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1. **Global coastal wetland change under sea-level rise and related**
2. **stresses: the DIVA Wetland Change Model**

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34 **Abstract**

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1. The Dynamic Interactive Vulnerability Assessment Wetland Change Model (DIVA\_WCM) comprises
2. a dataset of contemporary global coastal wetland stocks (estimated at 756 x103 km2 (in 2011)),
3. mapped to a one-dimensional global database, and a model of the macro-scale controls on wetland
4. response to sea-level rise. Three key drivers of wetland response to sea-level rise are considered: 1)
5. rate of sea-level rise relative to tidal range; 2) lateral accommodation space; and 3) sediment
6. supply. The model is tuned by expert knowledge, parameterised with quantitative data where
7. possible, and validated against mapping associated with two large-scale mangrove and saltmarsh
8. vulnerability studies. It is applied across 12,148 coastal segments (mean length 85 km) to the year
9. 2100. The model provides better-informed macro-scale projections of likely patterns of future
10. coastal wetland losses across a range of sea-level rise scenarios and varying assumptions about the
11. construction of coastal dikes to prevent sea flooding (as dikes limit lateral accommodation space
12. and cause coastal squeeze). With 50 cm of sea-level rise by 2100, the model predicts a loss of 46 –
13. 59% of global coastal wetland stocks. A global coastal wetland loss of 78% is estimated under high
14. sea-level rise (110 cm by 2100) accompanied by maximum dike construction. The primary driver for
15. high vulnerability of coastal wetlands to sea-level rise is coastal squeeze, a consequence of long-
16. term coastal protection strategies. Under low sea-level rise (29 cm by 2100) losses do not exceed
17. ca. 50% of the total stock, even for the same adverse dike construction assumptions. The model
18. results confirm that the widespread paradigm that wetlands subject to a micro-tidal regime are
19. likely to be more vulnerable to loss than macro-tidal environments. Countering these potential
20. losses will require both climate mitigation (a global response) to minimise sea-level rise and
21. maximisation of accommodation space and sediment supply (a regional response) on low-lying
22. coasts.

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1. *Keywords:* tidal wetlands, wetland vulnerability, wetland transitions, wetland loss, accommodation
2. space, sea-level rise

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# 1. Introduction

1. Millennial, centennial and decadal records of changing patterns of coastal wetlands,
2. including mangrove forests, saltmarshes, mudflats and associated habitats, show that they are
3. particularly sensitive to environmental change (e.g. Morris et al., 2002; French, 2006; Fitzgerald et
4. al., 2008; Mudd et al., 2009). More recent system changes also reflect the impacts of human
5. activities superimposed on these natural dynamics, such as drainage and conversion to agriculture
6. (e.g. Gedan et al., 2009) and aquaculture (e.g. Murdiyarso et al., 2015). There is concern, therefore,
7. as to how near-future global environmental change will further modify these systems **(**e.g. Alongi,
8. 2008; Kirwan et al., 2010; Fagherazzi et al., 2012).
9. On contemporary timescales, tidal wetlands are biologically productive ecosystems of high
10. biodiversity supplying multiple ecosystem services. At the same time they are subject to significant,
11. and accelerating, rates of global coastal wetland loss due to natural and anthropogenic drivers (e.g.
12. Adam, 2002; Millennium Ecosystem Assessment, 2005; Barbier et al., 2011; Nicholls et al., 2011).
13. Davidson (2014) estimates that natural coastal wetlands have declined by 46–50% since the
14. beginning of the 18th century and by 62–63% over the course of the 20th century and Leadley et
15. al.’s (2014) Wetland Global Extent Index estimates an almost 50% decline between 1970 and 2008.
16. Ecosystem services and loss rates have become linked over the last decade with the
17. recognition of the role of low-lying wetlands in natural coastal protection (e.g. Shepard et al., 2012),
18. following the interactions between mangrove ecosystems and the wave fields of the 2004 Asian
19. tsunami (e.g. McIvor et al., 2012) and between coastal marshes and 2005 Hurricanes Katrina and
20. Rita on the Gulf coast, USA (e.g. Barbier et al., 2013) amongst others. Much remains to be done,
21. however, on identifying the exact linkages between mosaics of coastal habitat area and habitat
22. fragmentation and the maintenance of a coastal protection function (e.g. Barbier et al., 2008; Koch
23. et al., 2009; Loder et al., 2009; Gedan et al., 2011). Furthermore, these debates are embedded in a
24. context where the knowledge of the general spatial distribution of coastal wetland ecosystems is
25. currently poor, particularly for saltmarshes (e.g. Rebelo et al., 2009, Saintilan et al., 2009; Chmura,
26. 2011). There are serious gaps in the information base and much of the data that has been collected
27. has come from different sources and different time periods and at a range of scales (Friess and
28. Webb, 2014). Indeed, Friess et al. (2012) goes as far as to argue that the under-reporting of
29. saltmarsh from the tropics underpins the presumption that mangrove replaces saltmarsh in the
30. tropical intertidal zone. These shortcomings have hampered the assessment of the extent and
31. condition of wetlands and proper estimations of the rate of loss. Thus one review concludes ‘a
32. number of prognostications have been made regarding the future of the world’s mangrove forests
33. in the face of climate change with local, regional, and global forecasts ranging from extinction to no
34. or little change in areal coverage’ (Alongi, 2008, 8).
35. Accelerated sea-level rise is a major threat to wetland futures at regional to global scales.

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However, most detailed studies on wetland vulnerability to accelerated sea-level rise have been

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over small spatial scales and short timescales and most concentrate on the likelihood of vertical

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drowning (Webb et al., 2013), when sediment accumulation on the platform cannot keep vertical

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pace with sea-level rise. There has been less emphasis on rates of horizontal retreat, associated

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with wave-induced marsh boundary erosion (Mariotti and Carr, 2014). Thus, for example, the

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Surface Elevation Table – Marker Horizon (SET-MH) methodology has the necessary precision to

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allow annual surface elevation change to be related to annual rates of sea level change (Cahoon et

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al., 2002) although inter-site and inter-annual variations in surface response characteristics are high

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(Cahoon et al., 2006). Historically, the SET-MH global network of sites has been patchy and not

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focussed on those areas where wetland loss rates are thought to be particularly high (Webb et al.,

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2013). However, there has been an expansion of sites globally in the last few years and it is

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becoming possible to use this network to model larger scale patterns in wetland vulnerability, as

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has been shown for Indo-Pacific mangrove SET sites (Lovelock et al., 2015). Furthermore, SET

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datasets have been used as calibration datasets in other models of wetland change, most notably

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the SLAMM model; we return to this usage below.

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The generic problems of large-scale analysis have been addressed in part by the

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development of macro-scale landscape models. These models vary in structure, complexity and the

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ease with which they can be applied. The more sophisticated landscape models use geomorphic

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and hydrologic sub-models to distribute fluxes of water, sediments and nutrients across a raster

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grid (e.g. CELSS model: Sklar et al., 1985) to calculate likely changes in wetland type extent.

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However, the data and computational requirements of such an approach largely preclude its

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application as a broad-scale tool for wetland analysis (Martin et al., 2002; Reyes, 2009; Couvillion

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and Beck, 2013). Simpler models, such as cellular automata (Ross et al., 2009), capture the key

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characteristics of wetland dynamics empirically, require fewer data, and are easily applied, but the

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ability to deal with low frequency, high magnitude impacts and the recognition of the interaction

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and feedback of geo-morphological and ecological processes are missing (Kirwan and

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Guntenspergen, 2009). Nevertheless, these approaches are useful for calibration purposes, as we

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demonstrate below. Of this suite of large-scale models, the one that has been most widely applied

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is the ‘Sea Level Affecting Marshes Model’ (SLAMM) (Clough et al., 2010). SLAMM is open source

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and has a user-friendly interface for implementation; is based on empirical calculations so that

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computation times are substantially less than those required for complex numerical models; and

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implementation has low data demands.

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The pioneering Global Vulnerability Assessment (GVA), and its subsequent revision, is a

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macro-scale model which provided the first worldwide estimates of the impacts of accelerated sea-

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level rise on coastal systems (Hoozemans et al., 1993; Nicholls et al., 1999). This included a first-

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order perspective on coastal wetland loss. Subsequently, the data on coastal wetland stocks has

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improved (e.g. Vafeidis et al., 2008; Spalding et al., 2010; Giri et al., 2011), and the understanding of

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the main drivers of change, including sea-level rise, has increased (Nicholls, 2004; McFadden et al.,

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2007; Nicholls et al., 2007)**.** Hence, a re-evaluation of these earlier assessments of wetland

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vulnerability is timely. This paper discusses the further development and application of a broad-

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scale wetland change model: the Dynamic Interactive Vulnerability Assessment Wetland Change

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Model (DIVA\_WCM), originally developed within, and subsequently to, the European Community

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DINAS-COAST Project. In this paper we show how a newly constituted database of contemporary

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global coastal wetland extent can be linked to a revised conceptual model of the controls on

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wetland health and resilience. In comparison to its previous version (McFadden et al., 2007), the

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revised model has been parameterised with quantitative data where possible, calibrated by SLAMM

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and other model outputs and validated by expert knowledge, including map-based approaches.

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Thus, in its current form the model provides better-informed macro-scale projections of likely

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future wetland extents than have been available previously.

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# Methods

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*2.1 The DIVA modelling framework*

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DIVA is an integrated, global modelling framework of coastal systems that assesses

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biophysical and socio-economic consequences of sea-level rise and socio-economic development,

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taking into account coastal erosion, coastal flooding, wetland change and salinity intrusion

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(http://www.diva-model.net; Hinkel, 2005; Hinkel and Klein, 2009; Hinkel et al., 2010, 2013, 2014).

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The DIVA data modelling framework divides the world’s coast (excluding Antarctica) into 12,148

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variable length coastal segments (mean length: 85 km; range: 0.009 km to 5,213 km) and associates

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up to 100 data values with each segment (Vafeidis et al., 2008). Each segment represents a

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relatively homogenous unit based on geomorphology, population density and administrative

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boundaries; there are a greater number of segments in the more populated areas. Only the DIVA

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data associated with wetland change are considered in this paper.

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DIVA is driven by climate and socio-economic scenarios. Using the HadGEM2-ES Earth

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System model from Phase 5 of the Coupled Model Intercomparison Project (CMIP5), three sea-level

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rise scenarios have been investigated in this paper, representing a subset of the scenarios described

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by Hinkel et al. (2014). These scenarios consider three Representative Concentration Pathways (RCP)

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- RCP2.6, RCP4.5, and RCP8.5. The RCPs correspond to different levels of greenhouse gas

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concentration trajectories, ranging from a world of strong climate mitigation to one of increasing

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emissions. A major uncertainty in projecting future sea-level rise is the contribution of land-based

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ice. In Hinkel et al. (2014), each RCP scenario is associated with three levels of ice melt (low, median

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and high) to create a ‘very likely’ range. The scenarios represent patterns of change (representing

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thermal expansion and changes in ocean circulation, plus gravitational changes from ice sheets (the

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contribution from ice caps is assumed to be uniform)) where some parts of the world have higher

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or lower sea-level rise compared with the global mean. Projected global mean sea-level rise to the

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year 2100 with respect to 1995 (mean sea level during 1985-2005 baseline period) for each of the

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scenarios is given in Table 1 (median values, with 5% and 95% quantiles in parentheses; after Hinkel

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et al., 2014). In this demonstration paper, we cover the widest range of sea-level rise scenarios,

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from the lowest (5%) quantile of RCP2.6 (29 cm by 2100), through the median rate of sea-level rise

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for RCP4.5 (50 cm), to the 95% quantile of RCP8.5 (110 cm). Finally, the sea-level rise scenarios are

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downscaled to each DIVA database segment, including local land level change, following Peltier

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(2000a, 2000b), and a 2 mm a-1 subsidence in deltas, reflecting natural sediment compaction.

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Hence relative sea-level change varies from DIVA segment to segment.

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For socio-economic scenarios, the Shared Socioeconomic Pathways (SSPs) are used. DIVA

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considers population and gross domestic product (GDP) growth from the SSP2 scenario which sees

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the trends typical of recent decades continuing, with moderate global population growth, some

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progress toward achieving development goals and a slow decrease in the world’s dependency on

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fossil fuels (IIASA, 2012). The socio-economic scenarios can influence the construction of dikes, and

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hence the availability of accommodation space for wetlands, as explained below.

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*Table 1 near here*

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*2.2. The Wetland Change Model (DIVA\_WCM)*

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The DIVA\_WCM is one module in the DIVA modelling framework (Version 5.1

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([http://www.diva-model.net/)).](http://www.diva-model.net/))) DIVA\_WCM comprises i) a newly constituted global database of

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coastal wetlands built on the basis of the original DIVA database (Vafeidis et al., 2008) and ii) an

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impacts algorithm for coastal wetlands. It improves upon the existing DIVA-WCM (McFadden et al.

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2007) by extensive new parameterisation and calibration.

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The coastal wetland database employed in this paper was derived, under licence, from

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datasets held by the United Nations Environment Programme’s World Conservation Monitoring

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Centre (UNEP-WCMC), the specialist biodiversity assessment arm of UNEP. A global layer of

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mangrove forest data, revised after Giri et al. (2011), was augmented with a recently improved

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saltmarsh data layer; both layers were imported into the DIVA database and assigned to the

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appropriate coastal segment. Sub-sets of the data were checked to ensure that there was no

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corruption of data in the transfer from the original files to the database. In the database, six

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wetland types associated with coastline segments are considered: 1) coastal forest; 2) mangrove

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forest; 3) freshwater marsh (of limited extent); 4) saltmarsh; and unvegetated sediments, which are

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divided into 5) sabka and saline tidal flats (of very limited extent); and 6) mudflat/sand flat. The

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distinction between these latter two types is based on climatic setting and elevation (derived from

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the ETOPO2 (NGDC, 2001) dataset): type 5 is above (‘unvegetated high’) and type 6 is below

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(‘unvegetated low’) Mean High Water Springs (MHWS), respectively. Data on unvegetated

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sediments, freshwater marshes and coastal forested originate from the above mentioned GVA

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(Hoozeman et al., 1993).

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French (2006, 120) states that ‘… the existence and ecological function of tidally-dominated

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saltmarshes are ultimately contingent upon the operation of hydrodynamic and sedimentary

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processes within constraints imposed by the intertidal accommodation space and the sediment

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supply’. The DIVA\_WCM algorithm considers three key drivers that control wetland response to

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sea-level rise: 1) local sea-level rise relative to tidal range; 2) lateral accommodation space; and 3)

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sediment supply, following McFadden et al. (2007). A score of one to five is adopted to represent

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present and future forcing levels for each of these drivers: one corresponds to the lowest forcing

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and *vice versa*.

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Wetlands respond to sea-level rise over different time horizons. Mangroves and coastal

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forests respond more robustly to environmental change than the other wetland types because

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slower growth rates across a wide range of environmental tolerance allows for survival under

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moderate levels of stress (Ball, 1988). This is represented in the model by different response times

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for these wetland types. Macro-scale landscape models, and specifically output derived from the

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WARMER and SLAMM model applications, were used to characterise wetland resilience to sea-level

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rise and other stresses where possible. Expert judgment, from peer-reviewed literature and

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research project reports, was applied where necessary.

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Figure 1 outlines the primary components of the DIVA\_WCM, comprising four sets of

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calculations:

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1. Assessment of the vulnerability of all wetlands to sea-level rise. This vulnerability score (*VS*)

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depends upon sea-level rise, lateral accommodation space and sediment supply and is

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universal. As the score is calculated segment-by-segment, it is described as the Coastal

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Segment Vulnerability Score (*CSVS*);

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1. Conversion of the Coastal Segment Vulnerability Score into a wetland-specific Ecological

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Sensitivity Score (*ESS*);

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1. Calculation of the proportion of wetland loss/change that is expected for each wetland type

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based on the ESS score; and

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1. Calculation of wetland habitat successional changes and wetland loss to open water,

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generating new wetland areas, through a habitat translation model.

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*Figure 1 near here*

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It is important to note that the DIVA-WCM is a wetland loss model for sea-level rise; we

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acknowledge that near-future environmental change may also produce new areas of wetland in

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particular landscape settings, but the generation of new wetland is not considered further in this

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paper. The following sections detail each of these methodological steps. Further information is

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given in the Supplementary Material.

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* + 1. *Relative sea-level rise / tidal range*

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When relative sea-level rise is sudden and of high magnitude, as might result from tectonic

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activity, the wetland surface may be abruptly submerged (e.g. Atwater, 1987). More frequently,

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and the subject of concern here, coastal wetlands are subjected to slow, progressive relative sea-

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level rise caused by the combination of eustatic factors and regional to local subsidence. This

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process is reflected in the changing pattern of tidal submergence, or hydroperiod (Reed, 1995). If

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sediment supply is sufficient, marsh surfaces will accrete vertically, rapidly at first but then slowing

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over time as fewer tides inundate the progressively higher surface (Allen, 1990). Conversely, in

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sediment-poor wetlands, subject to a rise in relative sea level without equal increases in wetland

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surface elevation from sediment accretion, the duration and depth of tidal flooding will increase

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over time. In this situation, wetland vegetation may revert to a community composition more

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typical of lower elevations in the tidal frame (Huiskes, 1990; Mendelssohn and Morris, 2000).

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No clear relationships have been found at the large-scale between accretion rates and tidal

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range (Allen, 1990; French and Reed, 2001; Cahoon et al., 2006; Mossman et al., 2012). However,

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Stevenson et al. (1986) showed accretion deficits (rate of sea-level rise minus rate of near-surface

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accretion) to be greater in low tidal range saltmarshes than in higher tidal range marshes along the

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eastern seaboard of the USA. This has been attributed to the expanded intertidal range that can be

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occupied by vegetation (e.g. Day et al., 1995) and the increased flood-dominance, and thus

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enhanced sediment supply (e.g. Friedrichs and Perry, 2001), of macrotidal (> 4 m spring tidal range)

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marsh systems compared to meso-tidal (2-4 m tidal range) or micro-tidal (< 2 m tidal range)

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systems (Kirwan et al., 2010; Fagherazzi et al., 2012). Furthermore, in micro-tidal settings the

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expansion of the tidal prism on sea-level rise is disproportionately large, with increases in tidal

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channel geometries leading to loss of wetland area (Kirwan and Guntenspergen, 2010).

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The first environmental forcing factor captures this process through the dimensionless relative

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sea-level rise term *rslr\_d*, where the annual relative rise in sea level (RSLR) is scaled by the segment-

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specific tidal range (Equation 1 in Supplementary Material; and see Table A1 for details of the tidal

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range parameter). Unlike earlier applications of this approach (Nicholls et al., 1999), the

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dimensionless RSLR is described as a power function with an exponent of 1.4, based on the

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literature review described above and expert judgement. The scoring of *rslr\_d* is based on fixed

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class boundaries that are initialized before simulation. Assuming a current 3 mm a-1 global mean

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sea-level rise rate (after Church et al., 2013), we subtract the segment-specific uplift (obtained from

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[http://www.atmosp.physics.utoronto.ca/~peltier/data.php),](http://www.atmosp.physics.utoronto.ca/%7Epeltier/data.php)) and calculate the 95th, 84th, 50th, and

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16th percentiles of the cumulative distribution of the resulting *rslr\_d* parameter to derive the class

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boundaries of *rslr\_tidal\_score*, while only considering segments where wetlands are present (for

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exact values see Supplementary Material, Table A2). During simulation, the *rslr\_d* forcing factor is

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updated and scored at every time step and, hence, is driven by the associated sea-level rise

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scenario (see section 2.1 above and Table 1).

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* + 1. *Lateral accommodation space*

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The notion of ