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Future variability of solute transport in a macrotidal estuary

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ABSTRACT

The physical controls on salt distribution and river-sourced conservative solutes, including the potential implications of climate change, are investigated referring to model simulations of a macrotidal estuary. In the UK, such estuaries typically react rapidly to rainfall events and, as such, are often in a state of non-equilibrium in terms of solute transport; hence are particularly sensitive to climate extremes. Sea levels are projected to rise over the 21st century, extending the salinity maximum upstream in estuaries, which will also affect downstream solute transport, promoting estuarine trapping and reducing offshore dispersal of material. Predicted 'drier summers' and 'wetter winters' in the UK will influence solute transport further still; we found that projected river flow climate changes were more influential than sea-level rise, especially for low flow conditions. Our simulations show that projected climate change for the UK is likely to increase variability in estuarine solute transport and, specifically, increase the likelihood of estuarine trapping during summer, mainly due to drier weather conditions. Future changes in solute transport were less certain during winter, since increased river flow will to some extent counteract the effects of sea-level rise. Our results have important implications for non-conservative nutrient transport, water quality, coastal management and ecosystem resilience.

1. Introduction

Estuaries are regions where saline sea water interacts with fresh river water and other river-sourced materials. Predictions of future changes to either the mean sea level or river flow climatology will likely have a critical impact on estuaries (e.g. Ranasinghe et al., 2012; Rice et al., 2012). The fate of conservative solutes within an estuary depends on the oscillatory tidal mixing in conjunction with freshwater discharge from river systems and the morphology of the estuary (West and Cotton, 1981; Struyf et al., 2004). Hence, whether river-derived solutes will potentially damage coastal ecosystems is determined by processes within the estuary and will be altered by climate change. Yet such processes have rarely been investigated in realistic environments, or for projected future climatic conditions. Pronounced seasonal variations often occur in temperature, river flow, and solute concentrations. These variations, combined with extreme atmospheric events, are reported to be critical for estuarine functioning, water quality, and coastal ecology (Prandle, 2009). Thus an understanding of estuarine solute transport is important to improve prediction of coastal processes which can inform coastal management decisions and help mitigate potential pollution problems.

Tidally-energetic estuaries that are connected to steep river catchments may often be in a state of non-equilibrium because of nonlinear tidal influences on the mixing/stirring balance, the bathymetry of the estuary, and the asymmetric nature of the hydrograph (the river rises very quickly after rainfall but river flowrates reduce more slowly). This causes 'tidal-pumping' in some estuaries which predominates the upstream transport of salt and retains substances flowing from the river catchment, in competition with rapid downstream transport due to rainfall events which flush salt and nutrients further downstream and potentially offshore (Simpson et al., 2001). Slower responding catchments (i.e. with flatter or longer rivers) are more likely to be in equilibrium. Somewhat surprisingly, only sporadic observations of estuarine tidal pumping have been documented (e.g. Simpson et al., 2001) and there is generally a lack of knowledge about how solutes react to such events. Estuarine adjustment after a flushing event acts to restore a steady state salt distribution (Kranenburg, 1986; MacCready, 1998; Hetland and Geyer, 2004).

Climate change projections suggest that UK mean sea level will rise broadly in parallel to global sea level rise projections; between 18 and 59 cm over the 21st century, however rises above 1 m cannot be ruled out (Lowe et al., 2009; Lewis et al., 2011). It is therefore important to understand how estuarine dynamics and solute transport will adapt to sea-level rise (e.g. Friedrichs and Aubrey, 1988). Climate models also predict that over the 21st

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century UK summers are likely to be drier and winters are likely to be wetter (Lowe et al., 2009); changes which could have significant and, as yet, unquantified implications for estuarine functioning and the fate of solutes. Of particular importance to hydrological processes are extreme flows and their variability (e.g. Arnell, 2003). Moreover, due to the mixing-stirring balance within estuaries, changes to extreme river flows may be of great importance for estuarine dynamics. Hence, we hypothesise that the spatial variability in solute transport and dispersal of pollutants will increase due to climate change; sea-level rise acting to pump the salinity maximum further upstream, in conjunction with increased variability in river flows as predicted for the northwest UK.

To simulate potential future conditions in an estuary, numerical models may provide sufficiently high temporal and spatial resolution to allow effective simulations of estuarine transport. In this paper, we investigate present and future controls on estuarine solute transport, focussing on a model case study of the Conwy Estuary, UK – a macrotidal system which adjoins a steep river catchment. We develop a vertically-averaged, finite-element, model at high spatial resolution, using extensive observational data in the Conwy for validation. Adjustments to estuarine circulation due to sea-level rise and altered river flow under potential future climate conditions are then simulated, allowing us to investigate changes to the salt balance and conservative solute transport such as a river-sourced pollutant or conservative nutrient.

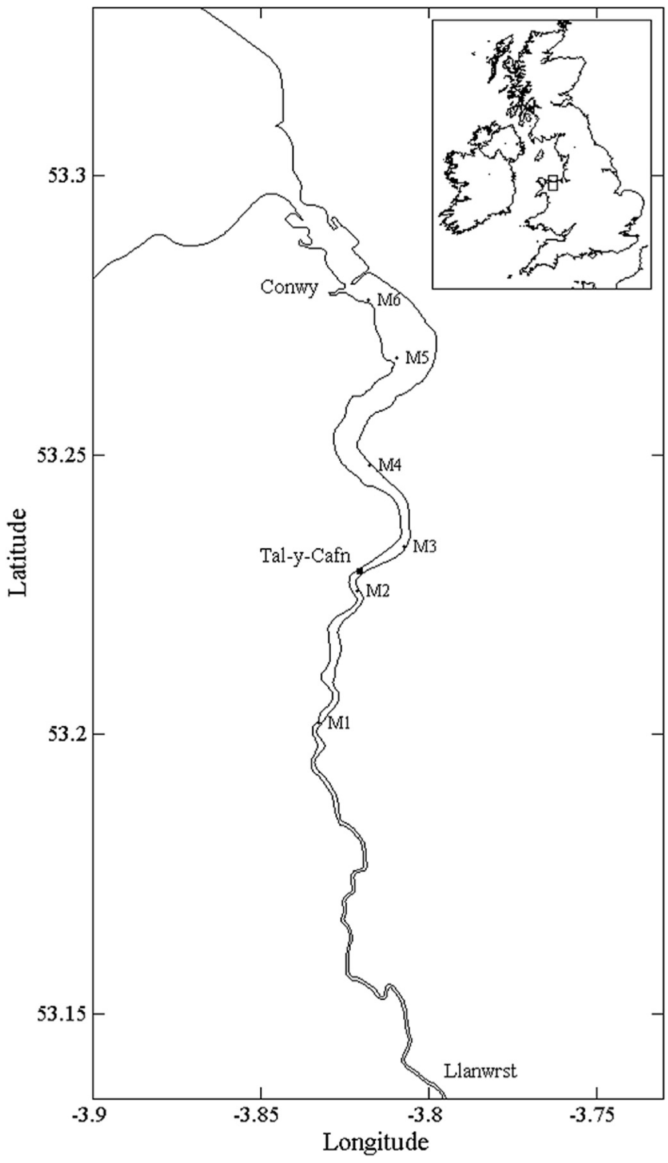

Fig. 1. Map of the Conwy Estuary, North Wales (boxed area in inset map of the UK). The positions of 6 deployment stations in 1986 (M1–M6) are marked, together with a deployment station in 1996 at Tal-y-Cafn. The river gauging station is located near the town of Llanrwst.

