelf Science 66, 335–345.
- Hartmann, W. & Ingthamjitr, S. 2007. Learning from Myanmar's inland fisheries. Catch and Culture 13, 12-14.
- Harvell, C., Kim, K., Burkholder, J., et al. 1999. Emerging marine diseases climate links and anthropogenic factors. Science 285, 1505–1510.
- Harvell, C.D., Mitchell, C.E., Ward, J.R., et al. 2002. Climate warming and disease risks for terrestrial and marine biota. Science 296, 2158–2162.
- Hassell, K.L., Coutin, P.C. & Nugegoda, D. 2008. Hypoxia, low salinity and lowered temperature reduce embryo survival and hatch rates in black bream *Acanthopagrus butcheri* (Munro, 1949). Journal of Fish Biology 72, 1623–1636.
- Heady, W.N., O'Connor, K., Kassakian, J., et al. 2014. An Inventory and Classification of U.S. West Coast Estuaries. The Nature Conservancy, Arlington, VA.
- Healey, M., Goodwin, P., Dettinger, M., et al. 2016. The state of Bay-Delta science 2016: An introduction. San Francisco Estuary and Watershed Science 14(2), 1–7.
- Heath, M.R., Neat, F.C., Pinnegar, J.K., et al. 2012. Review of climate change impacts on marine fish and shellfish around the UK and Ireland. Aquatic Conservation: Marine and Freshwater Ecosystems 22, 337–367.
- Henderson, P.A. 2007. Discrete and continuous change in the fish community of the Bristol Channel in response to climate change. Journal of the Marine Biological Association of the United Kingdom 87, 589–598.
- Henriques, S., Cardoso, P., Cardoso, I., et al. 2017a. Processes underpinning fish species composition patterns in estuarine ecosystems worldwide. Journal of Biogeography 44, 627–639.
- Henriques, S., Guilhaumon, F., Villéger, S., et al. 2017b. Biogeographical region and environmental conditions drive functional traits of estuarine fish assemblages worldwide. Fish and Fisheries 18, 752–771.
- Heuer, R.M. & Grosell, M. 2014. Physiological impacts of elevated carbon dioxide and ocean acidification on fish. American Journal of Physiology – Regulatory, Integrative and Comparative Physiology 307, R1061–R1084.
- Hickey, B.M. & Banas, N.S. 2003. Oceanography of the U.S. Pacific northwest coastal ocean and estuaries with application to coastal ecology. Estuaries 26, 1010–1031.
- Hillebrand, H. & Blenckner, T. 2002. Regional and local impact on species diversity-from pattern to processes. Oecologia 132, 479–491.
- Hobday, A.J. & Lough, J.M. 2011. Projected climate change in Australian marine and freshwater environments. Marine and Freshwater Research 62, 1000–1014.
- Hodgkin, E.P. & Hesp, P. 1998. Estuaries to salt lakes: Holocene transformation of the estuarine ecosystems of southwestern Australia. Marine and Freshwater Research 49, 183–201.
- Hoeksema, S.D., Chuwen, B.M. & Potter, I.C. 2006. Massive mortalities of the black bream *Acanthopagrus butcheri* (Sparidae) in two normally-closed estuaries, following extreme increases in salinity. Journal of the Marine Biological Association of the United Kingdom 86, 893–897.
- Hoeksema, S.D., Chuwen, B.M. & Potter, I.C. 2009. Comparisons between the characteristics of ichthyofaunas in nearshore waters of five estuaries with varying degrees of connectivity with the ocean. Estuarine, Coastal and Shelf Science 85, 22–35.
- Hoffman, J.C., Kelly, J.R., Peterson, G.S., et al. 2014. Landscape-scale food webs of fish nursery habitat along a rivercoast mixing zone. Estuaries and Coasts 38, 1335–1349.
- Hogan, Z., Baird, I.G., Radtke, R., et al. 2007. Long distance migration and marine habitation in the tropical Asian catfish, *Pangasius krempfi*. Journal of Fish Biology 71, 818–832.
- Hogan, Z., Moyle, P., May, B., et al. 2004. The imperiled giants of the Mekong. American Scientist 92, 228–237.

- Honda, S., Muthmainnah, D., Suryati, N., et al. 2016. Current status and problems of the catch statistics on anguillid eel fishery in Indonesia. Marine Research in Indonesia 41, 1–13.
- Hortle, K.G. 2007. Consumption and yield of fish and other aquatic animals from the Lower Mekong Basin. MRC Technical Report No 16. Mekong River Commission, Vientiane.
- Hortle, K.G. & Suntornratana, U. 2008. Socio-economics of the fisheries of the lower Songkhram River Basin, northeast Thailand. MRC Technical Paper 17, 1–85.
- Horton, R., De Mel, M., Peters, D., et al. 2017. Assessing Climate Risk in Myanmar: Technical Report. Center for Climate Systems Research at Columbia University. WWF-US and WWF-Myanmar, New York, NY, USA.
- Hossain, M.S., Sharifuzzaman, S.M., Rouf, M.A., et al. 2019. Tropical hilsa shad (*Tenualosa ilisha*): biology, fishery and management. Fish and Fisheries 20, 44–65.
- Houde, E.D., Able, K.W., Strydom, N.A., et al. 2022. Chapter 3. Reproduction, ontogeny and recruitment. In: Fish and Fisheries in Estuaries: A Global Perspective (ed., Whitfield, A.K., Able, K.W., Blaber, S.J.M., et al.), pp. 60– 187. Wiley Blackwell, Oxford, UK.
- Houde, E.D. & Rutherford, E.S. 1993. Recent trends in estuarine fisheries: predictions of fish production and yield. Estuaries 16, 161–176.
- Howarth, R., Chan, F., Conley, D.J., et al. 2011. Coupled biogeochemical cycles: eutrophication and hypoxia in temperate estuaries and coastal marine ecosystems. Frontiers in Ecology and the Environment 9, 18–26.
- Hughes, B.B., Levey, M.D., Brown, J.A., et al. 2014. Nursery Functions of U.S. West Coast Estuaries: The State of Knowledge for Juveniles of Focal Invertebrate and Fish Species. The Nature Conservancy, Arlington, VA.
- Human, L.R.D., Snow, G.C., Adams, J.B. 2016. Responses in a temporarily open/closed estuary to natural and artificial mouth breaching. South African Journal of Botany 107, 39–48.
- Hurst, T.P. 2007. Causes and consequences of winter mortality in fishes. Journal of Fish Biology 71, 315–345.
- Huston, M.A. & Wolverton, S. 2011. Regulation of animal size by eNPP, Bergmann's rule, and related phenomena. Ecological Monographs 81, 349–405.
- IPPC. 2007. Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change IPCC, Geneva, Switzerland.
- Islam, M.M., Sallu, S., Hubacek, K., et al. 2013. Vulnerability of fishery-based livelihoods to the impacts of climate variability and change: insights from coastal Bangladesh. Regional Environmental Change 14, 281–294.
- Ives, A.R. & Carpenter, S.R. 2007. Stability and diversity of ecosystems. Science 317, 58-62.
- Izzo, C., Doubleday, Z.A., Grammer, G.L., et al. 2016. Multi-species response to rapid environmental change in a large estuary system: a biochronological approach. Ecological Indicators 69, 739–748.
- James, N.C., Adams, J.B., Connell, A.D., et al. 2020. High flow variability and storm events shape the ecology of the Mbhashe Estuary, South Africa. African Journal of Aquatic Science 45(1–2), 131–151.
- James, N.C., Cowley, P.D. & Whitfield, A.K. 2018. The marine fish assemblage of the East Kleinemonde Estuary over 20 years: declining abundance and nursery function? Estuarine, Coastal and Shelf Science 214, 64–71.
- James, N.C., Cowley, P.D., Whitfield, A.K., et al. 2007. Fish communities in temporarily open/closed estuaries from the warm- and cool-temperate regions of South Africa: a review. Reviews in Fish Biology and Fisheries 17, 565–580.
- James, N.C., Lamberth, S.J., Midgley, C., et al. 2017. Resilience of fish assemblages in the Breede Estuary, South Africa, to environmental perturbations. Environmental Biology of Fishes 101, 109–126.
- James, N.C., van Niekerk, L., Whitfield, A.K., et al. 2013. Effects of climate change on South African estuaries and associated fish species. Climate Research 57, 233–248.
- James, N.C., Whitfield, A.K. & Cowley, P.D. 2008. Long-term stability of the fish assemblages in a warm-temperate South African estuary. Estuarine, Coastal and Shelf Science 76, 723–738.
- James, N.C., Whitfield, A.K. & Harrison, T.D. 2016. Grey mullet (Mugilidae) as possible indicators of global warming in South African estuaries and coastal waters. Marine Environmental Research 122, 188–195.

