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Impact of climate change on UK estuaries: A review of past trends and potential projections

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ABSTRACT

UK estuarine environments are regulated by inter-acting physical processes, including tidal, wave, surge, river discharge and sediment supply. They regulate the fluxes of nutrients, pollutants, pathogens and viruses that determine whether coastlines achieve the Good Environmental Status (GEnS) required by the EU's Marine Strategy Directive. We review 20th century trends and 21st century projections of changes to climatic drivers, and their potential for altering estuarine bio-physical processes. Sea-level rise will cause some marine habitats to expand, and others diminish in area extent. The overall consequences of estuarine morphodynamics to these habitat shifts, and vice-versa, are unknown. Increased temperatures could intensify microbial pathogen concentrations and increase public health risk. The patterns of change of other climatic drivers are difficult to predict (e.g., river flows and storm surges). Projected increased winter river flows throughout UK catchments will enhance the risks of coastal eutrophication, harmful algal blooms and hypoxia in some contexts, although there are spatial variabilities in river flow projections. The reproductive success of estuarine biota is sensitive to saline intrusion and corresponding turbidity maxima, which are projected to gradually shift landwards as a result of sea-level rise. Although more-frequent flushing events in winter and longer periods of drought in summer are predicted, whereby the subsequent estuarine mixing and recovery rates are poorly understood. With rising estuarine salinities, subtidal species can penetrate deeper into estuaries, although this will depend on the resilience/adaptation of the species. Many climate and impact predictions lack resolution and spatial cover. Long-term monitoring and increased research, which considers the catchment-river-estuary-coast system as a whole, is needed to support risk predicting and mitigatory strategies.

Contents

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1. Introduction

Estuaries are process centres for coastal hydrological, biogeo-chemical and biological cycles and their biological productivity rivals those of tropical rainforests and coral reefs (Cai, 2011). The high productivity combines with the provisioning of important ecosystem services, such as flood protection and providing recre-ational space, to maintain a Good Environmental Status (GEnS) so
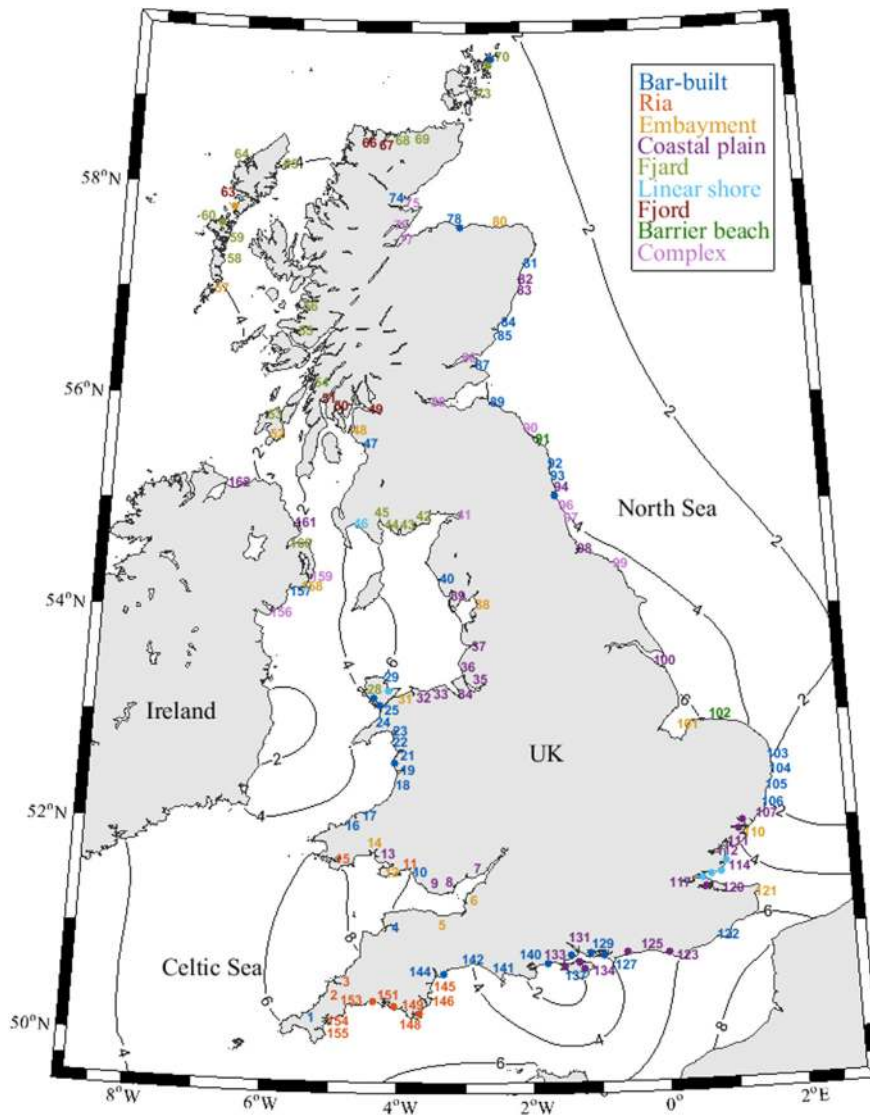
Fig. 1. Location and geomorphological classification of 162 UK estuaries, as defined by Davidson et al. (1991), who did not include estuaries in Northern Ireland. Tidal type can be inferred from tidal range (m) contours (calculated from a 3D ocean model, simulated and validated by Lewis et al., 2014), classified by Davies (1964) as: microtidal (tidal range < 2 m), mesotidal (2 m–4 m), macrotidal (4 m–6 m), and hypertidal (>6 m).

that they are socio-economically valuable and sustainable (Barbier et al., 2011; Elliott et al., 2015). Estuaries are also biologically important; many in the UK are biodiversity rich and most host a series of habitats that are important nursery areas for commercial fisheries species (Beck et al., 2001), or feeding and breeding areas for terrestrial populations, such as birds (Sharps et al., 2015). Many estuarine habitats have statutory protection and all have roles in prolific estuarine foodwebs and the cycling of organic material (Kaiser et al., 2005). The ecological functions of providing habitat and nursery areas are attributed to the discontinuity in productivity, sediment characteristics, and turbidity, when compared with the surrounding coasts (Power et al., 2000). Estuaries are known to trap, filter and recycle suspended particulate matter (Ittekkot, 1988), as well as components that are potentially harmful to human health, such as nutrients and pathogenic viruses. In the UK, estuarine processes are directly affected by intrinsic anthropogenic activities, such as aquaculture and shellfish fisheries, and indirectly by activities such as catchment agriculture, forestation/deforestation, urbanisation and coastal management decisions (Attrill, 1998).

Estuaries are commonly defined as "*a semi-enclosed coastal body of water which has free connection to the open sea, extending into the river as far as the limit of tidal influence and within which seawater is measurably diluted with freshwater derived from land drainage*" (Dyer, 1997). This definition has shortcomings: tidal influence often exceeds the salt water intrusion in the river, and the seaward region of fresh water influence (ROFI) may extend offshore, such as in the Severn Estuary. Thus, estuaries can be further classified based on salinity structure (well mixed through to stratified), tidal range (microtidal to hypertidal) and geomorphology (Appendix 2). A review of 155 UK 'estuaries' classified sites by their geomorphology and tidal type (Fig. 1; Appendix 1; Davidson et al. (1991)). Mesotidal estuaries (2 m < tidal range < 4 m) are typical in the eastern UK, whereas macrotidal (4 m < tidal range < 6 m) or hypertidal (tidal range > 6 m) estuaries occur in the west, and are generally connected to catchments that respond quickly to rainfall events (Brown et al., 1991).

The Intergovernmental Panel on Climate Change (IPCC) is 'virtually certain' that global temperatures will increase during the 21st century, with climate models projecting UK shelf seas to warm by between 1.5 °C and 4 °C by 2100 (Jenkins et al., 2009). In turn, global mean sea-levels are projected to rise by between 44 and 74 cm, depending on our future carbon dioxide emissions (IPCC, 2013), with UK mean sea-level closely matching the global trend (Woodworth et al., 2009). Climate models also predict changes to precipitation pattern in the UK, with more inter-regional and inter-seasonal variability, albeit with high model uncertainty and less confidence than sea-level projections (IPCC, 2013); but with high impact, such as the damaging storm clusters experienced during the winter of 2013/2014 along the western coasts of the UK (Wadey et al., 2014). It follows that the long-term climate-driven impacts that UK estuaries face include a deterioration in water quality and alterations to habitat composition, which could ultimately affect their ecological integrity (Kennish, 2002).

In this paper, we review current understanding of how anthropogenic climate change will impact UK estuaries and their complex and important environments. Trends suggest that, over the 21st century, estuarine ecosystems will be mostly impacted by habitat loss, as a consequence of climate change and urban expansion. Other projected high priority problems include excessive nutrient and sewage inputs to estuaries, leading to eutrophication, hypoxia and anoxia (Kennish, 2002). Further damaging problems will arise from over-fishing, chemical contaminant fluxes in urbanised regions, freshwater diversions, the introduction of invasive species, and coastal subsidence and erosion. Although it is difficult to quantify, these impacts on estuarine ecosystems are at least in part linked to climate change.

We amalgamate current knowledge from peer-reviewed literature on climate change trends and projections for the UK, and discuss how climate change has and will impact upon both physical and ecological processes in UK estuaries. This review paper is structured as follows: In Section 2, we summarise past trends and future projections of UK coastal climate change, relevant to physical estuarine processes; Sections 3 and 4 review observed trends and projected impacts of climate change to the physical and ecological processes in UK estuaries, respectively; finally, discussions and conclusions are summarised in Sections 5 and 6, respectively.

2. Climate changes to physical forcing

We focus on sea surface temperature (SST), sea-level rise and atmospheric shifts as the main climate drivers that influence UK estuarine environments and ecosystems, and their interactions. Table 1 summarises observed trends in the drivers over the past century and projected changes for the 21st century, and estimates the level of confidence associated with each projection. Fig. 2 illustrates the key expected spatial and seasonal variability across the UK, according to the UK Climate Change Projections Report (UKCP09; Lowe et al., 2009).

2.1. Temperature

The warming of the global climate system is unequivocal (Jenkins et al., 2009). Data reveals that SST has warmed since the 1970s, at a rate of 0.11 °C per decade, and there is some evidence that SST has warmed since the 1870s (IPCC, 2013). Despite considerable year-to-year and even decade-to-decade variability, Jenkins et al. (2009) have shown that there is a clear UK coastal SST increase of about 0.7 °C over the past three decades, based on several long-term and short-term data sets.

Climate models project global mean SST to increase by 0.6—2 °C by 2100, depending on emission scenarios (low, medium, or high) and taking into account model uncertainty and seasonal variability (IPCC, 2013). Over the UK shelf edge regions and northern North Sea, the UKCP09 report projected temperature increases of 1.5—2.5 °C to occur this century, with larger increases of 2.5—4 °C in the Celtic, Irish and southern North Sea — a trend of 0.3 °C per decade (Lowe et al., 2009), e.g., Fig. 2b and c. However, a potential shutdown of the North Atlantic current (Gulf Stream) would result in much lower winter SST in the UK — but consensus among current climate model projections is that this is considered a low-probability, high-impact event (Wood et al., 2003). There has been a 1—3 °C rise in river water temperatures over UK and Europe during the past 100 years; since river water temperatures closely positively correlate with air temperatures (Whitehead et al., 2009).

2.2. Sea-level rise

Global mean sea level has risen at a rate of 1—2 mm yr^{-1} since 1900 (Church et al., 2004; IPCC, 2013), with an apparent change in rate to 3 mm yr^{-1} during the past 30 years, determined from satellite altimetry measurements (Cazenave and Nerem, 2004; Church and White, 2006) and analysis of tide gauges worldwide (Woodworth and Blackman, 2004; Menendez and Woodworth, 2010). Climate models, assuming different emission scenarios, projected with "medium confidence" that global mean sea levels will increase by 44—74 cm by 2100 (IPCC, 2013); an increase on the 18—59 cm previously projected by IPCC (2007). Alternative estimates based on vast ice sheet melt suggest, with low confidence, that sea level could rise by 1.9 m by 2100 (Jevrejeva et al., 2014), although recent studies suggest this trend is increasingly more

Table 1
Observed trends and future projections of climate change to UK estuaries. The level of confidence (low, medium, or high) has been ascribed, based on the amount of evidence and the level of agreement of the evidence.

Process	Observed trend for the UK	CONFIDENCE and information sources	Projected 21st century change for the UK	CONFIDENCE and information sources
Sea surface temperature	+0.7 °C (1971–2010)	HIGH IPCC (2013) Jenkins et al. (2009)	+1.5 °C to +4 °C	HIGH Lowe et al. (2009)
Sea-level rise	+1 to +3 mm yr^{-1} during last century	HIGH Cazenave and Nerem (2004) Church et al. (2004) Woodworth and Blackman (2004) Church and White (2006) Menendez and Woodworth (2010) IPCC (2013)	+0.44 m to +0.74 m (Steric and eustatic processes) +1.9 m (Ice sheet melt)	HIGH Worth et al. (2006) Lowe et al. (2009) Woodworth et al. (2009) Lewis et al. (2011) IPCC (2013) LOW Jevrejeva et al., 2014 Knight et al., 2015
Storm surge	Pole-ward shift of storm tracks. Increased intensity/decreased frequency extra-tropical cyclones. High variability in storm surges Clustering of extreme sea-levels	MEDIUM McCabe et al. (2001) HIGH Paciorek et al. (2002) Geng and Sugi (2003) Woodworth and Blackman (2004) Woodworth et al. (2009) Menendez and Woodworth (2010) MEDIUM Allan et al. (2009) LOW Wadey et al. (2014).	Centennial changes in extreme water levels are marginally significant (i.e., are of the same order as the natural climatological variability).	LOW Bijl et al. (1999) Alexandersson et al. (2000) Wakelin et al. (2003) Worth et al. (2006) Debernard and Røed (2008) Lowe et al. (2009) Howard et al. (2010) Lewis et al. (2011)
River flows	1969–2008: Increased flow during autumn and winter, decreased flow during spring, no trends during summer Rainfall (river flows) occurring in clustered events	HIGH Keef et al. (2009) Hannaford and Buys (2012) Hannaford (2015) Watts et al. (2015) LOW Jenkins et al. (2008) Burt and Ferranti (2012) Jones et al. (2013) Watts et al. (2015)	Increase in winter mean flow by up to 25% Decrease in summer mean flow by 40–80%	MEDIUM Fowler and Kilsby (2007) Jenkins et al. (2009) Prudhomme and Davies (2009a; 2009b) Fowler and Wilby (2010) Christierson et al. (2012) Prudhomme et al. (2012) Smith et al. (2013) Kay et al. (2013a,b)

likely (Knight et al., 2015).

Sea-level rise is not spatially uniform, with variability due to steric and eustatic processes, as well as vertical land movements such as isostatic adjustment (e.g., Shennan and Horton, 2002). It is this relative mean sea-level which is most important for coastal communities (Lewis et al., 2011). In the UK, observed sea level trends are broadly consistent with the global average (Woodworth et al., 2009). Although, the effects of vertical land movements after the last ice age will augment this trend in the far southwest UK (by 1.2 mm yr^{-1}) and reduce it in parts of Scotland (by +2 mm yr^{-1}) (Shennan and Horton, 2002). Therefore, estuaries in southeast England are most at risk from the combined effects of land sinking and sea levels rising (Davidson et al., 1991), e.g., Fig. 2a. What is clear from recent analysis is that relative mean sea level around the UK is rising, albeit at a spatial varying rate, which could have significant consequences on estuarine systems (e.g., Robins et al., 2014), and shall be considered in Sections 3 and 4.

2.3. Storm surges

Analysis of atmospheric observations, and application of this data to models, show that the second half of the twentieth century experienced a pole-ward shift in the majority of storm tracks across the Northern Hemisphere (McCabe et al., 2001). Further, there was an increase in intensity, but a decrease in frequency, of extra-tropical cyclones (Paciorek et al., 2002; Geng and Sugi, 2003). However, storm surge incidents in the UK were un-related with changes to extra-tropical cyclones: extreme water levels were driven by a rise in mean sea-level, rather than an increase in storminess (Woodworth et al., 2009; Menendez and Woodworth, 2010). Therefore, it is uncertain whether the UK storm surge frequency has changed beyond natural levels of variability (Allan et al., 2009) — yet, some limited evidence is now emerging of clustering of extreme sea-levels (Wadey et al., 2014). For instance, there was an exceptional number of extreme high waters during the 2013/14 winter in the UK (Wadey et al., 2014).

Mean sea-level rise will change the dynamics of storm surge generation and propagation (Mcinnes et al., 2003) and estuarine flood risk and inundation (Senior et al., 2002). Models indicate future centennial changes in extreme water levels, which pose the greatest flood risk to estuaries, will be moderate and of the same order as the natural climatological variability (Worth et al., 2006; Debernard and Røed, 2008; Lewis et al., 2011). The uncertainty within atmospheric models is high due to the parameterization of sub grid-scale processes, such as cloud formations and inter-decadal variability (Howard et al., 2010; Lowe et al., 2009), and quantification of decadal variability of storm surge climates (Alexandersson et al., 2000; Wakelin et al., 2003); therefore, future changes to the storm surge climate in the UK appear to be unresolved at present.