2. Methods

2.1. Case study: The Conwy Estuary, North Wales, UK

The Conwy Estuary (Fig. 1) has been chosen as our case study since considerable observational data, collected in the estuary and in the catchment, was available for model validation and input parameterisation. Also, the estuary is macrotidal with shallow water depths, which is typical for the UK and therefore representative of other estuarine systems. The estuary is tidally dominated (spring tidal range is typically 6 m) with tidal volume exchange exceeding mean river input by a factor of 20 (Simpson et al., 2001). Mean flow in the River Conwy between 1964 and 2012 was approximately 19 $m^3\ s^{-1}$, based on National Resources Wales (http://naturalresourceswales.gov.uk) flow gauge measurements near Llanrwst (Cwm Llanerch; SH80258) which is beyond the extent of the tidally-influenced river (see Fig. 1), although flow varied in the range 0.3–433 $m^3\ s^{-1}$. Based on standard average annual rainfall (SAAR) data, available for the period 1961–2012 on a 1 km grid, approximately 73% of the freshwater input into the estuary enters from the River Conwy and the remaining input enters directly from slope run-off and smaller streams. Strong tidal mixing results in a vertically near-homogeneous salinity structure for the majority of the tidal cycle (Simpson et al., 2001). The estuary extends approximately 25 km upstream from the estuary mouth which is located on the north Wales coast and connects with the Irish Sea. The morphology is such that the estuary almost entirely drains each tidal cycle, and resembles a meandering river channel at low slack water, flanked by muddy tidal flats in the upper estuary and sand in the lower estuary. Due to the mountainous topography of the surrounding catchment, combined with some lowland reclamation for agricultural purposes, the estuary does not contain extensive tidal flats or salt marsh regions.

2.2. Hydrodynamic modelling

A finite-element hydrodynamic model (TELEMAC Modelling System V6.2; Hervouet, 2007) was applied to the Conwy Estuary (Fig. 1), named hereafter the Conwy Model. The Conwy Model was used to predict at high resolution barotropic circulation and baroclinic processes which generate the longitudinal salinity structure and circulation. The hydrodynamic module (TELEMAC-2D) is based on the depth-averaged shallow water Saint–Venant equations of momentum and continuity, derived from the Navier–Stokes equations (Hervouet, 2007). The hydrostatic assumption of the model is valid in this region where bed slopes are small and vertical accelerations caused by the pressure are also small. The classical k–ε turbulence model has been adapted into vertically averaged form to include additional dispersion terms (Rastori and Rodi, 1978); a constant internal friction coefficient of 3×10^{-2} m was implemented in Nikuradse's law of bottom friction (Hervouet, 2007). Turbulent viscosity has been set constant with the overall viscosity (molecular + turbulent) coefficient equal to 10^{-6}. Coriolis effects have also been included although are unlikely to affect the hydrodynamics at estuary scale. The unstructured model mesh, created using BlueKenue® grid generation software, has variable resolution being relatively fine (approximately 15 m) within the estuary and coarser (50–500 m) offshore (the domain extended approximately 20 km offshore in order to resolve the shallow water tidal propagation). The mesh was mapped onto a bathymetric grid comprising data of the same order of resolution; gridded Admiralty data available in offshore regions at 250 m resolution (EDINA, 2008), LIDAR data in intertidal regions (available from the Environment Agency Wales) at 10 m resolution, and a single-beam echosouder survey of the sub-tidal estuarine channel at 10 m resolution which was conducted by Bangor University in 2003. Wetting and Drying is implemented in our hydrodynamic model, allowing inter-tidal regions that 'dry out' during low waters to be simulated which is a key feature of estuaries. TELEMAC-2D has been proven to accurately simulate similar estuarine processes (e.g. Robins and Davies, 2010; Jones and Davies, 2011).

2.3. *Model validation*

Simulations were performed to validate the Conwy Model against hydrodynamic and salinity measurements observed previously in the estuary. Data from three separate surveys were available, collected at stations within the estuary by Bangor University during September 1986 (duration of 25 days; see Turrel et al. (1996) for details about the field survey), January 1996 (duration of 10 days; see Simpson et al. (2001)), and September 2013 (unpublished data of duration of 15 days). A model spin-up and initialisation period preceding each simulation was required so that the salt balance approximately reached a steady–state equilibrium, from an initial 'zero salinity' condition of the model. Therefore, an idealized baseline simulation was performed, forced by semi-diurnal lunar (M_2) and solar (S_2) tidal constituents, which describe the dominant spring-neap cycle of tides in the UK, and realistic flowrates from the River Conwy (constant minimum observed value of 0.32 $m^3\ s^{-1}$). Ocean salinity values of 35 were introduced at the offshore boundary which diffused laterally into the model domain. Tidal pumping of salt was simulated, in agreement with the findings of Simpson et al. (2001), until steady–state equilibrium was reached after approximately 120 days (Fig. A1). We chose minimum observed flowrates for the baseline simulation so that all subsequent model scenarios would be spun-up in the same way from a minimum-flow salinity distribution.

Each validation simulation was initialized with the final salinity distribution of the 120-day baseline simulation (i.e. 'hot-started') and forced at the open offshore boundary with four principle tidal constituents: M_2, S_2, $N_{2,}$ and K_1. Validation simulations extended the model spin-up by 10 days prior to instrument deployment in order to capture the preceding freshwater influence on the salinity distribution (i.e. total simulation periods were 120 + 35 days for the 1986 data, 120 + 20 days for the 1996, and 120 + 25 days for 2013). Tidal harmonics at the model boundary were attained from a global tidal model at 1/16° resolution (FES-2012; http://www.aviso.oceanobs.com), which compared favourably with output generated from models of the European shelf (Neill and Hashemi, 2013) and the Irish Sea (Robins et al., 2013). Daily-averaged river flowrate data at Cwm Llanerch, which is at the position of the model boundary, were interpolated to the model time step and used to force the freshwater input into the model.

In order to accurately simulate the observations commencing on 31 January 1996, the modelled salinity diffusion coefficient was calibrated to equal 0.066 $m^2\ s^{-1}$, which corresponds with values estimated in the Conwy (using dye tracing techniques) by West and Cotton (1981). Field data consisted of a current meter and tide gauge suspended off the bridge at Tal-y-Cafn (which is in the river–estuary transition zone). Data-model comparisons of elevation, longitudinal velocity, and salinity are shown to be approximately in agreement (Fig. 2a–c). Root mean squared errors were 0.46 m, 0.25 m s^{-1}, and 3.94 for elevation, velocity, and salinity, respectively. Importantly, the model broadly captured the nonlinear asymmetry of the tidal forcing and the influence of the varying river flow, which is seen most clearly in the velocity time series