- Janousek, C.N. & Mayo, C. 2013. Plant responses to increased inundation and salt exposure: interactive effects on tidal marsh productivity. Plant Ecology 214, 917–928.
- Jaureguizar, A.J., Bava, B., Carozza, C.R., et al. 2003. Distribution of whitemouth croaker *Micropogonias furnieri* in relation to environmental factors at the Río de la Plata estuary, South America. Marine Ecology Progress Series 255, 271–282.
- Jaureguizar, A.J., Cortés, F., Milessi, A.C., et al. 2015. A trans-ecosystem fishery: environmental effects on the smallscale gillnet fishery along the Río de la Plata boundary. Estuarine, Coastal and Shelf Science 166, 92–104.
- Jaureguizar, A.J., Menni, R., Guerrero, R., et al. 2004. Environmental factors structuring fish communities of the Río de la Plata estuary. Fisheries Research 66, 195–211.
- Jaureguizar, A.J., Menni, R., Lasta, C., et al. 2006. Fish assemblages of the northern Argentine coastal system: spatial patterns and their temporal variations. Fisheries Oceanography 15, 326–344.
- Jaureguizar, A.J., Militelli, M.I., Guerrero, R. 2008. Distribution of *Micropogonias furnieri* at different maturity stages along an estuarine gradient and in relation to environmental factors. Journal of the Marine Biological Association of the United Kingdom 88, 175–181.
- Jaureguizar, A.J., Solari, A., Cortes, F., et al. 2016. Fish diversity in the Rio de la Plata and adjacent waters: an overview of environmental influences on its spatial and temporal structure. Journal of Fish Biology 89, 569–600.
- Jaureguizar, A.J., Waessle, J.A. & Guerrero, R.A. 2007. Spatio-temporal distribution of Atlantic searobins (*Prionotus* spp.) in relation to estuarine dynamics (Río de la Plata, Southwestern Atlantic coastal system). Estuarine, Coastal and Shelf Science 73, 30–42.
- Jensen, H.S. & Andersen, F.O. 1992. Importance of temperature, nitrate, and pH for phosphate release from aerobic sediments of four shallow, eutrophic lakes. Limnology and Oceanography 37, 577–589.
- Jian, C.Y., Cheng, S.Y. & Chen, J.C. 2003. Temperature and salinity tolerances of yellowfin sea bream, Acanthopagrus latus, at different salinity and temperature levels. Aquaculture Research 34, 175–185.
- Johnson, G.M., Sutherland, D.A., Roering, J.J., et al. 2019. Estuarine dissolved oxygen history inferred from sedimentary trace metal and organic matter preservation. Estuaries and Coasts 42, 1211–1225.
- Johnson, P.T. & Carpenter, S.R. 2008. Influence of eutrophication on disease in aquatic ecosystems: patterns, processes, and predictions. In: Infectious disease ecology Effects of Ecosystems on Disease and of Disease on Ecosystems (eds., Ostfeld, R., Keesing, F. & Eviner, V.) Book 1, pp. 71–99. Princeton University Press, Princeton, New Jersey.
- Jones, P.D. 2006. Water quality and fisheries in the Mersey estuary, England: a historical perspective. Marine Pollution Bulletin 53, 144–154.
- Kearney, M.S. & Turner, R.E. 2016. Microtidal marshes: can these widespread and fragile marshes survive increasing climate-sea level variability and human action. Journal of Coastal Research 32, 686–699.
- Kennedy, C.J. & Walsh, P.J. 1997. Effects of temperature on xenobiotic metabolism. Seminar Series Society for Experimental Biology, Book 61. Cambridge University Press
- Kerr, L.A., Secor, D.H. & Piccoli, P.M. 2009. Partial migration of fishes as exemplified by the estuarine-dependent white perch. Fisheries 34, 114–123.
- Killen, S.S., Marras, S., Metcalfe, N.B., et al. 2013. Environmental stressors alter relationships between physiology and behaviour. Trends in Ecology & Evolution 28, 651–658.
- Kimmerer, W.J. 2002a. Effects of freshwater flow on abundance of estuarine organisms: physical effects or trophic linkages? Marine Ecology Progress Series 243, 39–55.
- Kimmerer, W.J. 2002b. Physical, biological, and management responses to variable freshwater flow into the San Francisco Estuary. Estuaries 25, 1275–1290.
- Kingsford, R.T., Walker, K.F., Lester, R.E., et al. 2011. A Ramsar wetland in crisis the Coorong, Lower Lakes and Murray Mouth, Australia. Marine and Freshwater Research 62, 255–265.

- Kirwan, M.L., Guntenspergen, G.R., d'Alpaos, A., et al. 2010. Limits on the adaptability of coastal marshes to rising sea level. Geophysical Research Letters 37, L23401.
- Kirwan, M.L. & Megonigal, J.P. 2013. Tidal wetland stability in the face of human impacts and sea-level rise. Nature 504, 53–60.
- Klausmeyer, K.R. & Shaw, M.R. 2009. Climate change, habitat loss, protected areas and the climate adaptation potential of species in Mediterranean ecosystems worldwide. PLoS One 4, e6392.
- Knowles, N. & Cayan, D.R. 2002. Potential effects of global warming on the Sacramento/San Joaquin watershed and the San Francisco estuary. Geophysical Research Letters 29, 38-31–38-34.
- Knowles, N. & Cayan, D.R. 2004. Elevational dependence of projected hydrologic changes in the San Francisco estuary and watershed. Climatic Change 62, 319–336.
- Koehn, J.D., Hobday, A.J., Pratchett, M.S., et al. 2011. Climate change and Australian marine and freshwater environments, fishes and fisheries: synthesis and options for adaptation. Marine and Freshwater Research 62, 1148–1164.
- Kulbicki, M., Parravicini, V., Mouillot, D. 2015. Patterns and processes in reef fish body size. In: Ecology of Fishes on Coral Reefs (ed., Mora, C.), pp. 104–115. Cambridge University Press, Cambridge.
- Lacroix, G., Barbut, L. & Volckaert, F.A.M. 2018. Complex effect of projected sea temperature and wind change on flatfish dispersal. Global Change Biology 24, 85–100.
- Lafferty, K.D. 2009. The ecology of climate change and infectious diseases. Ecology 90, 888-900.
- Lafferty, K.D. & Holt, R.D. 2003. How should environmental stress affect the population dynamics of disease? Ecology Letters 6, 654–664.
- Lassalle, G. & Rochard, E. 2009. Impact of twenty-first century climate change on diadromous fish spread over Europe, North Africa and the Middle East. Global Change Biology 15, 1072–1089.
- Lauchlan, S.S. & Nagelkerken, I. 2019. Species range shifts along multistressor mosaics in estuarine environments under future climate. Fish and Fisheries 21, 32–46.
- Lavergne, E., Pedron, N., Calves, I., et al. 2015. Does the chronic chemical contamination of a European flounder population decrease its thermal tolerance? Marine Pollution Bulletin 95, 658–664.
- Le Pape, O., Chauvet, F., Désaunay, Y., et al. 2003. Relationship between interannual variations of the river plume and the extent of nursery grounds for the common sole (*Solea solea*, L.) in Vilaine Bay. Effects on recruitment variability. Journal of Sea Research 50, 177–185.
- Le Quesne, W.J. & Pinnegar, J.K. 2012. The potential impacts of ocean acidification: scaling from physiology to fisheries. Fish and Fisheries 13, 333–344.
- Legendre, P., Borcard, D., Peres-Neto, P.R. 2005. Analyzing beta diversity: partitioning the spatial variation of community composition data. Ecological Monographs 75, 435–450.
- Lehman, P.W. 2000. The influence of climate on phytoplankton community biomass in San Francisco Bay Estuary. Limnology and Oceanography 45, 580–590.
- Lehman, P.W. 2004. The influence of climate on mechanistic pathways that affect lower food web production in Northern San Francisco Bay estuary. Estuaries 27, 311–324.
- Leis, J.M. 2018. Paradigm lost: ocean acidification will overturn the concept of larval-fish biophysical dispersal. Frontiers in Marine Science 5, 47.
- Lellis-Dibble, K.A., McGlynn, K., Bigford, T.E. 2008. Estuarine fish and shellfish species in US commercial and recreational fisheries: economic value as an incentive to protect and restore estuarine habitat. NOAA Technical Memorandum NMFS-F/SPO-90. U.S. Department of Commerce, National Oceanic and Atmospheric Administration National, Marine Fisheries Service.
- Lennox, R.J., Paukert, C.P., Aarestrup, K., et al. 2019. One hundred pressing questions on the future of global fish migration science, conservation, and policy. Frontiers in Ecology and Evolution 7, 286.