2.4. Rainfall and river flow

River flows represent the integrated response of all
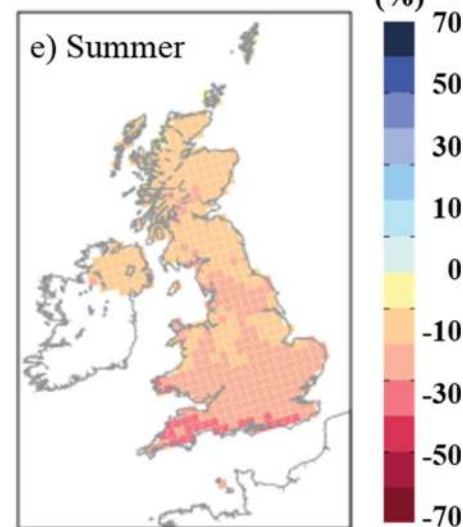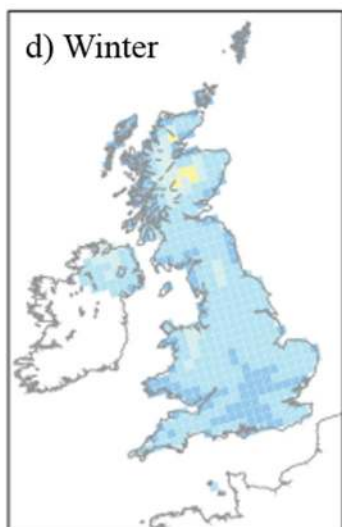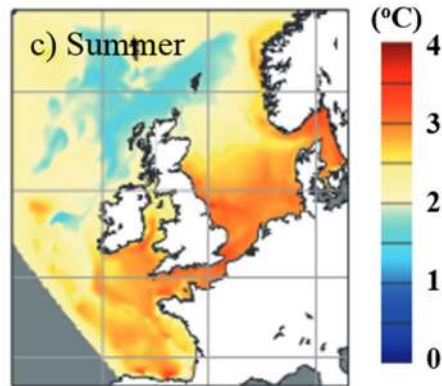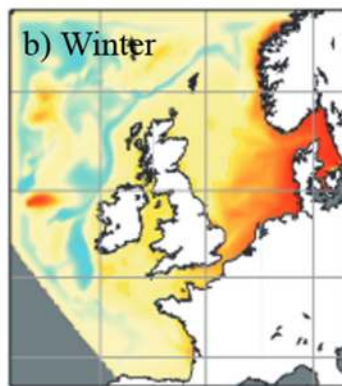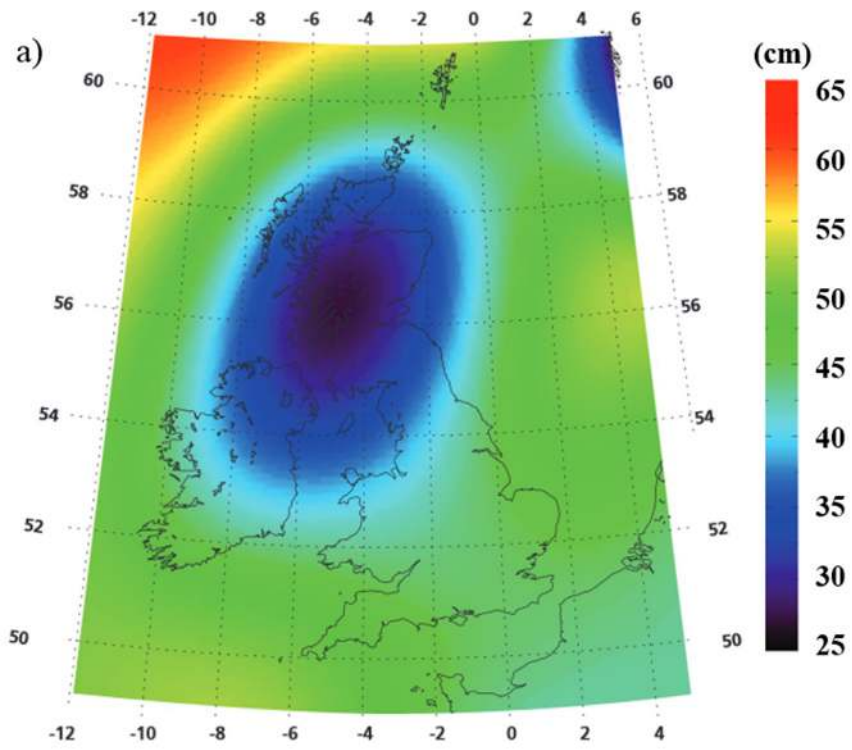
hydrometeorological processes acting upon a catchment, and are therefore a key indicator of potential impacts of climate change to UK estuaries (Hannaford, 2015). Atmospheric warming has been shown by Watts et al. (2015) to intensify the hydrological cycle; although there is a complex, non-linear process leading from increasing temperatures, through changes in precipitation to river flow response (Hannaford, 2015). Past trends show that, while annual average rainfall in the UK is unchanged since the 1960's, winter rainfall has intensified and increasingly occurred in clustered events (Burt and Ferranti, 2012; Jones et al., 2013; Watts et al., 2015). For the same period, river-flows have increased in winter also, especially in upland western areas; autumn flows rose and spring flows decreased slightly, while there was no consistent change in the summer (Hannaford and Buys, 2012; Hannaford, 2015; Watts et al., 2015). For past extreme events, data from extensive UK river-gauge networks show that the relationship between precipitation and river flow is spatially variable and dependent on catchment characteristics (Keef et al., 2009).

Many studies have investigated potential changes to future UK river flow rates (e.g., Kay et al., 2009; Prudhomme and Davies, 2009a; 2009b), including impacts to groundwater (e.g., Jackson et al., 2011), and more recent studies have analysed the latest UKCP09 future river flow data series (1 km-resolution and catchment-bias-corrected; Prudhomme et al., 2012). The scientific consensus on projected future UK precipitation patterns is the "wetter winters and drier summers" signal (Arnell, 2003; Christierson et al., 2012), with some spatial variability in the UK to this pattern. Indeed, the UKCP09 Report explains that little change is expected in the median precipitation amount by 2100, but significant changes in the trends of winter and summer precipitation are expected (Jenkins et al., 2009) (Fig. 2d and e). The report generally found a slight increase in the winter mean precipitation and a decrease in summer mean precipitation for the western UK (Jenkins et al., 2009). River catchment models indicate river flows are projected to reduce in summer (by 40–80%) and increase in winter (by up to 25%), again particularly in mountainous regions of western UK (Fowler and Wilby, 2010; Christierson et al., 2012; Prudhomme et al., 2012). These projections were biased through inability to capture monthly precipitation climates (Prudhomme and Davies, 2009a; 2009b; Smith et al., 2013). Furthermore, when future precipitation patterns are projected through to future river flow rates there is considerable uncertainty in GCM projections (Prudhomme and Davies, 2009a), as well as downscaling and catchment model uncertainties (Kay et al., 2009; Prudhomme and Davies, 2009b).

3. Climate impacts on the physical estuarine environment

3.1. Flooding and inundation

Increased estuarine flood risk and inundation occurrence are caused primarily by a combination of increased sea-level (sea-level rise, storm surge and changes to the wave climate) and changes to incidences of high river flow events (Lewis et al., 2013; Quinn et al., 2014). Estuarine flooding can also be the volumetric combination of storm surge and extreme river flow that can drastically increase flood hazard (e.g., Maskell et al., 2013). Further, there is evidence for a positive relation in some UK estuaries between storm surge and peak river flows (Samuels and Burt, 2002; Senior et al., 2002; Svensson and Jones, 2002), suggesting that surges should to be taken into account for flood risk estimation (Keef et al., 2009; Zheng et al., 2013). Both sea-level and river flow climates in the UK appear to be changing due to climate change, potentially resulting in increased flood risk to UK estuaries (King, 2004). Potential negative impacts of estuarine flooding to ecosystem goods and services, in the absence of mitigation, include: damage to infrastructure and population displacement (Hanson et al., 2011; Nicholls et al., 2011), increased erosion and loss of land to the sea (Zhang et al., 2004), pollution and increased human health risk, and significant loss of wetland habitats and ecosystems (Day et al., 1995; Nicholls et al., 1999).

Modelling approaches are important for predicting flood risk, assessing impact, and mitigation decision-making — where risk is generally defined as the probability of the occurrence multiplied by the consequences. It is important to simulate the nonlinear coupling of surge-wave-river events, since static (sea-level-only) events will underestimate the cost significantly (Prime et al., 2015). Using published climate change scenarios, a UK panel (King, 2004) found that a combination of sea-level rise and increased storminess will allow storm surges to propagate much further inland, so that UK estuaries will be subject to both higher water levels and more energetic wave attack. However, Worth et al. (2006) simulated no significant increased storm surge on the east coast of the UK over the next century. Although the likelihood of extreme sea-level rise (up to 1.9 m over the 21st century) due to rapid ice sheet melt is apparently low (Jevrejeva et al., 2014), the consequence would be an increase to the projected risk; for the case of the UK's largest estuary — the Severn, the projected flood risk has been reported by Quinn et al. (2013) to increase by 30%.

A key consideration for estuarine flood risk and inundation occurrence, under accelerating rates of sea-level rise is the potential for geomorphological features and flood depths to adjust, through sediment transport (i.e., the sediment flux into or out of an estuary), estuarine morphology (erosion and accretion), and sediment compaction (Dickson et al., 2007; Shennan et al., 2012). Sediment infilling will cause flood risk to increase incrementally over long (decadal) timescales, in which case mitigation measures can be foreseen and employed in advance to reduce risk. However, based on analysis of historical data, Pye and Blott (2006) showed that morphological changes on the Suffolk coast (around Blyth, Ore-Alde-Butley, and Deben Estuaries) over the past 50 years have actually been lower than historically. Instead, rapid morphological changes (and potential increased flood risk) will result from extreme surge/wave/river events (Horrillo-Caraballo et al., 2013). Interestingly, Dissanayake et al. (2015) explain that the initial storm in a cluster of storms has most morphological impact on the coastline. In addition to these natural trends, shoreline protection has greatly reduced the input of new sediment sources, resulting in much contemporary coastal erosion (Nicholls et al., 2013). The capacity for estuaries to respond to sea-level rise will depend on landward constraints to transgression, available accommodation space and available sediment supply (marine and fluvial) (Nicholls et al., 2013). Rossington et al. (2007) predicted the maximum rate of sea-level rise that UK estuaries can experience without losing all intertidal areas. Their results suggest that some estuaries are more vulnerable to sea-level rise (e.g., Southampton Water) and others

Fig. 2. A selection of UKCP09 projected changes to UK climate drivers over the 21st century, showing spatial and seasonal variability (all figures published by Lowe et al. (2009)). (a) Shows relative sea-level change (cm) projections by 2100 (combined land movements and absolute sea-level change estimates from the central simulation (out of 11 ensemble members) for the IPCC medium emissions scenario). The middle panels show simulated seasonal mean trends in SST (°C), from (1961–1990) to (2070–2098), for (b) winter and (c) summer. Bottom panels show projected changes in mean precipitation (%) during (d) winter and (e) summer, at the 50% probability levels, for the 2080s under the IPCC medium emissions scenario.

are more resilient (e.g., the Ribble), depending on estuary area, sediment supply, sediment transport potential and human interference. Some studies of intertidal and saltmarsh areas suggest low resilience to sea-level rise (e.g., Orford and Pethick, 2006; French, 2006; Marani et al., 2007).

Projected increases in UK average temperatures will increase evapotranspiration throughout many UK catchments, which may alter vegetation types and reduce soil moisture — all of which could reduce runoff into estuaries and potentially dampen flood risk slightly (Boorman, 2003). As an example, modelled catchment runoff from IPCC climate change scenarios in the Ouse-Humber Estuary showed increased evapotranspiration (5% in winter, 15% in summer), although total runoff generally increased by up to 30% in winter due to projected increases in rainfall (Boorman, 2003).

From our review, it is clear that uncertainty is high in predicting future changes to physical drivers in UK estuaries, due to the spatial variability associated with projected changes to rainfall patterns. Furthermore, considering GCM model uncertainty, downscaling uncertainties, hydrological model uncertainties and land use changes, as well as variability of the climate system (e.g., Kay et al., 2009; Prudhomme and Davies, 2009a, 2009b; Christierson et al., 2012), prediction of future estuary changes requires further research to determine confidence in future river flow rates. Furthermore, surge–rainfall association requires further investigation (Zheng et al., 2014) and their implications to biology, water quality and sediment transport processes (Lowe et al., 2009; Robins et al., 2014).

3.2. Hydrodynamics and mixing

In the UK, the majority of the main estuaries are relatively shallow, and are classified as either mesotidal, macrotidal or hypertidal, in terms of tidal forcing (Davidson et al., 1991; Prandle, 2009). The strong tidal forcing produces a vertically well-mixed water column, though weak vertical stratification is sometimes evident, e.g., approaching slack high water in the Conwy (Howlett et al., 2015), or more commonly during the ebb tide depending on the river flow (Pye and Blott, 2014). Typically, the tide becomes asymmetric upon entering the estuary due to shallow water frictional effects — resulting in a shorter, stronger flooding phase and a longer, weaker ebbing phase (Dronkers, 1986; Moore et al., 2009). Influences of waves on hydrodynamics and mixing are usually small within the estuary due to wave dissipation over the UK shelf, particularly over offshore sand banks (Carini et al., 2015), and prevented from penetrating into bar-built estuaries because of the constricted entrance. Of high importance for hydrodynamics and mixing are variabilities in river flows — particularly due to the nature and frequency of extreme flow events (Struyf et al., 2004). Rivers in the UK tend to rise very quickly after rainfall, especially in the north and west, but flowrates reduce more slowly; as such, UK estuaries are often in a state of non-equilibrium in terms of stratification and salt balance, and nutrient/material fluxes; processes which are critical for ecosystem goods and services (Prandle, 2009).

Consequently, projected alterations to sea-levels and river flows due to climate change could have significant implications for estuarine functioning (Whitehead et al., 2009). Modelling in the Conwy by Robins et al. (2014) shows interaction of sea-level rise with maximal projected changes in river flow over the 21st century will potentially increase the saline intrusion length. Sea-level rise combined with long periods of low flow conditions (drought) simulated the saline intrusion and turbidity maximum to be pushed further up-estuary from today's limit and retaining river-borne substances in the estuary for extended periods. On the other hand, increased river flows simulated the saline intrusion to be pushed down-estuary, and potentially flush nutrients and

pollutants offshore (Robins et al., 2014). These findings relate to similar studies in the US by Yang et al. (2015). The extent of the saline pumping effect is dependent on the strength and frequency of river flow events, which are projected to be more variable in the UK with climate change. Correspondingly, Prandle and Lane (2015) investigated 96 UK estuaries and concluded that a sea-level rise of 1 m will increase the saline intrusion length by >7% in deep estuaries and by >25% in estuaries shallower than 10 m. Further, a change in river flow of 25% (either increase or decrease) will have significant effects on both vertical mixing and salinity intrusion. Accordingly, estuarine recovery after rainfall events restores a steady state salt distribution (MacCready, 1998; Hetland and Geyer, 2004). However, few observations of estuarine tidal-pumping/recovery have been documented in the UK (e.g., Simpson et al., 2001) and there is generally a lack of knowledge about how estuarine hydrodynamics react to extreme climatic events for UK case studies.

Estuarine hydrodynamics in the majority of UK systems is further complicated by the presence of estuarine fronts — small-scale regions which exhibit significant changes in hydrographic variables (O'Donnell, 1993). Estuarine fronts have significant roles in estuarine circulation, and affect sediment grain size distribution (Neill, 2009) and larval dispersal (Robins et al., 2012). Since estuarine fronts are driven by horizontal salinity gradients (controlled by river flow and tidal range), it is anticipated that the role of fronts in estuarine processes will become increasingly important within the context of 21st century climate change, when extremes in river flow, coupled with increased estuarine tidal range due to sea-level rise, are likely.

3.3. Sediment transport and morphology

The development of UK estuaries has typically resulted from the latest post-glacial rise in sea level (of the order hundreds of metres), which inundated the UK coastline and drowned river mouths (Dyer, 1997). However, Tessier et al. (2012) have demonstrated that the majority of morphological development across northwest European estuaries has occurred following centennial-scale periods of major climate change (enhanced storminess), in addition to the (lesser) continual effect of relative sea-level rise, given an available sediment supply. In the present climate, the morphological development of such systems, other than impacts caused by human activities, has been determined by the changing relative importance of the river, wave and tidal influences (Prandle, 2009), with tidal flow asymmetry being considered as the main factor influencing net sediment transport and hence being responsible for the stability of the system (Hoitink et al., 2003). Both the magnitude and duration of the flood and ebb tides affect the morphological outcome and feedback effects can keep an estuary in a state of dynamic equilibrium, oscillating between morphological states (Pethick, 1994), subject to the influence of sea-level rise. According to Lanzoni and Seminara (1998) and more recently Pye and Blott (2014), deep or bar-built estuaries tend to generate ebb-current dominance, whereas in shallow or wide-mouthed estuaries, flood-current dominance occurs, which will affect sediment transport pathways. But classification of flood/ebb flow asymmetry (e.g., Friedrichs and Aubrey, 1988) does not necessarily give an accurate indication of the net sediment transport.

State-of-the-art, process-based, sediment transport models are believed to have the correct general behaviour over the wide range of wave-current conditions encountered on site. However, the morphological models in which these processes are embedded are capable of predicting only the statistics defining channel geometry and meandering on time scales of decades or longer. The seabed is inherently unstable and morphological modelling outcomes are

capable of reconstructing only the distribution of channels and sand flats that characterise many shallow tidally-dominated estuaries. Small bathymetric irregularities introduced at the start of a numerical simulation evolve into larger scale bed features. McCann et al. (2011) demonstrated how a tidal channel/sand flat network appears spontaneously and exhibits 'self-organisation' linked to the formation of ebb-dominant channels and flood-dominant shoals. In McCann's simulations sediment was exported seaward in the deeper channels, while net transport was shown to be landward (i.e., up-estuary) on the tidal flats.