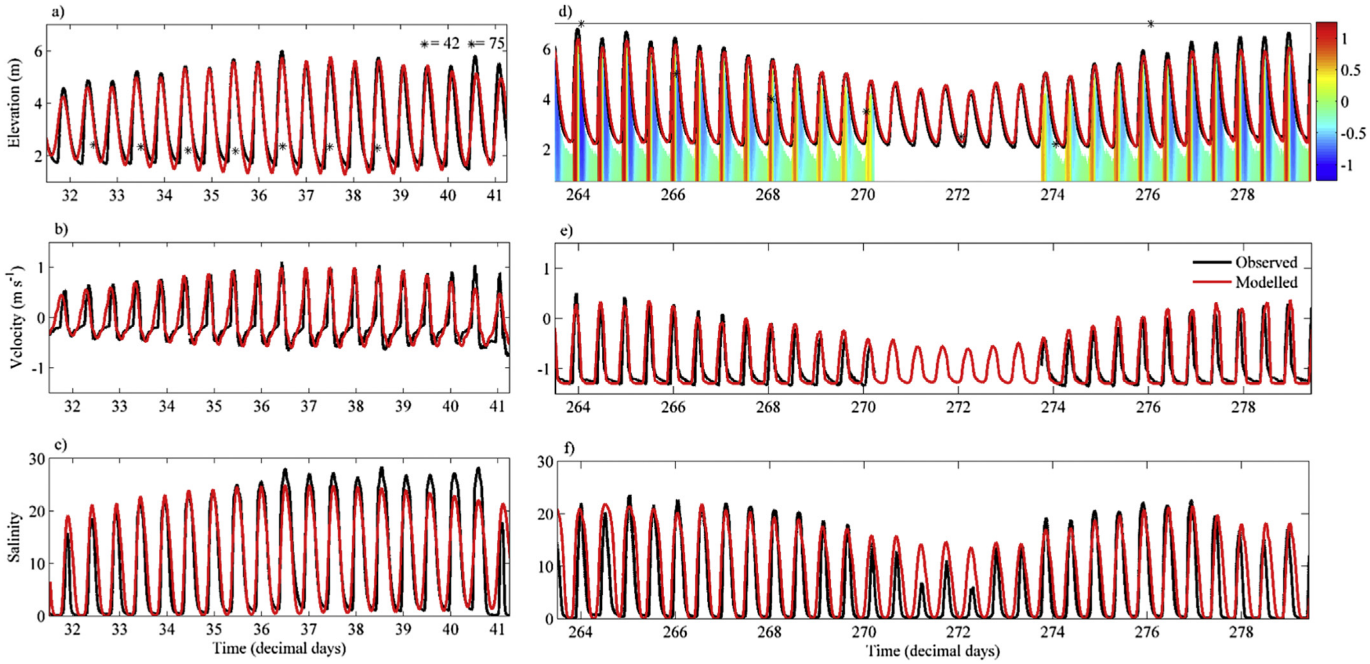

Fig. 2. Model – data comparisons at Tal-y-Cafn Bridge (*see* Fig. 1). Left panels (a–c) show comparisons commencing 31 January 1996, while right panels (d–f) commence 20 September 2013. Elevations (a and d) are in m above Ordnance Datum Newlyn. Also depicted by asterisks are daily-averaged river flowrates ($m^3\ s^{-1}$), recorded at Llanrwst (river model boundary). Time series of ADCP longitudinal velocities ($m^2\ s^{-1}$) throughout the water column are also shown in (d). Longitudinal velocities in (b) compare mid-depth current meter data to depth-averaged model data, whereas in (e) depth-averaged data are compared. Finally, mid-depth salinities in (c) are compared to depth-averaged model data, whereas both surface and bottom (dashed black line – difficult to see) measured salinities are plotted in (f) and compared to depth-averaged model data.

(Fig. 2b). Discrepancies in salinity may in part be due to neglecting additional freshwater sources downstream of Llanrwst, which amount to approximately 27% of the total freshwater input.

With no further calibration of the salinity diffusion coefficient, the second validation simulation, commencing on 20 September 2013, was performed. An Acoustic Current Doppler Profiler (ADCP) and a Seabird Conductivity-Temperature-Depth (CTD) profiler were deployed in the centre of the channel at Tal-y-Cafn Bridge. Again, the model performed well at reproducing the observed elevations, depth-averaged velocities, and depth-averaged salinities, resulting in root mean squared errors of 0.26 m, 0.20 m s^{-1}, and 3.68, respectively (Fig. 2d–f). The ADCP velocity data throughout the water column (Fig. 2d), together with CTD salinity profiles at the surface and bottom (Fig. 2f), enables the near-homogeneous nature of the water column to be visualised which helps explain how the 2D simulations are able to reproduce the circulation and solute transport.

Finally, validation of Aanderaa (RCM4) current meter data collected in September 1986, was performed. The model output were also approximately in agreement with the observations at six stations along the estuary (Fig. 3) and, therefore, represents some validation of the longitudinal salinity gradient. Averaged over stations M1–M6, the root mean squared error in salinity between the observations and simulations was 4.95 (unfortunately, measurements of elevation and velocity over this period were erroneous and cannot be compared with the model simulations). Over-prediction of low water salinity at Stations M4–M6 was simulated and is likely due to un-simulated freshwater input into the lower section of the estuary, or a degree of stratification at low slack water which was not reproduced by the depth-averaged model.

Since the model validates well for longitudinal velocities and salinities, and also conserves mass, we can assume that the cross-channel modelled flow is also accurate. Further, for each validation scenario, we assume the estuary geometry at the time of our bathymetric surveys (2003 and 2008), since data at the validation times were not available. However, changes to bathymetry over this period are likely to be negligible in comparison to other uncertainties. As an example, we have performed a simulation in which the bathymetry has been modified by 1 m, in accordance with the water-level uncertainty due to the absence of storm surge, and we find that the simulated results make negligible difference in terms of velocities and salinities (Fig. A2). Moreover, because the estuary is predominantly vertically mixed (Simpson et al., 2001, Fig. 2), it is reasonable to assume that the 2D model simulations will reproduce the longitudinal hydrodynamics and the salinity structure, as corroborated by our validation results described above. However, to check this assumption, a 3D sensitivity test simulation was performed for a period of 25 h, at considerably higher computational cost. The 3D simulation was again spun-up using the 120-day baseline (2D) simulation, and then compared to a similar 2D simulation of 25 h. In terms of the vertically averaged circulation and salinity, output between the 2D and 3D simulations did not vary significantly (Fig. A3), confirming that the 2D model is indeed appropriate for this study.

2.4. Idealised simulations

Our validated Conwy Model was used to investigate controls on the longitudinal salinity structure and conservative tracer transport, in terms of tidal forcing, freshwater flow, and projected climate change over the next century. All our idealised simulations were initiated (hotstart) with the 120-day baseline (spin-up) simulation described previously and then run for a further 10 days, starting 3 days before neap tides so that river flow has time to

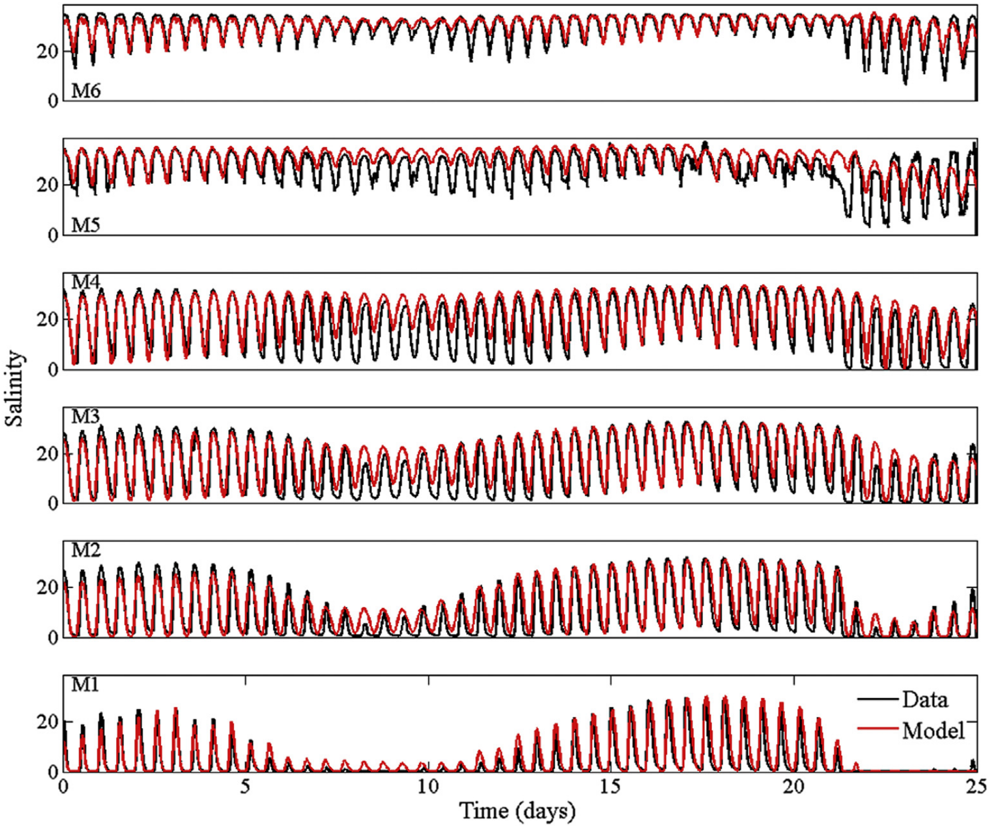

Fig. 3. Model-data comparisons of salinity at six stations within the Conwy Estuary (*see* Fig. 1). The records commence on 19 September 1986. Practical salinity units are used. (n.b. no further calibration was performed from the previous validation simulations; *see* Fig. 2).

Table 1
Summary of Conwy Model idealised simulations.