- Lenoir, S., Beaugrand, G. & Lecuyer, É. 2011. Modelled spatial distribution of marine fish and projected modifications in the North Atlantic Ocean. Global Change Biology 17, 115–129.
- Lett, C., Ayata, S.-D., Huret, M., et al. 2010. Biophysical modelling to investigate the effects of climate change on marine population dispersal and connectivity. Progress in Oceanography 87, 106–113.
- Levin, L.A., Ekau, W., Gooday, A.J., et al. 2009. Effects of natural and human-induced hypoxia on coastal benthos. Biogeosciences 6, 2063–2098.
- Lifavi, D.M., Targett, T.E. & Grecay, P.A. 2017. Effects of diel-cycling hypoxia and acidification on juvenile weakfish *Cynoscion regalis* growth, survival, and activity. Marine Ecology Progress Series 564, 163–174.
- Limburg, K.E. & Waldman, J.R. 2009. Dramatic declines in North Atlantic diadromous fishes. BioScience 59, 955–965.
- Loneragan, N.R. 1999. River flows and estuarine ecosystems: implications for coastal fisheries from a review and a case study of the Logan River, southeast Queensland. Australian Journal of Ecology 24, 431–440.
- Long, W.C. & Seitz, R.D. 2008. Trophic interactions under stress: hypoxia enhances foraging in an estuarine food web. Marine Ecology Progress Series 362, 59–68.
- Lonthair, J., Ern, R., Esbaugh, A.J., et al. 2017. The early life stages of an estuarine fish, the red drum (*Sciaenops ocellatus*), are tolerant to high pCO<sub>2</sub>. ICES Journal of Marine Science 74, 1042–1050.
- Ludsin, S.A., Zhang, X., Brandt, S.B., et al. 2009. Hypoxia-avoidance by planktivorous fish in Chesapeake Bay: implications for food web interactions and fish recruitment. Journal of Experimental Marine Biology and Ecology 381, S121–S131.
- Luiz, O.J., Allen, A.P., Robertson, D.R., et al. 2013. Adult and larval traits as determinants of geographic range size among tropical reef fishes. Proceedings of the National Academy of Sciences 110, 16498–16502.
- Lund-Hansen, L.C., Jensen, K.T., Andersen, T.J., et al. 2018. Impacts and effects of a historical high and ENSO linked freshwater inflow in the tropical estuary Nha Phu, southeast Vietnam. Regional Studies in Marine Science 17, 28–37.
- Lushchak, V.I. & Bagnyukova, T.V. 2006. Temperature increase results in oxidative stress in goldfish tissues. 2. Antioxidant and associated enzymes. Comparative Biochemistry and Physiology Part C: Toxicology & Pharmacology 143, 36–41.
- Lutjeharms, J.R.E., Monteiro, P.M.S., Tyson, P.D., et al. 2001. The oceans around southern Africa and regional effects of global change. South African Journal of Science 97, 119–130.
- Mac Nally, R., Thomson, J.R., Kimmerer, W.J., et al. 2010. Analysis of pelagic species decline in the upper San Francisco Estuary using multivariate autoregressive modeling (MAR). Ecological Applications 20, 1417–1430.
- Macchi, G.J., Acha, E.M. & Lasta, C. 1996. Desove y fecundidad de la corvina rubia *Micropogonias furnieri* Desmarest, 1823 del estuario del Río de la Plata, Argentina. Boletín Instituto Español de Oceanografía 12, 99–113.
- Macdonald, R. 2000. Arctic estuaries and ice: a positive negative estuarine couple. In: The Freshwater Budget of the Arctic Ocean (eds., Lewis, E.L., Jones, E.P., Lemke, P., Prowse, T.D., & Wadhams, P.), pp. 383–407. Springer, Dordrecht.
- Mach, M.E., Martone, R.G. & Chan, K.M.A. 2015. Human impacts and ecosystem services: Insufficient research for trade-off evaluation. Ecosystem Services 16, 112–120.
- Madeira, D., Narciso, L., Cabral, H.N., et al. 2013. Influence of temperature in thermal and oxidative stress responses in estuarine fish. Comparative Biochemistry and Physiology Part A: Molecular & Integrative Physiology 166, 237–243.
- Maes, J., Stevens, M., Breine, J. 2007. Modelling the migration opportunities of diadromous fish species along a gradient of dissolved oxygen concentration in a European tidal watershed. Estuarine, Coastal and Shelf Science 75, 151–162.
- Magurran, A.E. & Henderson, P.A. 2003. Explaining the excess of rare species in natural species abundance distributions. Nature 422, 714.
- Mahardja, B., Farruggia, M.J., Schreier, B., et al. 2017. Evidence of a shift in the littoral fish community of the Sacramento-San Joaquin Delta. PLoS One 12, e0170683.

Mann, K.H. 2009. Ecology of Coastal Waters: With Implications for Management. John Wiley & Sons

- Marcek, B.J., Brill, R.W. & Fabrizio, M.C. 2019. Metabolic scope and hypoxia tolerance of Atlantic croaker (*Micropogonias undulatus* Linnaeus, 1766) and spot (*Leiostomus xanthurus* Lacepede, 1802), with insights into the effects of acute temperature change. Journal of Experimental Marine Biology and Ecology 516, 150–158.
- Marcogliese, D.J. 2001. Implications of climate change for parasitism of animals in the aquatic environment. Canadian Journal of Zoology 79, 1331–1352.
- Marcogliese, D.J. 2008. The impact of climate change on the parasites and infectious diseases of aquatic animals. Revue Scientifique et Technique 27, 467–484.
- Maree, R.C., Whitfield, A.K. & Booth, A.J. 2000. Effect of water temperature on the biogeography of South African estuarine fishes associated with the subtropical/warm temperate subtraction zone. South African Journal of Science 96, 184–188.
- Marengo, J.A., Alves, L.M., Alvala, R.C.S., et al. 2018. Climatic characteristics of the 2010-2016 drought in the semiarid Northeast Brazil region. Anais da Academia Brasileira de Ciências 90, 1973–1985.
- Martin, T.J., Cyrus, D.P. & Forbes, A.T. 1992. Episodic events: the effects of cyclonic flushing on the ichthyoplankton of St Lucia estuary on the southeast coast of Africa. Netherlands Journal of Sea Research 30, 273–278.
- Martinho, F., Azeiteiro, U.M., Cabral, H.N., et al. 2012. Estuarine nurseries for marine fish. Management of Environmental Quality: An International Journal 23, 414–433.
- Martinho, F., Leitao, R., Viegas, I., et al. 2007. The influence of an extreme drought event in the fish community of a southern Europe temperate estuary. Estuarine, Coastal and Shelf Science 75, 537–546.
- Martino, E.J. & Able, K.W. 2003. Fish assemblages across the marine to low salinity transition zone of a temperate estuary. Estuarine, Coastal and Shelf Science 56, 969–987.
- Mbande, S., Whitfield, A.K., Cowley, P.D. 2005. The ichthyofaunal composition of the Mngazi and Mngazana estuaries: a comparative study. Smithiana 4, 1–20.
- McCann, K.S. 2000. The diversity-stability debate. Nature 405, 228-233.
- McCarty, J.P. 2001. Ecological consequences of recent climate change. Conservation Biology 15, 320–331.
- Meccia, V.L., Simionato, C.G., Guerrero, R.A. 2013. The Río de la Plata estuary response to wind variability in synoptic timescale: salinity fields and salt wedge structure. Journal of Coastal Research 29, 61–77.
- Medeiros, A.P.M., Xavier, J.H., da Silva, M.B., et al. 2018. Distribution patterns of the fish assemblage in the Mamanguape River Estuary, North-eastern Brazil. Marine Biology Research 14, 524–536.
- Meehl, G.A., Covey, C., Delworth, T., et al. 2007. THE WCRP CMIP3 multimodel dataset: a new era in climate change research. Bulletin of the American Meteorological Society 88, 1383–1394.
- Mekong River Commission. 2017. The Council study: study on the sustainable management and development of the Mekong River, including impacts of mainstream hydropower projects. Biological Resource Assessment Interim Technical Report 1: Volume 1 Specialists Report. Mekong River Commission, Vientiane, Lao PDR.
- Milessi, A.C., Bruno, I., Cozzolino, E., et al. 2018. Cambio climático global frente a las costas de Mar del Plata: evidencias de tropicalización? Informe de Investigación INIDEP 131, 11.
- Milessi, A.C., Bruno, I., Cozzolino, E., et al. 2019. Aluterus monoceros (Acanthopterygii, Tetraodontiformes) southwards distribution range extension In Argentine waters. Marine and Fishery Sciences 32, 43–46.
- Miller, A.W., Reynolds, A.C., Sobrino, C., et al. 2009. Shellfish face uncertain future in high CO<sub>2</sub> world: influence of acidification on oyster larvae calcification and growth in estuaries. Plos One 4, e5661.
- Miller, K.G., Kopp, R.E., Horton, B.P., et al. 2013. A geological perspective on sea-level rise and its impacts along the U.S. mid-Atlantic coast. Earth's Future 1, 3–18.
- Miller, R.L. & McPherson, B.F. 1991. Estimating estuarine flushing and residence times in Charlotte Harbor, Florida via salt balance and a box model. Limnology and Oceanography 36, 602–612.