Robins and Davies (2010) attempted to systematise this general behaviour, classifying estuary regions as 'flood-dominant' if the asymmetry in the tide causes a net, up-estuary, sediment accumulation (because the stronger flood flow may be above the threshold of sediment motion for longer than the ebb flow). They referred to the opposite behaviour as 'ebb-dominant'. They demonstrated that an estuary is likely to become more flood-dominant as the tidal wave travels landward, while providing clear evidence of the role of ebb channels in promoting ebb dominance in net sediment transport. Their modelling suggested that, for idealised estuary shapes characterising mesotidal systems found along the UK coast, net flood-dominance in sediment transport occurs in regions of shallower water (e.g., tidal flats and salt marshes in the upper estuary) where $a/h > 1.2$ (a = local tidal amplitude, h = local, cross-sectionally averaged, water depth), while net ebb-dominance occurs in regions of deeper water (Robins and Davies, 2010). However, the magnitude of net ebb-dominance was less pronounced than the net flood-dominance. Sea-level rise caused the threshold value of a/h to be reduced, with the estuary shifting from ebb-dominance towards flood-dominance, with less transport overall. Reducing the sediment grain size increased both the gross and net sediment transport but did not significantly affect the transport pattern. Importantly, the inclusion of an ebb channel promoted ebb-dominance in the channel. Applying these outcomes to a realistic case (the Dyfi Estuary), Robins and Davies (2010) concluded that, due to a projected sea-level rise of 1 m in the next 100 years, greater flood-dominance (i.e., net sediment accumulation into the estuary) but less sediment transport overall would occur. In a similar vein, Pye and Blott (2014) postulated generally that sediment accumulation would continue until the creation of a central channel and high flats, which would switch the system from flood-to ebb-dominance. Pye and Blott (2014) also stated that the effects of extreme river events and storm surges on the long-term sediment estuarine budget remain unclear and require further research.

The mobilisation of historic, consolidated, heavy metal contaminants in semi-industrialised UK estuaries (e.g., from mudflats and saltmarshes in the Humber and Mersey) poses a serious risk to water quality and human health (Lee and Cundy, 2001). On the other hand, a potential positive outcome is that this re-mobilisation, which is often a result of extreme stormy weather, accelerates sediment exchange with the open sea (e.g., the exchange of mercury-rich fine sediments from the Mersey Estuary into Liverpool Bay (Harland et al., 2000)).

3.4. Fluxes of nutrients

Biogeochemical cycles vary over both time and space. There is a requirement in the UK to further understand such variability and nutrient coupling and processing in different catchment types to aid development of mitigation actions under climate change. Temporal changes include diurnal cycling and event based changes between high and low flow conditions, seasonal, decadal and longer. Spatially, estuarine biogeochemical cycles form a continuum from freshwater to marine, transporting and processing many forms of macronutrients with discontinuities in the retention and storage of macronutrients (Statham, 2012). Estuaries, therefore, play a crucial role in biogeochemical cycling of nutrients, which in turn influence the growth and resilience of plants and animals (Nedwell et al., 1999; Malham et al., 2014).

The impact of nutrient loading in UK estuaries depends on the physical and geological setting (Jickells et al., 2000) and the physical and chemical conditions prevalent at any particular time. Importantly, projections for the UK suggest that winter river flows will increase by between 4% and 9% by 2020, with summer decrease of roughly 11% (1—32% range), based on IPCC climate change projections (Whitehead et al., 2009). Nutrient projections suggest a possible rise of macronutrient concentrations (such as nitrogen and phosphorus) under climate change conditions, with consequent impacts of augmented eutrophication, hypoxia and harmful algal blooms (Tappin, 2002; Statham, 2012). Projected 21st century sea-level rise, combined with reduced river flows (projected during summer in the UK), could heighten the likelihood of estuarine nutrient trapping, in between river flushing events (Robins et al., 2014). Conversely, nutrient retention will be less likely during winter if projected increases in river flow frequency occur, which could counteract the effects of sea-level rise to some extent (Struyf et al., 2004). Nutrification in summer is more prone to cause eutrophication than in winter: only summer production is nutrient limited; winter production is light limited. Thus, the dry summer effect has the greatest potential for augmenting eutrophication.

However, current understanding, let alone prediction, of biogeochemical processes in estuaries is highly complex and requires further study (Jickells et al., 2000; Andrews et al., 2006; Najjar et al., 2010; Statham, 2012). Overall, more is known about fluxes of inorganic than organic nutrients (Jarvie et al., 2012). Further research is needed on bacterial cycling of organic nutrients (Asmala et al., 2014), nutrient reactivity (Valdemarsen et al., 2014), and on nutrient speciation and interaction with fresh and salt water under different pH and dissolved organic matter conditions (Morgan et al., 2012). Catchment-estuary model coupling methods are improving (Uncles, 2003; Huang et al., 2013), and there is currently a UK drive in data generation (e.g., NERC Macronutrients Cycles Programme), which will prove valuable for model validation.

4. Climate impacts on estuarine ecosystems

4.1. Human health: pollutants and pathogenic microorganisms

By 2100, an increase in average temperatures, together with an increase in winter rainfall and longer drought periods will ultimately have impacts upon public health risk to UK estuaries, associated with water-borne and food-borne diseases caused by microbial pathogens (McMichael et al., 2006). There are over 100 recognised types of microbial pathogen found in contaminated water used for recreation and as potable water (Rose et al., 2001), with the potential for food-borne transmission via seafood (Marques et al., 2010). Projected increased precipitation and flooding may result in untreated sewage discharge via combined sewer overflows (CSOs) and increased agricultural runoff (Perciasepe, 1998). In addition, toxic pollutants, ammonia, organic solids and oxygen-demanding substances are also discharged into the water system (Friedman-Huffman and Rose, 1999; Stachel et al., 2004) and may increase concurrently with high precipitation events. There are precedents for precipitation events preceding outbreaks of water-borne disease (Curriero et al., 2001; Rose, 2005); for example, outbreaks of the protozoan pathogen *Cryptosporidium* have been observed following extreme precipitation events, which reduces the efficiency of water treatment processes resulting in increased human health risk (Rose et al., 2001). In

addition, faecal indicator organism concentrations have been found to be higher during a winter of high precipitation, when compared with the rest of that year (Lipp et al., 2001), and high precipitation also results in an increase in the viral load of estuarine environments, particularly after CSOs, where a significant increase in Adenovirus and Norovirus concentrations have been observed in receiving waters (Rodríguez et al., 2012). It is well established that increased rainfall may influence the concentrations of toxic metals, organic chemicals, algal toxins, and human pathogen contaminants in seafood (Marques et al., 2010). Conversely, drought conditions will result in decreased river flow and lower concentrations of microbial contaminants (Rose et al., 2001). Sea-level rise may lead to the loss of coastal wetlands, which filter nutrients, microbial agents, and chemical agents (McMichael et al., 2006). However, the implications of sea-level rise on the fate of microbial, nutrient and chemical agents and their interactions with the wider ecological dynamics of the system are poorly understood, and could be addressed by integrated catchment-river-estuary models (Rose et al., 2001).

The cholera model perhaps represents the best-studied mechanism for understanding the role of climate on infectious disease transmission (Lipp et al., 2002). *Vibrio cholerae* (the bacterial species responsible for cholera) and other *Vibrio* spp. are endemic in marine and coastal environments and demonstrate strong associations with warmer sea surface temperatures that drive their seasonality and ecological distribution (Rose et al., 2001). Furthermore, favourable environmental conditions (dependent on nutrients and temperature) promote the formation of algal blooms that act as a reservoir for *Vibrio* spp. and can cause reversion from a quiescent to infectious state (Rose et al., 2001). Future temperature shifts, therefore, have the potential to expand the prevalence of *V. cholera* at both temporal and geographical scales (Lipp et al., 2002). However, cholera outbreaks have disappeared from much of the developed world (including the UK), and the endemicity of cholera in the tropics and subtropics exemplifies the favourable environmental conditions that drive their ecological distribution. In contrast to cholera, which is autochthonous in estuarine systems and favours dry weather and warmer temperatures, the prevalence of other bacterial and viral pathogens of allochthonous origin are governed by different conditions. For example, faecal indicator organisms demonstrate enhanced survival in estuarine environments at lower temperatures and their abundance has a strong correlation with high precipitation events (Lipp et al., 2001; Malham et al., 2014). Further research is required to understand the projected effects of future climate scenarios on estuarine pathogen load and subsequent human health risk.

4.2. Estuarine habitats

There is little evidence-based research on the effects of climate change drivers on estuarine habitats, except for the effects of temperature and sea-level rise (Mieszkowska et al., 2013; Fagherazzi et al., 2012). Here, we highlight current knowledge for the most researched habitats, the majority of which are intertidal.

Many UK salt marshes were lost to agricultural expansion and coastal development in the 18th and 19th centuries (Doody, 2008). The total area of 40,522 ha in England and Wales was still declining, by <100 ha y^{-1} at the last survey (1989–2009; EA, 2011), mainly due to losses in the southeast associated with isostatic adjustment. Other geographical areas had marsh expansion over the same period (EA, 2011) and it is not known whether these patterns of change are associated with emergent climate change. In principle, climate change might be both 'good' and 'bad' for salt marshes. A moderate rise in the annual average temperature can boost vegetation productivity and favour marsh expansion, although a greater frequency of dry summer spells could depress plant growth through desiccation and evaporation-driven increases in sediment salinity (Gedan and Bertness, 2010). Increased frequency of high rainfall events is likely to boost riverine sediments supply to marshes, which is a key stimulant of marsh vertical growth (Fagherazzi et al., 2013). Conversely, increased storminess could accelerate erosion of marshes located in wide estuaries, where the fetch, and thus potential wave energy, is higher. Evidence suggest marshes are well-capable of keeping pace with sea-level rise if given the space to transgress inland, and if exceptional land subduction and depressed sediment supplies are absent (Fagherazzi et al., 2012). Coastal squeeze remains the greatest threat to salt marshes in terms of sea-level rise, the elimination of which is linked to policy decisions, such as shoreline management planning. Marsh vertical and horizontal growth is strongly dependent on sediment supply, which, if diminished, can switch marshes from accreting to eroding and preclude their keeping pace with sea-level rise (Bouma et al., 2014). Section 2 showed sediment supply is governed by hydrodynamics, determining for instance the up-estuary transport of marine sediments, as well as the down-estuary transport of catchment and riverine particles. Tendencies of increased fluvial transport of sediments bode well for the capacity of marshes to keep pace with sea-level rise. Emergent evidence for the past 70 years show that many marshes on the west coast of England and Wales have undergone dynamic changes in areal extent, but the collective area has expanded, particularly since the 1960s (Martin Skov, *pers. comm.*). These changes are not linked to isostatic adjustment, which is commonly invoked as a cause for national variation in marsh erosion rates (EA, 2011). Future projections of changes to estuarine function might do well to carefully consider the causes for such regional differences in ecosystem change, and the implications to estuarine management.