Observed river flowrate, Q ($m^3\ s^{-1}$)		Tidal regime	2014	2100: Sea-level rise projections			
				H_{min} +0.18 m	H_{max} + 0.59 m	H^{++} + 1.9 m	H_{max} + 0.59 m
Extreme low:	0.3	$M_2 + S_2$	RUN-1.1	RUN-2.1	RUN-3.1	RUN-4.1	**Q − 40%**: RUN-5.1
Summer mean:	9		RUN-1.2	RUN-2.2	RUN-3.2	RUN-4.2	**Q − 40%**: RUN-5.2
Annual mean:	19		RUN-1.3	RUN-2.3	RUN-3.3	RUN-4.3	–
Winter mean:	29		RUN-1.4	RUN-2.4	RUN-3.4	RUN-4.4	**Q + 20%**: RUN-5.4
Extreme high:	433		RUN-1.5	RUN-2.5	RUN-3.5	RUN-4.5	**Q + 25%**: RUN-5.5

adjust from minimum-flow conditions, and ending during spring tides. The offshore boundary was forced with salinity values of 35, tidal elevations comprised M_2 and S_2 constituents, and varying mean sea level to represent sea-level rise. The river boundary was forced with varying freshwater flow and a continuous tracer concentration – note that the dispersion of salt behaves differently to that of a tracer due to density affects (Fig. A4). In total, 24 separate simulations were produced as summarised in Table 1.

2.4.1. Scenario 1: present-day

Based on daily-averaged river flowrates measured at Cwm Llanerch between 1964 and 2012, the following 5 present-day (i.e. no sea-level rise) simulations have been produced (*see* Table 1): Extreme low flow (RUN-1.1); summer mean flow (RUN-1.2); annual mean flow (RUN-1.3); winter mean flow (RUN-1.4); extreme high flow (RUN-1.5). Simulations of tracer transport are used to estimate the proportion of a conservative solute that travels from river to estuary and potentially offshore.

2.4.2. Scenarios 2–4: sea-level rise

Global mean sea level has risen at a rate of 1–2 mm yr^{-1} since the last century (Church et al., 2004; IPCC, 2007) with an apparent acceleration to 3 mm yr^{-1} since the 1990's, 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). As carbon dioxide emissions increase global temperatures, steric (thermal expansion) and eustatic (ocean volume changes) processes will result in further sea-level rise (IPCC, 2007).

Three-dimensional coupled Atmosphere-Ocean Generalized Circulation Models (AOGCMs) simulate climatic sea-level change, assuming several carbon dioxide emissions scenarios (IPCC, 2000; Lowe and Gregory, 2006). Based on an ensemble of AOGCMs and emissions scenarios, global mean sea level is predicted to rise by between 0.18 m and 0.59 m by 2100 depending upon social, oceanic and atmospheric interactions, and intra-model uncertainties such as sub-grid-scale model parameterisations (e.g. IPCC, 2007; Lowe et al., 2009). Uncertainties within the AOGCM approach has led to parametric relationships between sea level and air temperature to be developed (Jevrejeva et al., 2010) which suggest that sea-level rise up to 1.9 m is possible by 2100 (Rahmstorf, 2007; Grinsted et al., 2009).

Regional variations in mean sea level are important for coastal communities, and are caused by atmospheric processes (e.g. El-Nino), heating, circulation patterns (Jevrejeva et al., 2006; IPCC, 2007), and local vertical land movements including natural and anthropogenic (e.g. groundwater extraction) processes (Shennan and Horton, 2002; Nicholls et al., 2007). Observed sea level trends in the UK are broadly consistent with the global average (Woodworth et al., 2009). Further, when considering the uncertainties within future sea level projections, global mean sea-level rise projections are generally used for the UK (e.g. Lewis et al., 2011). Therefore, we re-simulated Scenario-1 for three sea-level rise projections for 2100 using the UKCP09 convention (*see* Lowe et al., 2009): Scenario-2 – minimum sea-level rise (H_{min} = 0.18 m), Scenario-3 - maximum sea-level rise (H_{max} = 0.59 m), and Scenario-4 - a worst case scenario (H^{++} = 1.9 m) – *see* Table 1.

2.4.3. Scenario 5: future flowrates in the Conwy

The UKCP09 report, which utilised the 150 year HadRM3 Regional Climate Model simulation with 11 ensemble members (to incorporate model parameter uncertainty and decadal variability), 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). Although a high degree of bias (hence the ensembles inability to capture monthly precipitation climates) has been found within modelled precipitation series (Prudhomme and Davies, 2009; Smith et al., 2013), statistical down-scaling techniques through river catchment models have revealed a high likelihood of reduced summer flows and increased winter flows for western mountainous regions (Christierson et al., 2012) – where the Conwy catchment is located. The consensus of future rainfall projections agree with a "drier summers/wetter winters" signal (e.g. Hulme and Jenkins, 1998; Hulme et al., 2002; Jenkins et al., 2009; Smith et al., 2013), with bias-corrected dynamically downscaled models simulating daily river flows with some degree of accuracy (e.g. Fowler and Kilsby, 2007; Smith et al., 2013).

To understand if modelled river flow projections could simulate future flows in the Conwy catchment, freely available model output from HadRM3 (downloaded from http://www.ceh.ac.uk/data/nrfa/data/futureflows.html: *see* Prudhomme et al., 2012, 2013) were compared with gauged daily flowrates at Cwm Llanerch (for the same period between 1964 and 2012). Using q–q plots and a two-sample Kolmogorov–Smirnov goodness-of-fit hypothesis test (produced in Matlab®) with a null hypothesis (at the 5% significance level), the two samples were found to not be of a similar distribution (Fig. A5). Only the upper-extreme flowrate quantiles deviate from the linear equal distribution trend; i.e., the HadRM3 model did not catch the extreme (upper 1–2%) river flowrates (Fig. A5). Therefore, we found the simulated river flow hindcast did not match the observed river flow climate; hence the model output for this catchment cannot be used without further work (e.g. statistical downscaling), which is beyond the scope of our research.

Fowler and Kilsby (2007) dynamically downscaled HadRM3 into a northwest UK river catchment model, and found a significant change in annual river rates between 1960–1990 and 2070–2100. Further, Fowler and Kilsby (2007) found summer reductions of 40–80% and winter increases of up to 20% (extreme high flows (5% exceedence) increased by up to 25%) in the mean monthly river rates. Therefore, based on the more conservative results of Fowler and Kilsby (2007), we produce a fifth set of idealised simulations incorporating one sea-level rise scenario (2100 H_{max}) with the following future river flowrates (*see* Table 1): Extreme low – 40%, summer mean – 40%, winter mean + 20%, and extreme high + 25%.

Our idealised experimental design was intended as a sensitivity test of mean and extreme river flow/sea-level rise conditions which are based on historical flow conditions and the latest climate predictions. The Conwy catchment is relatively short and steep which, together with dominant tidal forcing of the estuarine circulation, results in a rapid response to changes in river flow. Consequently, our experimental design will, in this case, accurately represent an isolated rainfall event occurring at low spring tide or high neap tide, for example.

3. Results

The normalised longitudinal salinity distribution was calculated from the ratio S_x/S_0, where S_x is the sectionally-averaged salinity x km upstream of the estuary mouth and S_0 is the offshore salinity (35). The estuarine longitudinal salinity distribution is shown here to be influenced by both tidal forcing and river flow, based on realistic simulations of the Conwy Estuary under mean and extreme conditions (Fig. 4). In terms of tidal forcing, as one might expect, the estuary is generally more saline at high water than at low water, but the salinity distribution during spring tides is markedly different to that during neaps (Fig. 4). Averaged over a tidal cycle, for example, waters are largely more saline during neaps than springs, and the salinity maximum during neaps is positioned further upstream during neaps (Fig. 4). This can be attributed to reduced turbulent mixing during neap tides.