- Miller, S.H., Breitburg, D.L., Burrell, R.B., et al. 2016. Acidification increases sensitivity to hypoxia in important forage fishes. Marine Ecology Progress Series 549, 1–8.
- Mohan, J. & Walther, B. 2016. Out of breath and hungry: natural tags reveal trophic resilience of Atlantic croaker to hypoxia exposure. Marine Ecology Progress Series 560, 207–221.
- Monaco, M.E., Emmett, R.L., Nelson, D.M., et al. 1990. Distribution and abundance of fishes and invertebrates in West Coast estuaries Volume I: data summaries. ELMR Rp No 4 NOAA/NOS Strategic Environmental Assessments Division, Silver Spring, MD.
- Montagna, P.A. & Kalke, R.D. 1992. The effect of freshwater inflow on meiofaunal and macrofaunal populations in the Guadalupe and Nueces estuaries, Texas. Estuaries 15, 307–326.
- Moore, K.A. & Jarvis, J.C. 2008. Environmental factors affecting recent summertime eelgrass diebacks in the lower Chesapeake Bay: implications for long-term persistence. Journal of Coastal Research, 135–147.
- Morgan, J.D., Sakamoto, T., Grau, E.G., et al. 1997. Physiological and respiratory responses of the Mozambique tilapia (*Oreochromis mossambicus*) to salinity acclimation. Comparative Biochemistry and Physiology Part A: Physiology 117, 391–398.
- Morrongiello, J.R., Beatty, S.J., Bennett, J.C., et al. 2011. Climate change and its implications for Australia's freshwater fish. Marine and Freshwater Research 62, 1082–1098.
- Morson, J.M., Grothues, T., Able, K.W. 2019. Change in larval fish assemblage in a USA east coast estuary estimated from twenty-six years of fixed weekly sampling. PLoS One 14(11), e0225526.
- Mouillot, D., Graham, N.A., Villeger, S., et al. 2013. A functional approach reveals community responses to disturbances. Trends in Ecology & Evolution 28, 167–177.
- Moyle, P.B. 2008. The future of fish is response to large-scale change in the San Francisco Estuary, California. American Fisheries Society Symposium 64, 357–374.
- Moyle, P.B. & Stompe, D.K. 2022. Chapter 11. Non-native fishes in estuaries. In: Fish and Fisheries in Estuaries: A Global Perspective (ed., Whitfield, A.K., Able, K.W., Blaber, S.J.M., et al.), pp. 684–705. Wiley Blackwell, Oxford, UK.
- Munday, B., Zilberg, D. & Findlay, V. 2001. Gill disease of marine fish caused by infection with *Neoparamoeba pemaquidensis*. Journal of Fish Diseases 24, 497–507.
- Munday, P.L. 2014. Transgenerational acclimation of fishes to climate change and ocean acidification. F1000prime reports 6, 99.
- Munday, P.L., Dixson, D.L., Donelson, J.M., et al. 2009. Ocean acidification impairs olfactory discrimination and homing ability of a marine fish. Proceedings of the National Academy of Sciences 106, 1848–1852.
- Munday, P.L., Dixson, D.L., McCormick, M.I., et al. 2010. Replenishment of fish populations is threatened by ocean acidification. Proceedings of the National Academy of Sciences 107, 12930–12934.
- Munday, P.L., Jones, G.P., Pratchett, M.S., et al. 2008. Climate change and the future for coral reef fishes. Fish and Fisheries 9, 261–285.
- Murawski, S.A. 1993. Climate change and marine fish distributions: forecasting from historical analogy. Transactions of the American Fisheries Society 122, 647–658.
- Næsje, T.F., Cowley, P.D., Diserud, O.H., et al. 2012. Riding the tide: estuarine movements of a sciaenid fish, *Argyrosomus japonicus*. Marine Ecology Progress Series 460, 221–232.
- Nagelkerken, I. 2009. Evaluation of nursery function of mangroves and seagrass beds for tropical decapods and reef fishes: patterns and underlying mechanisms. In: Ecological Connectivity Among Tropical Coastal Ecosystems (ed., Nagelkerken, I.), pp. 357–399. Springer
- Nagy, G., Pshennikoy-Severova, V. & Robatto, P. 2002. Monthly variation of salinity in the frontal zone of the Río de la Plata off Montevideo, in response to consecutive ENSO fluctuations and the flow of the Uruguay River (1998–2000). In: The Río de la Plata Research to Manage the Environment, Fish Resources and the Fishery in the Saline Front (ed., Vizziano, D.), pp. 21–30. Ecoplata Program, Uruguay.

- Neuman, M.J. & Able, K.W. 2003. Inter-cohort differences in spatial and temporal settlement patterns of young-ofthe-year windowpane, *Scopthalmus aquosus*, in southern New Jersey. Estuarine, Coastal and Shelf Science 56, 527–538.
- Nguyen, T.T., Truong, T.T., Vu, V.A., et al. 2002. Fish larvae and their variation in the beginning of flood season in the Mekong Delta. In: Journal of the Mekong Fisheries (eds., Nguyen, V.H., Nguyen, T., Nguyen, V.T., et al.), pp. 9–22. Bien Dong.
- Nicholls, R.J., Wong, P.P., Burkett, V., et al. 2007. Coastal systems and low-lying areas. In: Climate Change 2007: Impacts, Adaptation and Vulnerability Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (eds., Parry, M.L., Canziani, O.F., Palutikof, J.P., et al.), pp. 315–356. Cambridge University Press, Cambridge, UK.
- Nichols, F.H., Cloern, J.E., Luoma, S.N., et al. 1986. The modification of an estuary. Science 231, 567–573.
- Nicola, G.G., Elvira, B., Jonsson, B., et al. 2018. Local and global climatic drivers of Atlantic salmon decline in southern Europe. Fisheries Research 198, 78–85.
- Nicolas, D., Chaalali, A., Drouineau, H., et al. 2011. Impact of global warming on European tidal estuaries: some evidence of northward migration of estuarine fish species. Regional Environmental Change 11, 639–649.
- Nicolas, D., Lobry, J., Lepage, M., et al. 2010. Fish under influence: a macroecological analysis of relations between fish species richness and environmental gradients among European tidal estuaries. Estuarine, Coastal and Shelf Science 86, 137–147.
- Nilsson, G.E., Dixson, D.L., Domenici, P., et al. 2012. Near-future carbon dioxide levels alter fish behaviour by interfering with neurotransmitter function. Nature Climate Change 2, 201–204.
- Nodo, P., James, N.C., Childs, A.R., et al. 2017. The impact of river flooding and high flow on the demersal fish assemblages of the freshwater-dominated Great Fish Estuary, South Africa. African Journal of Marine Science 39, 491–502.
- Nodo, P., James, N.C., Childs, A.R., et al. 2018. Response of demersal fish assemblages to an extreme flood event in a freshwater-deprived estuary in South Africa. Marine and Freshwater Research 69, 253–266.
- Norbis, W., Paesch, L., Verocai, J. & Villamarin, G. 1992. Cambios en la estructura de la población de corvina rubia (*Micropogonias furnieri*) capturada en la pesquería artesanal de Pajas Blancas, Montevideo, Uruguay. Frente Marit 11, 37–45.
- Oberdoff, T., Guégan, J.F., Hugueny, B. 1995. Global scale patterns of fish species richness in rivers. Ecography 18, 345–352.
- Olin, J.A., Cerrato, R.M., Nye, J.A., et al. 2019. Evidence for ecosystem changes within a temperate lagoon following a hurricane-induced barrier island breach. Estuaries and Coasts 43, 1625–1639.
- Oliveira, R.E.M.C.C. & Pessanha, A.L.M. 2014. Fish assemblages along a morphodynamic continuum on three tropical beaches. Neotropical Ichthyology 12, 165–175.
- Orth, R.J., Heck, K.L. & van Montfrans, J. 1984. Faunal communities in seagrass beds: a review of the influence of plant structure and prey characteristics on predator: prey relationships. Estuaries 7, 339–350.
- Paerl, H.W. & Huisman, J. 2009. Climate change: a catalyst for global expansion of harmful cyanobacterial blooms. Environmental Microbiology Reports 1, 27–37.
- Pankhurst, N.W. & Munday, P.L. 2011. Effects of climate change on fish reproduction and early life history stages. Marine and Freshwater Research 62, 1015–1026.
- Pasquaud, S., Béguer, M., Larsen, M.H., et al. 2012. Increase of marine juvenile fish abundances in the middle Gironde estuary related to warmer and more saline waters, due to global changes. Estuarine, Coastal and Shelf Science 104–105, 46–53.
- Patra, R.W., Chapman, J.C., Lim, R.P., et al. 2007. The effects of three organic chemicals on the upper thermal tolerances of four freshwater fishes. Environmental Toxicology and Chemistry 26, 1454–1459.

- Paulic, J.E. & Papst, M.H. 2013. Larval and early juvenile fish distribution and assemblage structure in the Canadian Beaufort Sea during July–August, 2005. Journal of Marine Systems 127, 46–54.
- Peck, M.A., Catalán, I.A., Damalas, D., et al. 2020. Climate Change and European Fisheries and Aquaculture: 'CERES' Project Synthesis Report. Hamburg.
- Pereira, M.P.S., Justino, F., Malhado, A.C.M., et al. 2014. The influence of oceanic basins on drought and ecosystem dynamics in Northeast Brazil. Environmental Research Letters 9, 124013.
- Perkins, S.E. & Pitman, A.J. 2008. Do weak AR4 models bias projections of future climate changes over Australia? Climatic Change 93, 527–558.
- Perry, A.L., Low, P.J., Ellis, J.R., et al. 2005. Climate change and distribution shifts in marine fishes. Science 308, 1912–1915.
- Pihl, L., Baden, S.P., Diaz, R.J., et al. 1992. Hypoxia-induced structural-changes in the diet of bottom-feeding fish and crustacea. Marine Biology 112, 349–361.
- Pistevos, J.C., Nagelkerken, I., Rossi, T., et al. 2017. Ocean acidification alters temperature and salinity preferences in larval fish. Oecologia 183, 545–553.
- Pittock, J. & Finlayson, C.M. 2011. Australia's Murray Darling Basin: freshwater ecosystem conservation options in an era of climate change. Marine and Freshwater Research 62, 232–243.
- Pollard, D.A. & Hannan, J.C. 1994. The ecological effects of structural flood mitigation works on fish habitats and fish communities in the lower Clarence River system of south-eastern Australia. Estuaries 17, 427.
- Pomfret, J.R., Elliott, M., O'Reilly, M.G., et al. 1991. Temporal and spatial patterns in the fish communities of two North Sea estuaries. In: Estuaries & Coasts: Spatial and Temporal Intercomparisons (eds., Elliott, M. & Ducrotoy, J.-P.), pp. 277–274. Olsen & Olsen, Fredensborg, Denmark.
- Portner, H.O. & Knust, R. 2007. Climate change affects marine fishes through the oxygen limitation of thermal tolerance. Science 315, 95–97.
- Portner, H.O. & Peck, M.A. 2010. Climate change effects on fishes and fisheries: towards a cause-and-effect understanding. Journal of Fish Biology 77, 1745–1779.
- Possamai, B., Vieira, J.P., Grimm, A.M., et al. 2018. Temporal variability (1997-2015) of trophic fish guilds and its relationships with El Niño events in a subtropical estuary. Estuarine, Coastal and Shelf Science 202, 145–154.
- Potter, I., Tweedley, J., Elliott, M., et al. 2015. The guilds representing the different ways fish use estuaries: a refinement and expansion. Fish and Fisheries 16, 230–239.
- Potter, I.C., Chuwen, B.M., Hoeksema, S.D., et al. 2010. The concept of an estuary: a definition that incorporates systems which can become closed to the ocean and hypersaline. Estuarine, Coastal and Shelf Science 87, 497–500.
- Potter, I.C. & Hyndes, G.A. 1999. Characteristics of the ichthyofaunas of southwestern Australian estuaries, including comparisons with holarctic estuaries and estuaries elsewhere in temperate Australia: a review. Austral Ecology 24, 395–421.
- Potter, I.C., Veale, L., Tweedley, J.R., et al. 2016. Decadal changes in the ichthyofauna of a eutrophic estuary following a remedial engineering modification and subsequent environmental shifts. Estuarine, Coastal and Shelf Science 181, 345–363.
- Potts, W.M., Götz, A., James, N. 2015. Review of the projected impacts of climate change on coastal fishes in southern Africa. Reviews in Fish Biology and Fisheries 25, 603–630.
- Prowse, T.D., Wrona, F.J., Reist, J.D., et al. 2006. General features of the arctic relevant to climate change in freshwater ecosystems. Ambio 35, 330–338.
- Pruzzo, C., Huq, A., Colwell, R.R., et al. 2005. Pathogenic Vibrio species in the marine and estuarine environment. In: Oceans and Health: Pathogens in the Marine Environment (eds., Belkin, S. & Colwell, R.R.), pp. 217–252. Springer, Boston, MA.