Rocky shores in the UK are predominantly located on open coasts, although they do occur in some estuaries. The following findings are mainly from coastal shores, but many are likely to be transferable to estuarine settings. The area cover of intertidal rocky shores is likely to decline with sea-level rise: as many shorelines increase in steepness with elevation, the distance between the low and high tidal mark, and thus the intertidal area, will decrease with sea-level rise (Jackson and McIlvenny, 2011). Temperature rise is already having some effects. Throughout the UK, a number of rocky shore species have been gradually moving north in response to rise in temperature, punctuated by cool years in which the advance has ceased, or reversed (Mieszkowska et al., 2013). The tendency is for species that are intolerant to warming to retreat northwards to remain in cooler conditions, while Mediterranean/Lusitanian species are coming up from the south, facilitated by an increasingly benign climate (Mieszkowska et al., 2013). Similar changes are occurring in other estuarine habitats, such as *Sabellaria* reefs (Frost et al., 2004), as well as in terrestrial and marine systems (Hiddink and Hofstede, 2008). Changes to community composition brings with it the potential for change in biodiversity and ecological functioning (organic production, degradation, habitat provisioning), as altered composition and biodiversity changes the interactions between species and their ecological roles (Balvanera et al., 2006). Rocky shore habitats are competed for by algae and grazing molluscs, and the relative dominance of these groups is determined by environmental conditions: increased wave exposure and higher temperatures favour animals (Hawkins et al., 2008). Thus, predictions are that algae will become less prevalent in a warming and more hydrologically energetic climate, which could diminish overall biological production of estuarine rocky shores (Hawkins et al., 2009).

Seagrass beds in Europe have dramatically declined over the past century, by 50–80% (Short and Wyllie-Echeverria, 1996). In the

UK, there is no research-based evidence on how the area cover of seagrass beds will respond to climate change (Mieszkowska et al., 2013), although they are considered to be vulnerable to a plethora of changes associated with climate change (Jones et al., 2011). Current patterns of change are variable: there are indications of areal increases in Scotland and Wales (Smith et al., 2002; Howsen, 2009), but retreat in the Isle of Wight (Mieszkowska et al., 2013). Seagrasses elsewhere have suffered from climate-related changes in temperature, storminess and prevailing winds (Bjork et al., 2008). Mediterranean seagrass beds are predicted to decline in response to temperature stress (Marba and Duarte, 2010). In the UK, forecasted temperatures are well within the tolerance levels for *Zostera noltii* (37 °C, Massa et al., 2009); however, *Zostera marina*, the other UK species, stops growing at >20 °C and starts dying at > 25 °C (Greve et al., 2003), and dieback during summer heat waves have been observed as far north as Denmark (Greve et al., 2003; Winters et al., 2011). It is not unreasonable to expect similar patterns in the UK (Mieszkowska et al., 2013).

Sedimentary habitats account for 90–95% of estuarine intertidal habitats and are powerhouse production areas of invertebrates that feed higher trophic levels, including birds (McLusky and Elliott, 2004). Sediment habitats are already undergoing geomorphological change from climatic drivers including increasing wave height, surges and sea-level rise (Lowe et al., 2009); they are likely to respond to changes in river flow, although there are no detailed studies to support this prediction in the UK. A projected more energetic wave climate is likely to steepen benthic profiles (Mieszkowska et al., 2013) and diminish the abundance and diversity of sediment living animals through coarsening of sediments (Pethick and Crooks, 2000; Fujii and Raffaelli, 2008; Yates et al., 2010; Fujii, 2012). Mudflats are thought to be particularly sensitive to climate change, through an assortment of drivers, including shifts in sediment supply and sediment coarsening (Gubbay and Earll, 2010; Jones et al., 2011), and the risk of coastal squeeze as enhanced wave and tidal energy require mudflats to move up-estuary to less energetic areas (Pethick and Crooks, 2000). Modelling indicates an upstream shift in the position of the most productive intertidal flats of the Humber estuary could reduce estuarine production by 23%, because the area for flats is reduced by the narrowing of the estuary (Fujii and Raffaelli, 2008; Fujii, 2012). Accurate predictions are difficult to make given the influences of factors such as inter-estuary variation in geomorphology, which regulates hydrological forcing and sediment supply, spatial variation in catchment processes, which regulate sediment supply and water chemistry, and the interactions of biological and physical variables on benthic organisms (Fujii, 2012). For instance, a decline in productivity with sediment coarsening might be countered by increased temperature and milder winters that boost benthic production, (Fujii, 2012) and/or by upper reaches of estuaries becoming more saline and accommodating of invertebrate diversity and biomass (Ysebaert et al., 2003).

The honeycomb worm *Sabellaria* sp. makes bulky biogenic reefs in intertidal and subtidal rocky areas. Reefs are colonised by an array of invertebrates and are a priority habitat for protection in the UK. *Sabellaria* is a 'warm-water' genus and the reefs have been extending northwards in the UK in recent decades, as a response to less severe winter temperatures (Frost et al., 2004); a pattern which is likely to continue into the future. Temperature rise is unlikely to markedly affect the distribution of UK mussel beds, as predicted conditions will well within their temperature tolerance; however, mussel condition might well be affected by eutrophication (Mieszkowska et al., 2013).

The potential impact of climate change on estuarine habitats is somewhat determined by their physical nature (biogenic, non-biogenic, hard, soft) and position (intertidal, subtidal, location in estuary). The persistence of biogenic habitats, such as mussel beds, is governed by the physiological tolerances of the habitat-forming species. Non-biogenic systems can continue to exist despite climate change, although some might become biologically unavailable. Thus, rocky shores will persist as a habitat, but are projected to decrease in area due to coastal squeeze (Jackson and McIlvenny, 2011). Intertidal habitats are naturally more influenced by short-term changes in weather than sub-tidal habitats. Ultimately, changes in community composition induced by climate change might affect the resilience of systems to other sources of environmental change, such as eutrophication and coastal development, although the effects of multi-stressor impacts on UK estuaries is limited. Major research gaps also exist on how biological feedbacks that affect physical conditions in estuaries will be influenced by sea-level rise. For instance, saltmarsh colonisation alters the forcing and distribution of hydrological energy, which facilitates further expansion of marshes and ultimately influences the tidal prism and sediment distribution. Whether or not such biological feedback mechanisms will be enhanced or reduced by change in climatic conditions is currently not known.

4.3. Larvae in estuaries

Environmental cues synchronize the reproductive cycle of many marine invertebrates and fish (Kingsford et al., 2002). A broad diversity of taxa have the capacity to detect variations in current direction, magnetism, water pressure, water chemistry (e.g., amino acids, salinity), sound and vibration (e.g., waves breaking or fish assemblages), and light gradients (Forward et al., 2001). Some organisms can detect multiple cues (e.g., decapods and fishes) and are likely to have integrated sensory responses (Kingsford et al., 2002). The type of response may be endogenous and regulated by an internal clock (e.g., tidal stream transport), or a direct response to the environment (e.g., diel transport) (Queiroga and Blanton, 2005). Further, the phasing and extent of vertical migration both change throughout ontogeny (Queiroga and Blanton, 2005). These cyclic cues may affect a number of reproductive parameters, including sex determination, gametogenesis, spawning, transport, settlement, retention, connectivity, and metamorphosis (Lawrence and Soame, 2004).

For estuarine species, primary environmental cues are tidal currents and photoperiod. European shore crab larvae in estuaries within the North Sea use selective tidal stream transport to swim to the surface during the flood phase of the tide, which aids in-shore dispersal towards estuarine feeding grounds (Moksnes et al., 2014). Conversely, decapod crustacean larvae may swim to the surface during the ebb phase of the tide and migrate offshore (Queiroga and Blanton, 2005). In other regions, shore crab display nocturnal (diel) vertical migration to avoid predation (Moksnes et al., 2014). Estuarine species live in a highly dynamic environment where recruitment variability is a key determinant of population dynamics. Environmental requirements for successful recruitment may differ between co-occurring species, and therefore species may be advantaged or disadvantaged under climate change (Allen et al., 2008; Jenkins et al., 2015). Climate change impacts on UK estuaries are likely to alter the environmental cues, which in turn will affect their reproductive parameters. For example, the phase relationship between temperature of the water column and photoperiod directly controls the metabolism and lifecycles of aquatic organisms, with most biological processes operating faster at higher temperatures, leading to shorter pelagic larval durations and time to metamorphosis (Whitehead et al., 2009; González-Ortegón and Giménez, 2014).