Variability in the longitudinal salinity distribution is also controlled by river flow, especially during low flow events. During periods of extreme low river flow, such as a drought, saline waters penetrate upstream into the tidally-influenced river, more than 20 km from the estuary mouth (Fig. 4a). Under more usual summer conditions (i.e. river flow of 9 $m^3\ s^{-1}$), salt does not penetrate more than 14 km from the estuary mouth (Fig. 4b), even though water depths are affected by tidal forcing approximately 5 km further upstream (results not shown). The freshwater plume dilutes salinities up to 4 km offshore (Fig. 4b). As the simulated river flow was increased to represent annual mean and winter mean levels (i.e. 19 $m^3\ s^{-1}$ and 29 $m^3\ s^{-1}$, respectively), the upstream saline

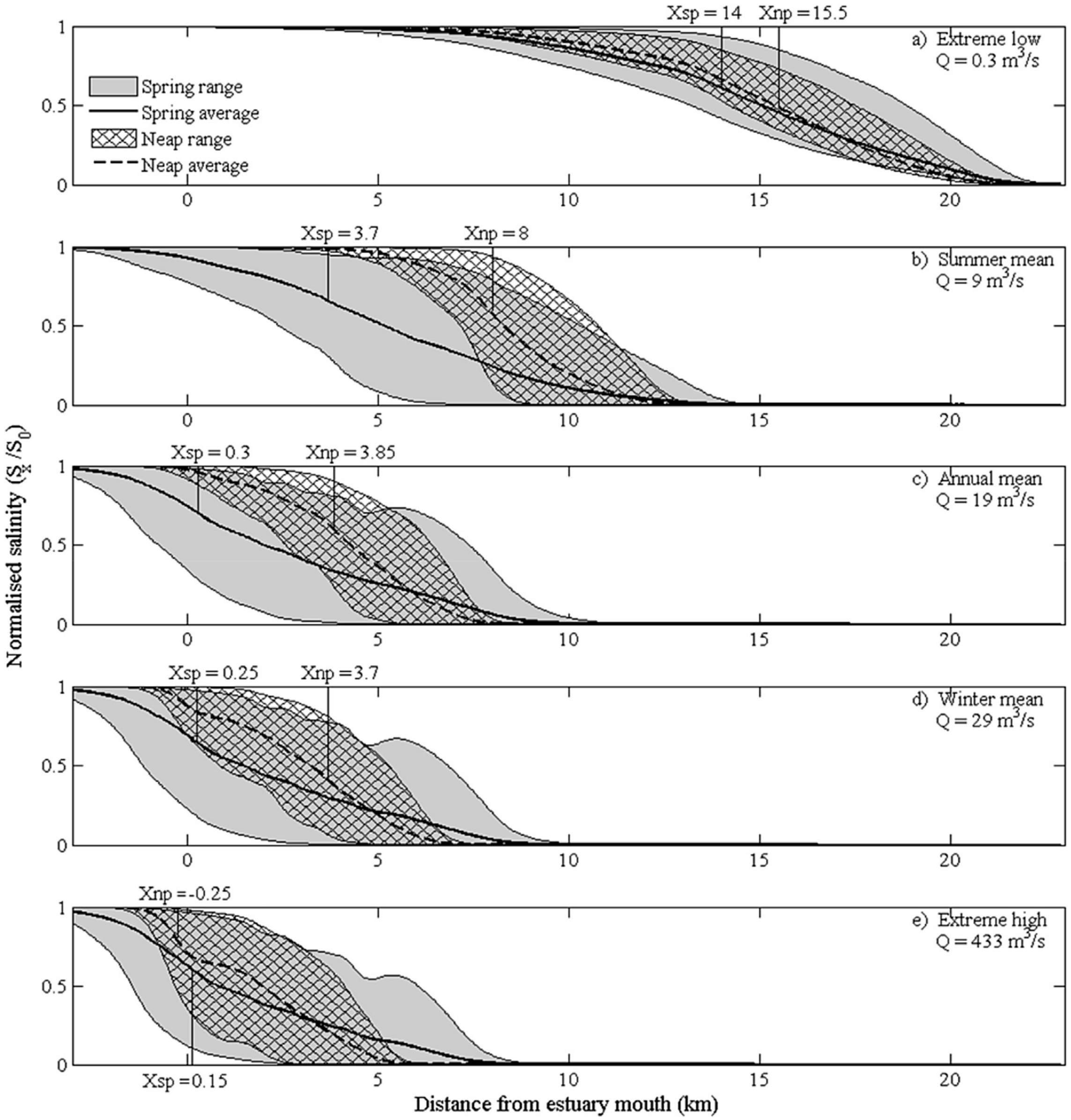

Fig. 4. Modelled longitudinal salinity distribution for the Conwy Estuary: A range of river flowrate conditions are presented (a–e), each for a typical spring-neap tidal cycle and with present-day sea levels. Results are based on depth-averaged currents and assume vertically homogenous salinity profiles.

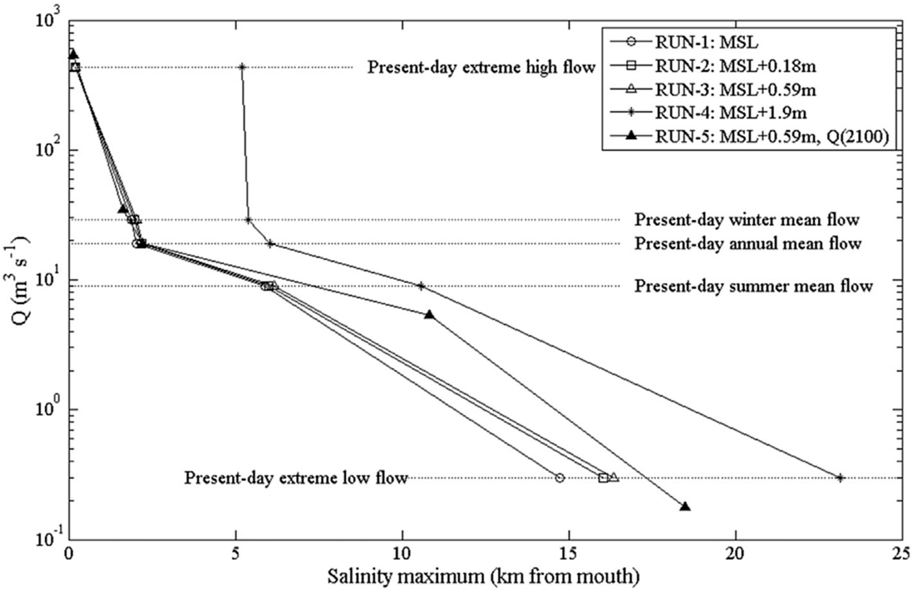

Fig. 5. Positions of the tidally-averaged (over a neap-to-spring period of 8 days) salinity maximum under varying flow conditions and sea-level rise.

intrusion was reduced to approximately 10 km from the estuary mouth, although the salinity structure and positions of the salinity maxima remained similar for both flowrate conditions (Fig. 4c and d). Even when flowrates were increased dramatically to extreme high values of 433 $m^3 s^{-1}$, the longitudinal salinity structure did not alter significantly from the annual mean case (Fig. 4e). Therefore, for periods when river flow exceeds annual mean values of 19 $m^3 s^{-1}$ (i.e. half of the year), our results suggest that variability in the longitudinal salinity structure is largely controlled by tidal forcing, rather than river flow.

Interestingly, our sea-level rise simulations of 0.18 m (our minimum scenario) and 0.59 m (our maximum scenario) did not significantly alter the salinity structure described above (Fig. 5) – salt penetrating only slightly further upstream during low river flow conditions. However, notable alterations from the present-day pattern were simulated with extreme (H^{++}) sea-level rise of 1.9 m (Figs. 5 and 6). It is worth pointing out that, with 1.9 m sea-level rise and extreme low river flow, the entire estuary will become markedly more saline (Fig. 7a). Yet, if future river flow in the UK is altered as projected (Fowler and Kilsby, 2007), resulting in drier summers and wetter winters, the estuary will become more saline still during periods of low flow, but largely unchanged during high flow (as shown for RUN-5 with 0.59 m sea-level rise; Fig. 5). Therefore, neglecting the H^{++} sea-level rise scenario, our results (Fig. 4) show that the prediction of drier summers in the UK is likely to cause the greatest change to the variability of the salinity distribution, for steep catchment estuaries like the Conwy.