- Rahman, M.J. & Cowx, I.G. 2006. Lunar periodicity in growth increment formation in otoliths of hilsa shad (*Tenualosa ilisha*, Clupeidae) in Bangladesh waters. Fisheries Research 81, 342–344.
- Rao, M., Saw, H., Platt, S.G., et al. 2013. Biodiversity conservation in a changing climate: a review of threats and implications for conservation planning in Myanmar. Ambio 42, 789–804.
- Reid, G.K. 1961. Ecology of Inland Waters and Estuaries. Reinhold Books, New York, NY, USA.
- Reis-Santos, P., Tanner, S.E., Vasconcelos, R.P., et al. 2013. Connectivity between estuarine and coastal fish populations: contributions of estuaries are not consistent over time. Marine Ecology Progress Series 491, 177–186.
- Remane, A. 1934. Die brackwasserfauna. Verh dt zool Ges 36, 34-74.
- Reum, J.C.P., Esstington, T.E., Greene, C.M., et al. 2011. Multiscale influence of climate on estuarine populations of forage fish: the role of coastal upwelling, freshwater flow and temperature. Marine Ecology Progress Series 425, 203–215.
- Rice, J.C. 2005. Understanding fish habitat ecology to achieve conservation. Journal of Fish Biology 67, 1–22.
- Rice, J.C. & Garcia, S.M. 2011. Fisheries, food security, climate change, and biodiversity: characteristics of the sector and perspectives on emerging issues. ICES Journal of Marine Science 68, 1343–1353.
- Richter, B. & Thomas, G. 2007. Restoring environmental flows by modifying dam operations. Ecology and Society 12, 12.
- Rijnsdorp, A.D., Peck, M.A., Engelhard, G.H., et al. 2009. Resolving the effect of climate change on fish populations. ICES Journal of Marine Science 66, 1570–1583.
- Robertson, A.W. & Mechoso, C.R. 1998. Interannual and decadal cycles in river flows of southeastern South America. Journal of Climate 11, 2570–2581.
- Robins, J.B., Halliday, I.A., Staunton-Smith, J., et al. 2005. Freshwater-flow requirements of estuarine fisheries in tropical Australia: a review of the state of knowledge and application of a suggested approach. Marine and Freshwater Research 56, 343–360.
- Robins, P.E., Skov, M.W., Lewis, M.J., et al. 2016. Impact of climate change on UK estuaries: a review of past trends and potential projections. Estuarine, Coastal and Shelf Science 169, 119–135.
- Robinson, P., Leight, A., Trueblood, D., et al. 2013. Climate sensitivity of the national estuarine research reserve system. NOAA Special Report.
- Robson, B.J. & Hamilton, D.P. 2003. Summer flow event induces a cyanobacterial bloom in a seasonal Western Australian estuary. Marine and Freshwater Research 54, 139–151.
- Rodrigues, K.A., Macchi, G.J., Acha, E.M., et al. 2008. Spawning and fecundity of Jenyns's sprat, *Ramnogaster arcu-ata*, a winter spawner in the temperate waters of the Río de la Plata estuary, Argentina–Uruguay. Journal of the Marine Biological Association of the United Kingdom 88, 423–429.
- Rodriguez, J.F., Saco, P.M., Sandi, S., et al. 2017. Potential increase in coastal wetland vulnerability to sea-level rise suggested by considering hydrodynamic attenuation effects. Nature Communications 8, 16094.
- Roessig, J.M., Woodley, C.M., Cech, J.J., et al. 2004. Effects of global climate change on marine and estuarine fishes and fisheries. Reviews in Fish Biology and Fisheries 14, 251–275.
- Rogers, N.J., Urbina, M.A., Reardon, E.E., et al. 2016. A new analysis of hypoxia tolerance in fishes using a database of critical oxygen level (P crit). Conservation Physiology 4, cow012.
- Roman, C.T. 2017. Salt marsh sustainability: challenges during an uncertain future. Estuaries and Coasts 40, 711–716.
- Roman, M.R., Brandt, S.B., Houde, E.D., et al. 2019. Interactive effects of hypoxia and temperature on coastal pelagic zooplankton and fish. Frontiers in Marine Science 6, 139.
- Rombough, P.J. 1997. The effects of temperature on embryonic and larval development. In: Seminar Series-Society For Experimental Biology Book 61, pp. 177–224. Cambridge University Press
- Rose, K.A., Adamack, A.T., Murphy, C.A., et al. 2009. Does hypoxia have population-level effects on coastal fish? Musings from the virtual world. Journal of Experimental Marine Biology and Ecology 381, S188–S203.
- Rose, K.A., Creekmore, S., Thomas, P., et al. 2018. Modeling the population effects of hypoxia on Atlantic croaker (*Micropogonias undulatus*) in the northwestern Gulf of Mexico: part 1 model description and idealized hypoxia. Estuaries and Coasts 41, 233–254.
- Roussel, J.-M. 2007. Carry-over effects in brown trout (*Salmo trutta*): hypoxia on embryos impairs predator avoidance by alevins in experimental channels. Canadian Journal of Fisheries and Aquatic Sciences 64, 786–792.
- Ruckelshaus, M., Doney, S.C., Galindo, H.M., et al. 2013. Securing ocean benefits for society in the face of climate change. Marine Policy 40, 154–159.
- Sabine, C.L., Feely, R.A., Gruber, N., et al. 2004. The oceanic sink for anthropogenic CO<sub>2</sub>. Science 305, 367–371.
- Sagasti, A., Schaffner, L.C., Duffy, J.E. 2001. Effects of periodic hypoxia on mortality, feeding and predation in an estuarine epifaunal community. Journal of Experimental Marine Biology and Ecology 258, 257–283.
- Sakabe, R. & Lyle, J.M. 2010. The influence of tidal cycles and freshwater inflow on the distribution and movement of an estuarine resident fish *Acanthopagrus butcheri*. Journal of Fish Biology 77, 643–660.
- Sammut, J., White, I. & Melville, M.D. 1996. Acidification of an estuarine tributary in eastern Australia due to drainage of acid sulfate soils. Marine and Freshwater Research 47, 669–684.
- Saraceno, M., Simionato, C.G. & Ruiz-Etcheverry, L.A. 2014. Sea surface height trend and variability at seasonal and interannual time scales in the Southeastern South American continental shelf between 27°S and 40°S. Continental Shelf Research 91, 82–94.
- Sardella, B.A. & Brauner, C.J. 2007. The osmo-respiratory compromise in fish: the effects of physiological state and the environment. In: Fish Respiration and Environment (ed., Fernandes, M.N.), pp. 147–165. CRC Press, Boca Raton.
- Saul, R., Barnes, R. & Elliott, M. 2016. Is climate change an unforeseen, irresistible and external factor A force majeure in marine environmental law? Marine Pollution Bulletin 113, 25–35.
- Scavia, D., Field, J.C., Boesch, D.F., et al. 2002. Climate change impacts on U.S. coastal and marine ecosystems. Estuaries 25, 149–164.
- Schiedek, D., Sundelin, B., Readman, J.W., et al. 2007. Interactions between climate change and contaminants. Marine Pollution Bulletin 54, 1845–1856.
- Schlacher, T.A. & Wooldridge, T.H. 1996. Origin and trophic importance of detritus-evidence from stable isotopes in the benthos of a small, temperate estuary. Oecologia 106, 382–388.
- Schuerch, M., Spencer, T., Temmerman, S., et al. 2018. Future response of global coastal wetlands to sea-level rise. Nature 2018, 231–234.
- Schulte, P.M. 2014. What is environmental stress? Insights from fish living in a variable environment. The Journal of Experimental Biology 217, 23–34.
- Selleslagh, J., Amara, R., Laffargue, P., et al. 2009. Fish composition and assemblage structure in three Eastern English Channel macrotidal estuaries: a comparison with other French estuaries. Estuarine, Coastal and Shelf Science 81, 149–159.
- Serpetti, N., Baudron, A.R., Burrows, M.T., et al. 2017. Impact of ocean warming on sustainable fisheries management informs the ecosystem approach to fisheries. Scientific Reports 7, 13438.
- Sheaves, M. & Johnston, R. 2008. Influence of marine and freshwater connectivity on the dynamics of subtropical estuarine wetland fish metapopulations. Marine Ecology Progress Series 357, 225–243.
- Sheaves, M. & Johnston, R. 2009. Ecological drivers of spatial variability among fish fauna of 21 tropical Australian estuaries. Marine Ecology Progress Series 385, 245–260.
- Sheaves, M., Sporne, I., Dichmont, C.M., et al. 2016. Principles for operationalizing climate change adaptation strategies to support the resilience of estuarine and coastal ecosystems: an Australian perspective. Marine Policy 68, 229–240.
- Shennan, I. & Horton, B. 2002. Holocene land- and sea-level changes in Great Britain. Journal of Quaternary Science 17, 511–526.

- Simionato, C.G., Berasategui, A., Meccia, V.L., et al. 2008. Short time-scale wind forced variability in the Río de la Plata Estuary and its role on ichthyoplankton retention. Estuarine, Coastal and Shelf Science 76, 211–226.
- Simionato, C.G., Dragani, W., Meccia, V., et al. 2004. A numerical study of the barotropic circulation of the Río de la Plata estuary: sensitivity to bathymetry, the Earth's rotation and low frequency wind variability. Estuarine, Coastal and Shelf Science 61, 261–273.
- Simionato, C.G., Luz Clara Tejedor, M., Campetella, C., Guerrero, R.A., & Moreira, M. 2010. Patterns of sea surface temperature variability on seasonal to sub-annual scales at and offshore the Río de la Plata estuary. Continental Shelf Research 30, 1983–1997.
- Simpson, S.D., Munday, P.L., Wittenrich, M.L., et al. 2011. Ocean acidification erodes crucial auditory behaviour in a marine fish. Biology Letters 7, 917–920.
- Sklar, F.H. & Browder, J.A. 1998. Coastal environmental impacts brought about by alterations to freshwater flow in the Gulf of Mexico. Environmental Management 22, 547–562.
- Smyth, K. & Elliott, M. 2016. Effects of changing salinity on the ecology of the marine environment. In: Stressors in the Marine Environment: Physiological and Ecological Responses; Societal Implications (eds., Solan, M. & Whiteley, N.), pp. 161–174. Oxford University Press, Oxford.
- Snickars, M., Weigel, B., Bonsdorff, E. 2015. Impact of eutrophication and climate change on fish and zoobenthos in coastal waters of the Baltic Sea. Marine Biology 162, 141–151.
- Soininen, J., Lennon, J.J., Hillebrand, H. 2007. A multivariate analysis of beta diversity across organisms and environments. Ecology 88, 2830–2838.
- Sokolova, I.M. & Lannig, G. 2008. Interactive effects of metal pollution and temperature on metabolism in aquatic ectotherms: implications of global climate change. Climate Research 37, 181–201.
- Sommer, T., Armor, C., Baxter, R., et al. 2007. The collapse of pelagic fishes in the Upper San Francisco estuary: El Colapso de los Peces Pelagicos en La Cabecera Del Estuario San Francisco. Fisheries 32, 270–277.
- Sosa-Lopez, A. & Mouillot, D. 2007. Spatiotemporal segregation regulates species richness along biomass gradients in estuarine fish assemblages. Marine Ecology Progress Series 342, 217–226.
- Spalding, M.D., Fox, H.E., Allen, G.R., et al. 2007. Marine ecoregions of the World: a bioregionalization of coastal and shelf areas. BioScience 57, 573–583.
- Staudinger, M.D., Mills, K.E., Stamieszkin, K., et al. 2019. It's about time: a synthesis of changing phenology in the Gulf of Maine ecosystem. Fisheries Oceanography 28, 532–566.
- Stone, G.W., Liu, B., Pepper, D.A., et al. 2004. The importance of extratropical and tropical cyclones on the short-term evolution of barrier islands along the northern Gulf of Mexico, USA. Marine Geology 210, 63–78.
- Stott, P. 2016. How climate change affects extreme weather events. Science 352, 1517–1518.
- Sullivan, M.C., Able, K.W., Hare, J.A., et al. 2006. *Anguilla rostrata* glass eel ingress into two U.S. east coast estuaries:patterns, processes and implications for adult abundance. Journal of Fish Biology 69, 1081–1101.
- Sunday, J.M., Bates, A.E. & Dulvy, N.K. 2012. Thermal tolerance and the global redistribution of animals. Nature Climate Change 2, 686–690.
- Sunday, J.M., Bates, A.E., Kearney, M.R., et al. 2014. Thermal-safety margins and the necessity of thermoregulatory behavior across latitude and elevation. Proceedings of the National Academy of Sciences 111, 5610–5615.
- Swanson, C. 1998. Interactive effects of salinity on metabolic rate, activity, growth and osmoregulation in the euryhaline milkfish (*Chanos chanos*). Journal of Experimental Biology 201, 3355–3366.
- Taddese, F., Schallenberg, M., Mikheev, P., et al. 2018. Ichthyofaunal assemblages in shallow littoral habitats of permanently open estuaries and intermittently closed and open lakes or lagoons in Otago, New Zealand. Marine and Freshwater Research 69, 1222–1230.
- Tam, B., Gough, W.A. & Tsuji, L. 2011. The impact of warming on the appearance of furunculosis in fish of the James Bay region, Quebec, Canada. Regional Environmental Change 11, 123–132.