Variability in salinity has been documented to both increase and

decrease time to metamorphosis, depending on species (Forward et al., 2001). The estuarine copepod *Eurytemora affinis* (which exhibits a combination of tidal, salinity, and temperature cues) will change its migration in response to the location of the estuarine salinity maximum (Hough and Naylor, 1991). Indeed, evidence from Chaalali et al. (2013) shows climate-driven up-estuary migration of the species in the Gironde Estuary, France. The presence of aquatic vegetation and increased humic acids (a decomposition product of terrestrial plant material; Fox, 1981), both accelerate the time to metamorphosis of brachyuran crabs (Forward et al., 2001). Alternatively, cues that delay metamorphosis include hypoxia and chemical odours from potential predators (Forward et al., 2001). Thus, species that cue reproduction based on these signals are likely to be particularly vulnerable in the future (Lawrence and Soame, 2004). Consequently, climate change may cause local extirpations of populations in the extreme, or more likely impact fecundity, spawning success, and recruitment significantly.

However, change will ultimately depend on the relative speed of adaptation to climate change (Allen et al., 2008) and the degree of mixing between populations across the Metapopulation (Lawrence and Soame, 2004). Some larvae of estuarine species appear to be resilient to environmental variability and may be able to tolerate changes related with climate change. For instance, an important species, inhabiting brackish waters in the UK is *Palaemon varians*; this is a shrimp found in saltmarshes, coastal ponds and lagoons. Adults of *P. varians* can potentially extend the upper thermal range through acclimation (Ravaux et al., 2012). Larvae of this species appear to be exported and develop in estuarine-coastal areas (Fincham, 1979). Larvae show a wide range of salinity tolerance (5–42: Antonopoulou and Emson, 1988). In addition, larvae appear to tolerate high temperatures as well as food limitation. Metabolic rates increase with temperature (range: 5–25 °C), but total energy losses, calculated from oxygen consumption are lower than at low temperature. In this shrimp, energy reserves at the time of hatching appear to confer larvae a high capacity to tolerate food limitation (Oliphant and Thatje, 2014). Oliphant and Thatje (2013) and Oliphant et al. (2014) showed that there is sufficient variability among broods in per offspring investment to produce variability in development and starvation tolerance; if such variability is heritable, it may confer this species a sufficient capacity to adapt to variable conditions of food, temperature, and salinity.

Besides the hypothesised effect of environmental factors as cues, such factors operate as stressors, in particular in the early life history stages of marine invertebrates. Meteorological forcing, such as extreme rainfall events, promote salinity decreases in estuaries. Thus, low salinities can reduce growth rates especially during the larval stages, if these are weak osmoregulators or osmoconformer (e.g., decapod crustaceans: Torres et al., 2011). Low salinity, experienced by parents or at the egg/embryo stage, can have knock-on effects along the life cycle of estuarine organisms (Giménez, 2006; Jensen et al., 2014; Chaparro et al., 2014). For instance, low salinity at the embryonic stage can affect the capacity of larvae to osmoregulate (Charmantier et al., 2002). Exposure of larvae to low salinity also affects juvenile size and growth rate (Giménez et al., 2004; Rey et al., 2015).

It is likely that multiple stressors will operate on estuarine organisms in a synergistic way (Dolbeth et al., 2011; Przeslawski et al., 2015). As a consequence of warming, temperature may modify the sensitivity to other environmental stressors, but also thermal sensitivity should be modulated by exposure to additional stressors (Sokolova and Portner, 2007). For instance, under low salinity, larvae may be more vulnerable to pollutants (González-Ortegón et al., 2013) or food limitation (González-Ortegón and Giménez, 2014). On the other hand, increased temperatures may increase the tolerance of low salinities (González-Ortegón and Giménez,

2014). Overall, an assessment of the impact of changes in estuaries will require an understanding of the correlated changes in environmental variables such as salinity and temperature.

4.4. Biota

Species living in estuaries experience a naturally highly variable physical environment that is often close to their physiological tolerances (Attrill and Rundle, 2002). Despite this, projected climate changes may have significant impacts on estuarine species and their ecosystems. Moreover, invasive species to estuaries have increased during the last 20 years, due to global temperature warming (Austin et al., 2010), leading to some dramatic effects on native assemblages (Thompson et al., 2002). Problems associated with invasive species, especially pathogens, are projected to increase over the next few decades (e.g., McDowell et al., 2014).

Projected sea-level rise and increased winter river discharge for the UK are thought to be the most important impacts of climate change on species, principally through associated habitat changes (Kimmerer, 2002; IPCC, 2014). Habitat loss from sea-level rise, such as coastal squeeze in the absence of managed retreat, is likely to have important implications for estuarine food webs (Fujii, 2007; Fujii and Raffaelli, 2008; Fujii, 2012). Although only from an individual study site in the Humber estuary, Fujii (2007) demonstrates that a 'squeezed' location can support fewer species and biomass of keystone benthic macroinvertebrates per unit area due to the narrower, steeper and less muddy beach profile, when compared with adjacent undefended shorelines. Yamanaka et al. (2013) have supported these findings with a study based on three estuaries with meiofauna included with macrofauna in the study. Evidence does appear to support that coastal squeeze will decrease benthos biomass; however Yamanaka et al. (2013) found that local conditions were important in describing macrobenthic biomass in intermediate-to-steep sloping estuary shores. This means that predicting the effect of coastal squeeze on macrobenthos without knowledge of complex local processes and conditions, and their interaction with the physical drivers affected by coastal squeeze, is likely complex if to be made reliably. Such observations do not incorporate how other feedbacks, such as changes in primary production, will interact with these trends.

Estuarine foodwebs may also be impacted by habitat modification from increased river discharge, changes in river-borne materials, and variations in the position and intensity of estuarine turbidity maxima and salinity stratification (Kirby, 2010; Uncles et al., 2014). Changes in the salinity experienced by fauna in estuaries is likely to have an impact on species diversity and distribution: a projected increase in salinity range may reduce species diversity, as found in the Thames Estuary (Attrill, 2002), and move the focus of biomass maximum higher up estuaries (Fujii, 2007). Additionally, the occurrence of hypoxic events and the vulnerability of species in estuaries to these events is likely to increase (Rabalais et al., 2009; Vaquer-Sunyer and Duarte, 2011). Key components such as benthic filter and deposit feeders have been observed to be, and are predicted to be, negatively affected through these changes (Fujii and Raffaelli, 2008). These functional groups are very important for nutrient cycling and supporting higher trophic levels such as predatory fish and birds (Austin and Rehfisch, 2003), and so can have far reaching implications for British estuarine ecology if affected beyond natural variation. Turbidity has been suggested to be important for survival and behaviour of mid-water organism such as juvenile fish through food supply and vulnerability to visual predators, which may present further implications to changes in river discharge and sediment suspension (Power et al., 2000).

In addition to these habitat modifications, climate changes to temperature may also indirectly affect populations and food webs

(Walther et al., 2002). For example, projected milder UK winters may negatively impact infaunal bivalve reproductive output and recruitment directly and through affecting predator phenology (Philippart et al., 2003; Beukema and Dekker, 2014). Predatory shore crabs *Carcinus maenas* and brown shrimp *Crangon crangon* recruit earlier after mild winters, which can result in a greater overlap with their post-larval bivalve prey and reduces bivalve and flatfish recruitment success, when compared with periods with proceeding colder winters (Power et al., 2000; Beukema and Dekker, 2014). This again may reduce the biomass of benthic macroinvertebrate species that support higher trophic levels and the estuarine ecosystem. However, as acknowledged already, the complexities of local systems need considering before broad climate change predictions are applied to individual estuaries.

5. Discussion

Global climate models, linked to impact models which predict biophysical and socio-economic impacts, propose that stabilizing global carbon dioxide levels at 550 ppm by 2150 could significantly reduce most stressors on the water cycle, compared with unmitigated emissions scenarios; for example, reducing flooding frequency by some 80—90% along the most vulnerable coastlines, which include estuaries in southeast England (Arnell et al., 2002). Even though these emissions targets will be difficult to achieve, the UK are attempting to show leadership, and are seeking international commitment to reduce carbon dioxide and other greenhouse gas emissions through the United Nations Framework Convention on Climate Change (UNFCCC). The UK government has committed to reducing the country's emissions by 60% from 1990 levels by 2050 — achieved by reducing the amount of energy we consume and by substantially increasing our use of renewable energy resources (King, 2004).

Even with these initiatives, in the short term (the 21st century) some consequences of climate change appear to be inevitable for the UK under any future emissions scenario (Arnell et al., 2002), such as sea-level rise. If we define the risk of climate-driven impact as the probability of the occurrence multiplied by the magnitude of the consequence, it is obvious that the level of risk is different for each affected party and for each region. Thus, the magnitude of potential climate-driven impacts on UK estuaries, described in this paper, and hence the level of risk, require further quantification in order to inform management strategies. Regarding flood risk and flood mitigation, Elliott et al. (2014) point out that the UK has a strategy to try to protect urban and industrial coasts but deem protection of farmland not to be economically justifiable. Therefore, rural areas require local adaptation strategies that rely on soft engineering and improved community awareness.