The dispersal of a continuous river-sourced conservative tracer was simulated with the Conwy Model, and concentrations of the tracer were dispersed offshore during all river flow conditions – to some extent (Figs. 7 and 8). However, tracer dispersal was modulated by tidal forcing; tracer concentrations mainly dispersed offshore during the ebb phase of the tide and concentrations at the estuary mouth were higher during spring ebb tides than during neap ebb tides (Fig. 8). As seen for the salinity simulations, variability in the offshore tracer concentration was also dependent of river flow; less than 5% of the tracer dispersed offshore during the extreme low river flow scenario (Fig. 7b). During mean summer flow, a maximum of 35% of the source tracer dispersed offshore (Fig. 7a and c), whereas up to 90% of the source tracer dispersed offshore during mean winter flow (Figs. 7c and 8e). Again, low river flow conditions caused the greatest variability in tracer concentrations.

Sea-level rise was simulated to reduce the proportion of tracer concentration that dispersed downstream and offshore (Fig. 8), since increasing water depths in our model resulted in the upstream migration of the position of the salinity maximum, as explained previously. When we simulated combined sea-level rise and altered river flow (Scenario-5), the downstream transport of the conservative tracer was reduced significantly during summer (Fig. 8b and c); by approximately 60% because of the combined actions of sea-level rise and reduced flow (i.e. present-day mean summer flow *minus* 40%). But less change was simulated during winter (Fig. 8e and f), since increased flow (i.e. present-day mean winter flow *plus* 20%) effectively cancelled out the impact of sea-level rise.

4. Discussion

Estuaries are at the transition zone between freshwater and marine environments and therefore are among the most productive systems in the world (Struyf et al., 2004); of vital importance are habitats for marine flora and fauna, food provision, and other ecosystem services such as tourism (Costanza et al., 1997; Wetz and Yoskowitz, 2013). Changes in terms of land use are resulting in the mobilization of nutrients such as carbon, nitrogen and phosphorus (Cloern 2001), and other contaminants including human health pathogens. For example, diffuse and point source nutrient releases arise from sewage outfalls, agriculture runoff, septic tank drainage, and industrial effluents. Over time, these nutrient concentrations have increased in estuaries leading to negative impacts such as eutrophication and poor water quality which can potentially threaten ecosystem health (Jarvie et al., 2012). Increasing fluxes of nutrients through the freshwater–estuarine interface are likely future drivers of changing biogeochemistry of the coastal seas (Jarvie et al., 2012). In addition, and although difficult to quantify, climate change leading to altered flushing regimes can retain harmful nutrients in estuaries and likely exacerbate economic, environmental and social impacts currently suffered in these areas (Jarvie et al., 2012). Indeed, sea-level rise and altered river flow

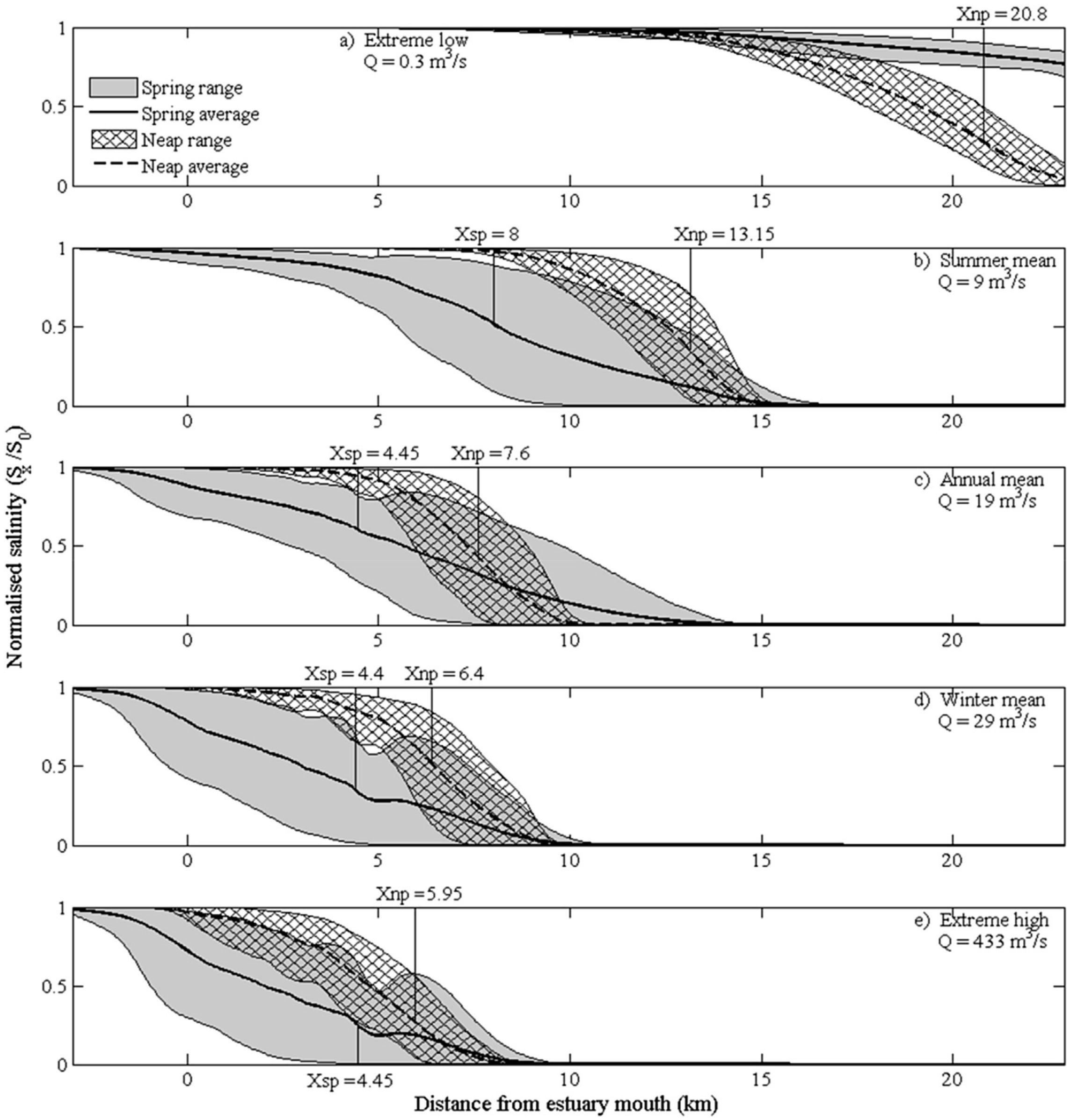

Fig. 6. Modelled longitudinal freshwater fraction for the Conwy Estuary: A range of river flowrate conditions are presented (a–e), each for a typical spring-neap tidal cycle and with extreme H^{++} projected sea levels in 2100 (i.e. + 1.9 m). Results are based on depth-averaged currents and assume vertically homogenous salinity profiles.

have been identified as key threats to the future of Britain's estuaries (e.g. Kennish, 2002; Wilby et al., 2005); yet much research has focused upon the future flood risk and morphology (e.g. Uncles, 2003; Prandle, 2009; Ranasinghe et al., 2012) rather than impacts to water quality - hence climate change and extreme event implications to estuarine functioning have been investigated here.

In the UK, many coastal systems link tidally energetic estuaries that are vertically-mixed to steep catchments where rainfall rapidly flows into the river system (Brown et al., 1991). Our study of one such system, the Conwy, suggests that estuarine trapping of river-sourced solutes occurs due to 'non-equilibrium' upstream tidal pumping. Our simulations with varying river flows intimate that high river flow events will rapidly 'flush' solutes offshore in between more gradual upstream estuarine trapping, whereas most variability in solute transport occurs during low river flow conditions. These results echo those of Struyf et al. (2004), who sampled the Schelde estuary, Belgium, and found that high discharge resulted in a downstream shift of the salinity gradient with lower salinities and higher nutrient concentrations at the mouth. Interactions between extreme low river flow (high nutrient trapping) followed by extreme high flow (nutrient flushing) have been noted to increase the overall downstream nutrient loading (Wetz and Yoskowitz, 2013).