- Tanner, S.E., Reis-Santos, P., Vasconcelos, R.P., et al. 2013. Population connectivity of *Solea solea* and *Solea senega-lensis* over time. Journal of Sea Research 76, 82–88.
- Targett, T.E., Grecay, P.A., Dixon, R.L. 2019. Growth of the estuarine fish *Fundulus heteroclitus* in response to dielcycling hypoxia and acidification: interaction with temperature. Canadian Journal of Fisheries and Aquatic Sciences 76, 1295–1304.
- Teixeira-de Mello, F., Meerhoff, M., Peckan-Hekim, Z., et al. 2009. Substantial differences in littoral fish community structure and dynamics in subtropical and temperate shallow lakes. Freshwater Biology 54, 1202–1215.
- Tett, P., Gowen, R.J., Painting, S.J., et al. 2013. Framework for understanding marine ecosystem health. Marine Ecology Progress Series 494, 1–27.
- Thomas, P. & Rahman, M.S. 2010. Region-wide impairment of Atlantic croaker testicular development and sperm production in the northern Gulf of Mexico hypoxic dead zone. Marine Environmental Research 69, S59–S62.
- Thomas, P. & Rahman, M.S. 2012. Extensive reproductive disruption, ovarian masculinization and aromatase suppression in Atlantic croaker in the northern Gulf of Mexico hypoxic zone. Proceedings of the Royal Society B: Biological Sciences 279, 28–38.
- Thomas, P., Rahman, M.S., Khan, I.A., et al. 2007. Widespread endocrine disruption and reproductive impairment in an estuarine fish population exposed to seasonal hypoxia. Proceedings of the Royal Society B: Biological Sciences 274, 2693–2702.
- Thomson, J.R., Kimmerer, W.J., Brown, L.R., et al. 2010. Bayesian change point analysis of abundance trends for pelagic fishes in the upper San Francisco Estuary. Ecological Applications 20, 1431–1448.
- Thorne, K., MacDonald, G., Guntenspergen, G., et al. 2018. U.S. Pacific coastal wetland resilience and vulnerability to sea-level rise. Science Advances 4, eaao3270.
- Thronson, A. & Quigg, A. 2008. Fifty-five years of fish kills in coastal Texas. Estuaries and Coasts 31, 802-813.
- Tilman, D. 2001. Functional diversity. Encyclopedia of Biodiversity 3, 109-120.
- Tisseuil, C., Cornu, J.F., Beauchard, O., et al. 2013. Global diversity patterns and cross-taxa convergence in freshwater systems. Journal of Animal Ecology 82, 365–376.
- Tittensor, D.P., Mora, C., Jetz, W., et al. 2010. Global patterns and predictors of marine biodiversity across taxa. Nature 466, 1098.
- Toft, J.D., Munsch, S.H., Cordell, J.R., et al. 2018. Impact of multiple stressors on juvenile fish in estuaries of the northeast Pacific. Global Change Biology 24, 2008–2020.
- Trigo, R.M., Pozo-Vázquez, D., Osborn, T.J., et al. 2004. North Atlantic oscillation influence on precipitation, river flow and water resources in the Iberian Peninsula. International Journal of Climatology 24, 925–944.
- Tweedley, J.R., Dittmann, S.R., Whitfield, A.K., et al. 2019. Hypersalinity: global distribution, causes, and present and future effects on the biota of estuaries and lagoons. In: Coasts and Estuaries (eds., Wolanski, E., Day, J.W., Elliott, M., et al.), pp. 523–546. Elsevier
- Tweedley, J.R., Warwick, R.M. & Potter, I.C. 2016. The contrasting ecology of temperate macrotidal and microtidal estuaries. In: Oceanography and Marine Biology: An Annual Review (eds., Hughes, R.N., Hughes, D.J., Smith, I.P., et al.) Book 54, pp. 73–171. CRC Press, Boca Raton.
- Utida, G., Cruz, F.W., Etourneau, J., et al. 2019. Tropical South Atlantic influence on Northeastern Brazil precipitation and ITCZ displacement during the past 2300 years. Scientific Reports 9, 1698.
- Uyarra, M.C., CÔTÉ, I.M., Gill, J.A., et al. 2005. Island-specific preferences of tourists for environmental features: implications of climate change for tourism-dependent states. Environmental Conservation 32, 11–19.
- Valesini, F.J., Tweedley, J.R., Clarke, K.R., et al. 2013. The importance of regional, system-wide and local spatial scales in structuring temperate estuarine fish communities. Estuaries and Coasts 37, 525–547.

- van Niekerk, L., Adams, J.B., James, N.C., et al. 2020. An estuary ecosystem classification that encompasses biogeography and a high diversity of types in support of protection and management. African Journal of Aquatic Science 45, 199–216.
- Vasconcelos, R., Reis-Santos, P., Cabral, H., et al. 2011. River-coast connectivity, estuarine nursery function and coastal fisheries. In: Treatise on Estuarine and Coastal Science (eds., Wolanski, E. & McLusky, D.), pp. 81–107. Academic Press, Waltham.
- Vasconcelos, R.P., Henriques, S., Franca, S., et al. 2015. Global patterns and predictors of fish species richness in estuaries. Journal of Animal Ecology 84, 1331–1341.
- Vezzulli, L., Colwell, R.R. & Pruzzo, C. 2013. Ocean warming and spread of pathogenic vibrios in the aquatic environment. Microbial Ecology 65, 817–825.
- Vezzulli, L., Previati, M., Pruzzo, C., et al. 2010. Vibrio infections triggering mass mortality events in a warming Mediterranean Sea. Environmental Microbiology 12, 2007–2019.
- Vicente-Serrano, S.M., Lopez-Moreno, J.-I., Beguería, S., et al. 2014. Evidence of increasing drought severity caused by temperature rise in southern Europe. Environmental Research Letters 9, 04401.
- Villéger, S., Ramos Miranda, J., Flores Hernandez, D., et al. 2008. Stable trophic structure across coastal nekton assemblages despite high species turnover. Marine Ecology Progress Series 364, 135–146.
- Vinagre, C., Costa, M.J., Cabral, H.N. 2007. Impact of climate and hydrodynamics on sole larval immigration towards the Tagus estuary, Portugal. Estuarine, Coastal and Shelf Science 75, 516–524.
- Vinagre, C., França, S., Cabral, H. 2006. Diel and semi-lunar patterns in the use of an intertidal mudflat by juveniles of Senegal sole, *Solea senegalensis*. Estuarine, Coastal and Shelf Science 69, 246–254.
- Vinagre, C., Madeira, C., Dias, M., et al. 2019. Reliance of coastal intertidal food webs on river input Current and future perspectives. Ecological Indicators 101, 632–639.
- Vinagre, C., Madeira, D., Narciso, L., et al. 2012. Effect of temperature on oxidative stress in fish: lipid peroxidation and catalase activity in the muscle of juvenile seabass, *Dicentrarchus labrax*. Ecological Indicators 23, 274–279.
- Vinagre, C., Santos, F.D., Cabral, H.N., et al. 2009. Impact of climate and hydrology on juvenile fish recruitment towards estuarine nursery grounds in the context of climate change. Estuarine, Coastal and Shelf Science 85, 479–486.
- Vitousek, P.M., Aber, J.D., Howarth, R.W., et al. 1997. Human alteration of the global nitrogen cycle: sources and consequences. Ecological Applications 7, 737–750.
- Vivier, L., Cyrus, D.P., Jerling, H.L. 2010. Fish community structure of the St Lucia estuarine system under prolonged drought conditions and its potential for recovery after mouth breaching. Estuarine, Coastal and Shelf Science 86, 568–579.
- Vorwerk, P.D., Whitfield, A.K., Cowley, P.D., et al. 2003. The influence of selected environmental variables on fish assemblage structure in a range of Southeast African estuaries. Environmental Biology of Fishes 66, 237–247.
- Wallace, J., Kok, H., Beckley, L., et al. 1984. South African estuaries and their importance to fishes. South African Journal of Science 80, 203–207.
- Walsh, C.T., Reinfelds, I.V., Ives, M.C., et al. 2013. Environmental influences on the spatial ecology and spawning behaviour of an estuarine-resident fish, *Macquaria colonorum*. Estuarine, Coastal and Shelf Science 118, 60–71.
- Walther, G.-R., Post, E., Convey, P., et al. 2002. Ecological responses to recent climate change. Nature 416, 389–395.
- Wannamaker, C.M. & Rice, J.A. 2000. Effects of hypoxia on movements and behavior of selected estuarine organisms from the southeastern United States. Journal of Experimental Marine Biology and Ecology 249, 145–163.
- Wedderburn, S.D., Bailey, C.P., Delean, S., et al. 2016. Population and osmoregulatory responses of a euryhaline fish to extreme salinity fluctuations in coastal lagoons of the Coorong, Australia. Estuarine, Coastal and Shelf Science 168, 50–57.