More effective coastal management in the UK requires reducing the potentially large uncertainties, associated with climate change projections based on limited simulation of climatic variability and complex impact predictions, so that levels of risk can be assigned accurately (Arnell et al., 2002). For example, unresolved temporal and spatial variability within climate models needs to be parameterised in the boundary forcing conditions of local-scale impact models (e.g., Lewis et al., 2011). In terms of predicting climate-driven impact, many researchers are using species distribution modelling to examine how ranges may shift under climate change. However, this approach has been criticised because most models have not included non-climate stressors, such as species competition and predation, habitat loss and land-use changes, eutrophication and invasive species, that could themselves influence distributions (McDowell et al., 2014). Additionally, state-of-the-art, process-based, morphological models are, in general, capable of predicting only the statistical behaviour defining channel geometry

and meandering on time scales of decades or longer. Therefore, we must understand the relative importance of climatic and non-climatic variables in controlling present-day distributions to predict the impacts of climate change, and also understand the associated uncertainties so that we can measure the potential risk and improve management strategies.

6. Conclusions

In this review, we summarise published literature of historic observations and current anthropogenic climate change projections for the 21st century, and the anticipated processes and impacts to UK estuarine systems, including physical, biological, and ecological changes. Fig. 3 summarises the likely main, but not all, pathways from climatic drivers to processes to impacts on UK estuaries. There is also no indication from Fig. 3 of the scale of impact; for example, the consequences of flooding may be more significant than shifts in frontal positions, although this is difficult to quantify.

In terms of the climatic drivers shown in Fig. 3, sea surface temperatures around the UK have increased by 0.7 °C over the past 30 years, and are projected with high confidence to increase further by between 1.5 °C and 4 °C, by 2100, depending on our carbon emissions. Sea-levels have risen by 1—3 mm yr^{-1} during the 20th century, and are projected with high confidence to rise further by between 0.44 m and 0.74 m, by 2100, again depending on our carbon emissions but also with significant spatial variability. Precipitation and storminess have shown high variability over the past few decades, with a general pole-ward shift and increased intensity/decreased frequency of events. Although UK river flows have generally increased during autumn and winter over the past 50 years, flow rates have decreased during spring with no detectable trends during summer. Future UK projections, albeit with low confidence and high variability, are for more clustering of events, with increased winter precipitation, river flow, surge and wave intensity, and longer dry and calm periods in summer.

Potential changes to physical, *primary*, processes (see Fig. 3) include increased flooding and coastal squeeze, caused by increased sea-level, changing surge and wave climates, and changing river flow events. The capacity for estuaries to respond to flooding will depend in the short term on available 'floodable' space, and in the long term on morphodynamics relative to sea-level rise (e.g., sediment supply and accretion, and coastal realignment). Sea-level rise will likely cause a shift towards net sediment accretion in UK estuaries, but with reduced transport overall. Turbulent mixing — critical for water quality and coastal ecology — is primarily controlled by river flow variability. Hence, potential alterations to river flows will increase the temporal variability and length scales of estuarine fronts, stratification and mixing. For example, longer dry periods in summer, in combination with sea-level rise will push the salinity maximum further up-estuary and enhance estuarine nutrification, with negative impacts of eutrophication, harmful algal blooms and hypoxia. On the other hand, clustering of storm events will alter the temporal mixing variability in winter.

Some of the *secondary impacts* on the estuarine ecosystem (see Fig. 3) may ultimately result in a risk to human health, associated with ecological disruption and heightened water-borne and food-borne microbial pathogens concentrations, mainly due to projected increased temperatures and changing river flows. Changes in these climate drivers are also likely to influence the concentrations of toxic metals, organic chemicals, algal toxins, and human pathogen contaminants in seafood. The primary impact of coastal squeeze is the greatest threat to already declining UK salt marsh habitats, which in turn may reduce their species diversity and food webs. Some intertidal habitats and biogenic reefs — often priority
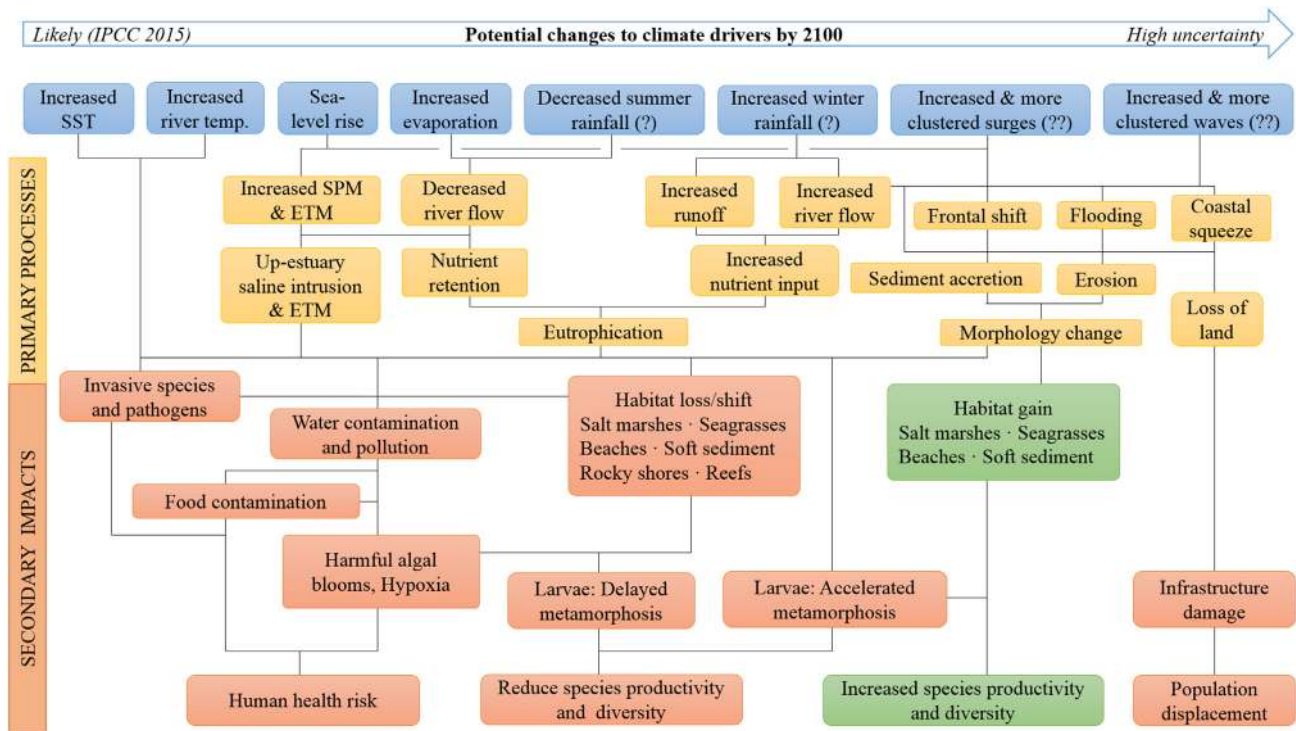
Fig. 3. Flow diagram showing the main climatic drivers, and primary and secondary impacts of climate change to UK estuaries. Pathways (black lines) are directed *from* the bottom/side *to* the top of a 'driver/impact box'. Predominantly positive impacts are coloured green. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

habitats for protection in the UK — have gradually moved north in response to rising temperatures, bringing changes in biodiversity and ecological functioning — for example, algae in rocky shores will likely become less prevalent. Projected increases in mild winters may negatively impact infaunal bivalve reproduction. Invasive species to estuaries have increased, due to temperature warming, affecting native assemblages and introducing pathogens. However, temperature rise is unlikely to affect species that live within their temperature tolerances, such as mussels. Sedimentary habitats, particularly mudflat-living species, are sensitive to an assortment of climate change drivers and primary impacts including wave attack and sediment coarsening. Changing climate drivers are likely to alter environmental cues which synchronize the reproductive cycle of many marine invertebrates and fish. Estuarine species that cue reproduction based on salinity signals are likely to be particularly vulnerable to climate change. Eventually, climate change may have implications on the conservation status of UK estuaries.

We recognise there is variety and diversity of estuarine ecosystems in Britain, and variety in drivers of change that require more detailed quantification/prediction. We therefore recommend that future research should focus on improving our understandings of the primary and secondary impacts to UK estuaries, on a case-by-case basis, caused by potential changes to regional climatic drivers; for example, through improved modelling and model coupling, accompanied by long-term monitoring. Current UK research funding, such as the NERC Macronutrient Cycles Programme, aims to contribute to this via integrated climate-catchment-river-estuary process modelling combined with intensive sampling across several UK systems. Some emerging outcomes of this programme are that climate-driven, coupled, soil-catchment-river models are required to accurately predict future river flows and nutrient fluxes. Further, uncertainties associated with commonly-used daily-averaged river flows to infer impact downstream in the estuary appear to be significant, emphasising the need for

catchment modelling.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.ecss.2015.12.016.

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