Sea-level rise will increase the upstream saline intrusion, although according to our simulations, alterations over the next century will be small relative to present-day variability caused by tidal mixing and river flow. This outcome assumes that the estuary shape does not markedly change due to coastal flooding – perhaps if the estuary abuts steep topography or its shape is maintained by man-made structures (e.g. coastal flood defences). Therefore, regional variations to flood risk policy should also consider water quality effects in addition to monetary flood risk based decisions to shoreline management plans. For example, building higher coastal defences, or 'managed retreat' (such as letting fields flood) need to

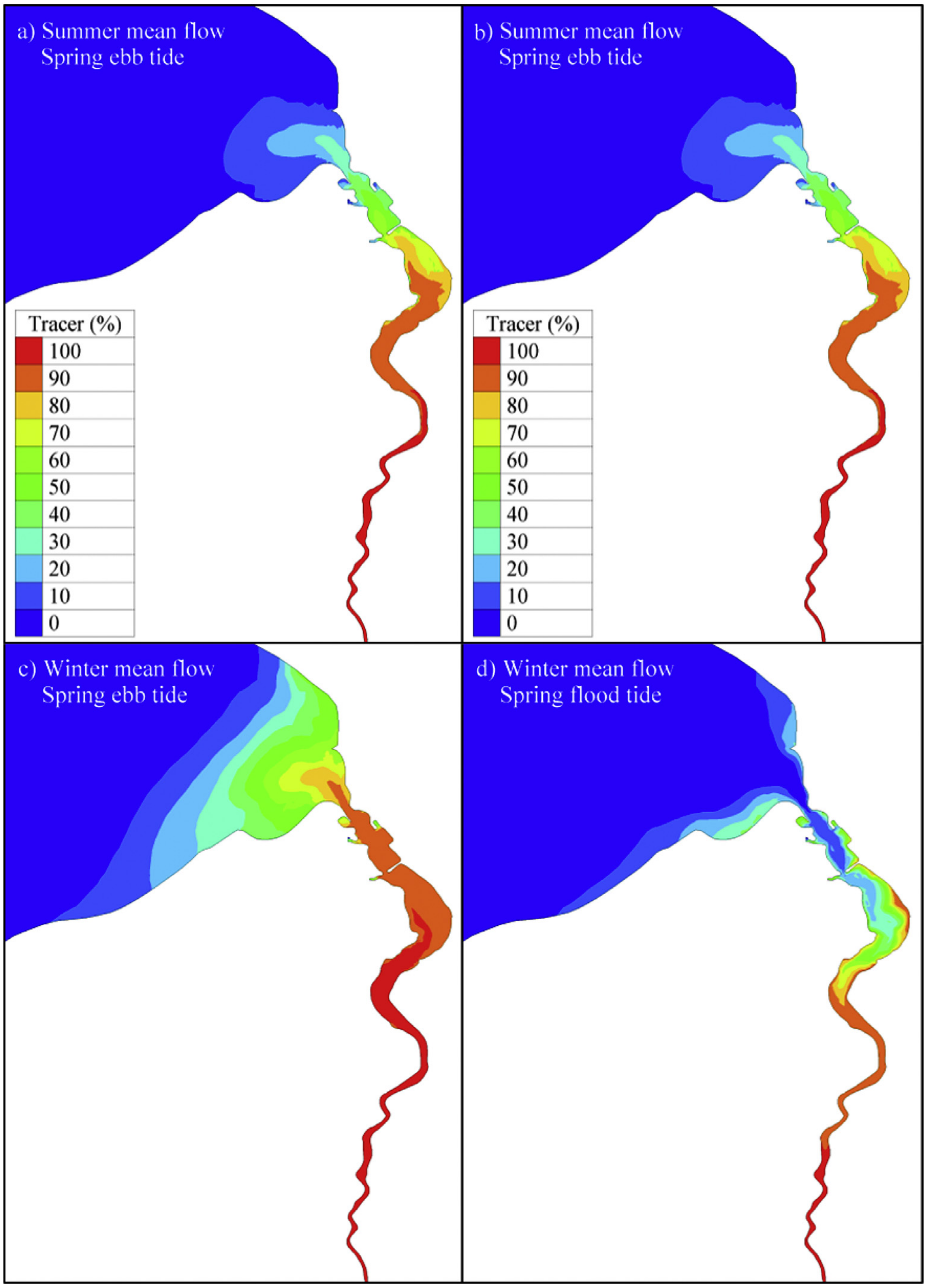

Fig. 7. The Conwy Estuary: Contour maps showing simulated percentage concentrations of a continuous-sourced tracer (input of 20 mg l^{-1} at the river boundary). Maps show concentrations for present-day sea levels with summer mean river flow (a, b) and winter mean river flow (c, d), during spring ebb flow and spring flood flow, respectively.

consider the effects to estuarine water quality. Loss of intertidal regions – important ecosystems in their own right – will also reduce carbon storage and nutrient retention (Jarvie et al., 2012). Beyond uncertainty within future shoreline management plans, if sea levels rise substantially over the 21st century, as predicted by models based on parametric relationships between sea level and air temperature (e.g. Rahmstorf, 2007; Grinsted et al., 2009; Jevrejeva et al., 2010), the estuarine longitudinal salinity structure will be altered significantly. Specifically, the estuary will become more saline and the salinity maxima will migrate upstream. In turn, our simulations suggest that sea-level rise will promote estuarine solute trapping and reduce their offshore transport due to the upstream migration of the salinity maximum – which could have far reaching consequences.

We show that the longitudinal salinity structure in the Conwy is sensitive to changes in seasonally *low* river flow but less sensitive to changes in seasonally *high* river flow. Consequently, variability in the salinity structure will be greater during summer months than winter months, implying that there is also more variability in the transport of conservative solutes and pollutants during summer. If the UK experiences drier summers and wetter winters in the future, as predicted (Jenkins et al., 2009; Christierson et al., 2012), seasonal

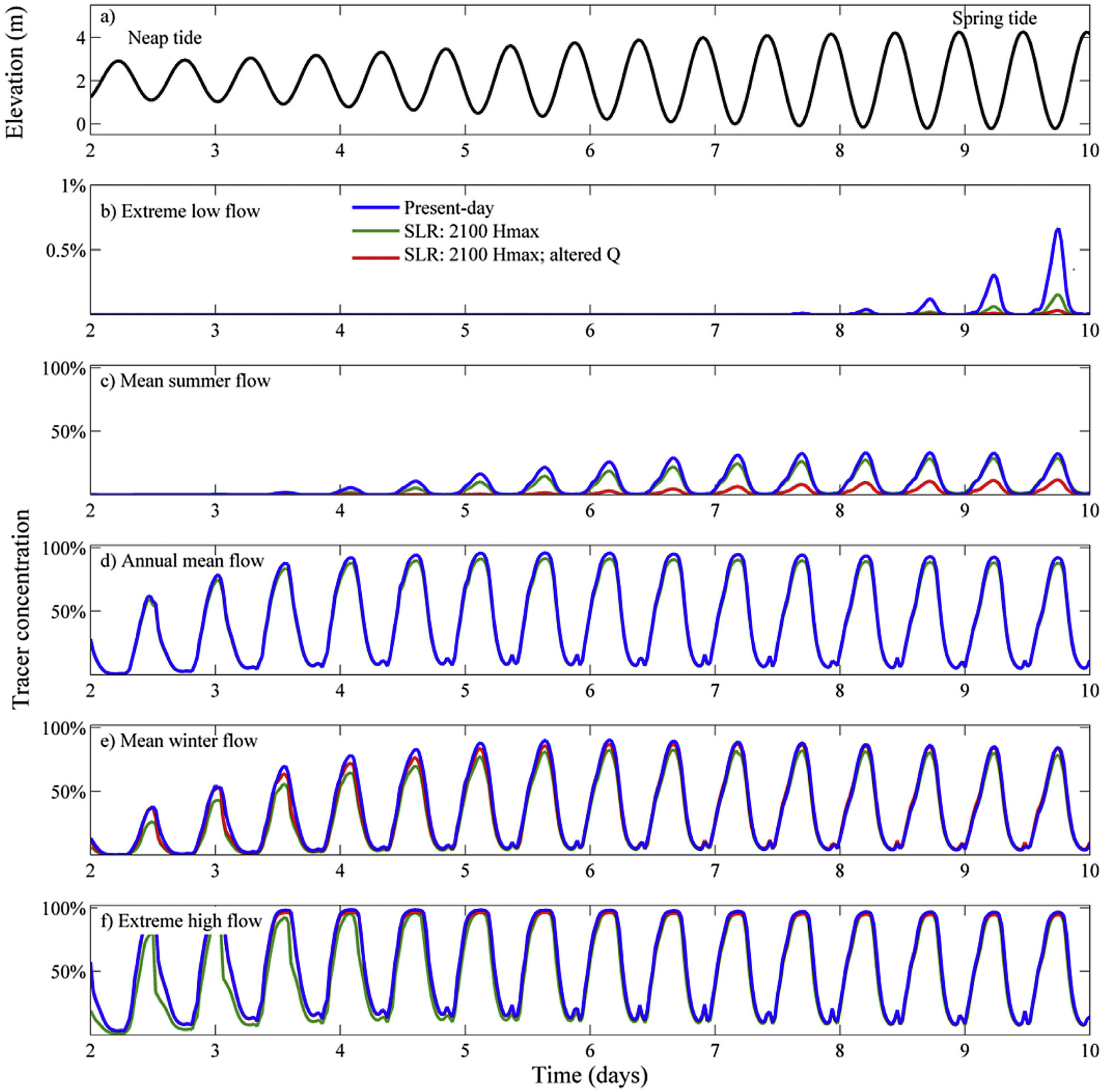

Fig. 8. (a) Surface elevations at the mouth of the Conwy Estuary during a typical period from neap to spring tide. Corresponding percentage concentrations of a continuous river-sourced tracer for different sea-level rise (SLR) scenarios are plotted during (b) extreme low flow, (c) mean summer flow, (d) annual mean flow, (e) mean winter flow, and (f) extreme high flow.

variability in estuarine salinities and conservative solutes will become more profound; estuarine waters becoming more saline in summer and more brackish in winter. In terms of nutrient cycling, for example, Jarvie et al. (2012) argue that stream concentrations of phosphorous would increase during both winter storms and summer conditions – especially for rapid-response catchments. Potentially longer periods of low river flow, or drought, appear to have the most important impact on estuarine functioning for short-catchment estuaries, causing more saline conditions and greater estuarine trapping. Within Europe, these results will affect compliance with the Water Framework directive (2000/60/EC); hence, effective management schemes will need to be in place, including the planning/modelling of future coastal flood defences within estuaries (e.g. Robins et al. 2011).

Certain assumptions have been made in our research, which should be taken into account in our interpretations and be considered for future studies in this topic. For instance, it is unlikely that the constant-flowrate simulations produced here would actually occur for a period of 10 days, especially for the extreme high flow scenario (the drought in July 1894, which produced the minimum flowrate of 0.32 m^3 s^{-1}, did in fact last for ~10 days). Nevertheless, to investigate the implications of our experimental design, we have compared our extreme low/high flowrate simulations (RUN-1.1/RUN-1.5) to simulations forced by observed hydrographs at the times of minimum/maximum flowrates (Fig. A6). Further, to investigate the estuarine response to a rainfall event during low flow conditions, a semi-idealised scenario was also simulated whereby a ‘rainfall event’ (i.e., 5 m^3 s^{-1} flowrate) was incorporated for the initial 2 days of the drought hydrograph (Fig. A6a). As one would expect, the hydrograph simulations do produce different estuarine structures to the constant-flowrate simulations, represented by Figs. A6c and A6d. Yet for the Conwy, the estuarine response to freshwater inflow is relatively quick – of the order of several days or less – and, therefore, the estuary is in a quasi-steady state. Consequently, for any given time, our idealised constant-flow simulations do broadly represent the same conditions as would occur from a time-varying hydrograph simulation. There is no way of quantifying river flow time-series due to aleatory uncertainty, hence for systems which respond slower to rainfall events, future work needs to determine potential idealised hydrographs of extreme events, in a similar way to recent studies of storm surge variability (e.g. Quinn et al., 2014).

The Conwy Estuary generates significant secondary flows which require three-dimensional modelling to resolve. A transverse (cross-channel) salinity gradient develops on the flood tide due to lateral velocity shear (Turrel et al., 1996) which induces secondary flows associated with an axial convergent front (Nunes and Simpson, 1985; Robins et al., 2012); a feature that occurs in many UK estuaries (Brown et al., 1991). Since a much weaker divergent front develops on the ebb tide, a net upstream transport of surface material (e.g. flotsam and detritus) is observed under normal riverine conditions (Brown et al., 1991). Estuaries with axial convergence fronts, therefore, can be expected to promote trapping of substances compared with estuaries without such secondary flows. The geometry of the model is based on data collected in 2003; however, bathymetric change over the ±10 years when model validation occurred is likely to be negligible compared to our model assumptions. For instance, we do not simulate storm surge in our model, which could add up to a metre of water level, thus our bathymetry only needs to be accurate to the nearest ~1 m. Further, the geometry of the tidally-influenced river channel is less likely to have changed significantly over this period because this region is subjected to weaker tidal processes. Nevertheless, our 2D model does in fact accurately simulate the longitudinal salinity distribution, confirmed thorough model validation against several in-situ surveys, and comparisons with a 3D simulation verify that the depth-averaged model is appropriate for the study.

Freshwater input to the Conwy, downstream of Llanwrst, was neglected in our simulations. This omission could be significant for accurate simulations and potentially account for our over-prediction of salinities in 1986 in the lower estuary (Fig. 3); however, our sensitivity results are unlikely to be affected since the salt/solute distribution is less sensitive to high flow conditions when the neglected freshwater inflow is large. Furthermore, it is unlikely that substantial groundwater discharge has been omitted since our model is generally well validated. That being said, if data is available, all freshwater input to the estuary should be incorporated, especially for larger catchments and for future climates with increased flow variability. Future work could also consider interannual-interdecadal variability and its influence to estuarine processes.

We have not attempted to simulate sediment transport, non-conservative nutrient transport, or the interactions between sediments and nutrients such as the aggregation/disaggregation (flocculation) of particles. Yet, more research on how non-conservative nutrients behave in the water system and how they interact with sediments is required, in order to address this complex topic. Estuarine nutrient trapping is potentially increased through sediment uptake on tidal flats (Sakamaki et al., 2006); however, Jickells et al. (2000) showed that if sea-level rise increases tidal inundation on tidal flats, reducing their coverage, nutrient trapping could be reduced and counteract nutrient retention. If sediment accumulation keeps track with sea-level rise, intertidal regions may be sustained due to managed realignment. Indeed, the re-introduction of saltmarshes has been said to be important for nutrient cycling and nutrient retention in the future (Jickells et al., 2000). However, considering our results presented here, if coastlines are maintained and intertidal wetlands are reduced, then the risk of coastal pollution will be increased.

5. Conclusions

By developing a high resolution (~15 m), depth-averaged, model of a real macrotidal UK estuary, we were able to reproduce the longitudinal circulation and salinity structure with some degree of accuracy (for example, root mean squared errors in salinity between modelled and observational data, over three separate lunar cycles, were <5). Through several simulations of idealised conditions representing the present-day and climate change, we have isolated the physical controls on the estuarine salt balance and continuous river-sourced solute transport; acting as a proxy for the transport of conservative nutrients or pollutants. We therefore demonstrate a powerful tool that should be part of an integrated approach to effectively understanding and managing the impacts of macronutrient cycles.

Salinity distribution was most sensitive to low river flow conditions - droughts. But for high river flow, salinities were mainly determined by tidal modulation alone and the estuary was more brackish. The fate, therefore, of nutrients from the catchment is dependent on the combination of these opposing forces; offshore transport only occurring during ebb tidal flow, and in greater proportions during spring tides and high river flow than during neap tides and low river flow.

Considering sea-level rise, the overall salinity of the estuary and retention of trace materials will increase slightly, but variability will remain controlled by river flow and tidal mixing. However, sea-level rise beyond predicted levels will extenuate this process and become an important control on the system provided the estuary shape remains unaltered. Predictions of drier summers and wetter winters in the UK will have greatest impact on estuarine functioning during summer, reducing the proportion of solute transport offshore – therefore the greatest impact of climate change to UK estuaries will be variations in summer rainfall climatology.

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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.2014.09.019.

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