- Wedderburn, S.D., Barnes, T.C., Hillyard, K.A. 2014. Shifts in fish assemblages indicate failed recovery of threatened species following prolonged drought in terminating lakes of the Murray–Darling Basin, Australia. Hydrobiologia 730, 179–190.
- Weisberg, S.B., Himchak, P., Baum, T., et al. 1996. Temporal trends in abundance of fish in the tidal Delaware River. Estuaries 19, 723–729.
- Weisberg, S.B., Morin, R.P., Ross, E.A., et al. 1986. *Eustrongylides* (Nematoda) infection in mummichogs and other fishes of the Chesapeake Bay region. Transactions of the American Fisheries Society 115, 776–783.
- Westlund, L. 2007. Disaster Response and Risk Management in the Fisheries Sector, Vol 479. Food & Agriculture Org.
- Wetz, M.S. & Yoskowitz, D.W. 2013. An 'extreme' future for estuaries? Effects of extreme climatic events on estuarine water quality and ecology. Marine Pollution Bulletin 69, 7–18.
- Whipple, A., Grossinger, R.M., Rankin, D., et al. 2012. Sacramento-San Joaquin Delta Historical Ecology Investigation: Exploring Pattern and Process. SFEI, Richmond.
- Whitfield, A. & Elliott, M. 2011. Ecosystem and biotic classifications of estuaries and coasts. In: Treatise on Estuarine and Coastal Science (eds., Wolanski, E. & McLusky, D.), pp. 99–124. Academic Press, Waltham.
- Whitfield, A., Elliott, M., Basset, A., et al. 2012. Paradigms in estuarine ecology a review of the Remane diagram with a suggested revised model for estuaries. Estuarine, Coastal and Shelf Science 97, 78–90.
- Whitfield, A. & Harrison, T. 2003. River flow and fish abundance in a South African estuary. Journal of Fish Biology 62, 1467–1472.
- Whitfield, A.K. 1994a. Abundance of larval and 0+ juvenile marine fishes in the lower reaches of three southern African estuaries with differing freshwater inputs. Marine Ecology-Progress Series 105, 257–257.
- Whitfield, A.K. 1994b. Fish species diversity in southern African estuarine systems: an evolutionary perspective. Environmental Biology of Fishes 40, 37–48.
- Whitfield, A.K. 1999. Ichthyofaunal assemblages in estuaries: a South African case study. Reviews in Fish Biology and Fisheries 9, 151–186.
- Whitfield, A.K. 2004. Distribution patterns of fishes in a freshwater deprived Eastern Cape estuary, with particular emphasis on the geographical headwater region. Water SA 29, 61–68.
- Whitfield, A.K. 2005a. Fishes and freshwater in southern African estuaries a review. Aquatic Living Resources 18, 275–289.
- Whitfield, A.K. 2005b. Preliminary documentation and assessment of fish diversity in sub-Saharan African estuaries. African Journal of Marine Science 27, 307–324.
- Whitfield, A.K., James, N.C., Lamberth, S.J., et al. 2016. The role of pioneers as indicators of biogeographic range expansion caused by global change in southern African coastal waters. Estuarine, Coastal and Shelf Science 172, 138–153.
- Whitfield, A.K. & Paterson, A.W. 2003. Distribution patterns of fishes in a freshwater deprived Eastern Cape estuary, with particular emphasis on the geographical headwater region. Water SA 29, 61–68.
- Whittaker, R.J., Willis, K.J., Field, R. 2001. Scale and species richness: towards a general, hierarchical theory of species diversity. Journal of Biogeography 28, 453–470.
- Wiens, J.J. 2011. The niche, biogeography and species interactions. Philosophical Transactions of the Royal Society B: Biological Sciences 366, 2336–2350.
- Williams, J., Hindell, J.S., Jenkins, G.P., et al. 2017. The influence of freshwater flows on two estuarine resident fish species show differential sensitivity to the impacts of drought, flood and climate change. Environmental Biology of Fishes 100, 1121–1137.
- Williams, S.E., Hobday, A.J., Falconi, L., et al. 2020. Research priorities for natural ecosystems in a changing global climate. Global Change Biology 26, 410–416.

- Willig, M.R., Kaufman, D.M., Stevens, R.D. 2003. Latitudinal gradients of biodiversity: pattern, process, scale, and synthesis. Annual Review of Ecology, Evolution, and Systematics 34, 273–309.
- Wolanski, E., Day, J.W., Elliott, M., et al. (eds) 2019. Coasts and Estuaries: The Future. Elsevier, Amsterdam.
- Wolanski, E. & Elliott, M. 2016. 6 Ecohydrology models. In: Estuarine Ecohydrology, 2nd Edition (eds., Wolanski, E. & Elliott, M.), pp. 195–218. Elsevier, Boston.
- Wong, V.N.L., Walsh, S., Morris, S. 2018. Climate affects fish-kill events in subtropical estuaries of eastern Australia. Marine and Freshwater Research 69, 1641–1648.
- Wood, C.M. 2018. The fallacy of the *Pcrit* are there more useful alternatives? Journal of Experimental Biology 221, jeb163717.
- Woodroffe, C.D. 2018. Mangrove response to sea level rise: palaeoecological insights from macrotidal systems in northern Australia. Marine and Freshwater Research 69, 917–932.
- Wooldridge, T. 1999. Estuarine zooplankton community structure and dynamics. Estuaries of South Africa 7, 141–166.
- Wooldridge, T.H., Adams, J.B., Fernandes, M. 2016. Biotic responses to extreme hypersalinity in an arid zone estuary, South Africa. South African Journal of Botany 107, 160–169.
- Wootton, R.J. 1990. Ecology of Teleost Fishes. Fish and Fisheries Series 1. Chapman & Hall, London, 404 pp.
- Wright, D.H. 1983. Species-energy theory: an extension of species-area theory. Oikos, 496-506.
- Wu, R.S. 2002. Hypoxia: from molecular responses to ecosystem responses. Marine Pollution Bulletin 45, 35-45.
- Wylie, J.L. & Currie, D.J. 1993. Species-energy theory and patterns of species richness: I. Patterns of bird, angiosperm, and mammal species richness on islands. Biological Conservation 63, 137–144.
- Yang, L.H. & Rudolf, V.H. 2010. Phenology, ontogeny and the effects of climate change on the timing of species interactions. Ecology Letters 13, 1–10.
- Yang, Z., Wang, T., Voisin, N., et al. 2015. Estuarine response to river flow and sea-level rise under future climate change and human development. Estuarine, Coastal and Shelf Science 156, 19–30.
- Young, G.C. & Potter, I.C. 2002. Influence of exceptionally high salinities, marked variations in freshwater discharge and opening of estuary mouth on the characteristics of the ichthyofauna of a normally-closed estuary. Estuarine, Coastal and Shelf Science 55, 223–246.
- Zampatti, B.P., Bice, C.M., Jennings, P.R. 2010. Temporal variability in fish assemblage structure and recruitment in a freshwater-deprived estuary: the Coorong, Australia. Marine and Freshwater Research 61, 1298–1312.
- Zander, C. & Reimer, L. 2002. Parasitism at the ecosystem level in the Baltic Sea. Parasitology 124, 119–135.
- Zapata, F.A. & Ross Robertson, D. 2007. How many species of shore fishes are there in the Tropical Eastern Pacific? Journal of Biogeography 34, 38–51.
- Zhang, H., Ludsin, S.A., Mason, D.M., et al. 2009. Hypoxia-driven changes in the behavior and spatial distribution of pelagic fish and mesozooplankton in the northern Gulf of Mexico. Journal of Experimental Marine Biology and Ecology 381, S80–S91.
- Zhang, J., Gilbert, D., Gooday, A.J., et al. 2010. Natural and human-induced hypoxia and consequences for coastal areas: synthesis and future development. Biogeosciences 7, 1443–1467.
- Zhong, L., Li, M., Foreman, M.G.G. 2008. Resonance and sea level variability in Chesapeake Bay. Continental Shelf Research 28, 2565–2573.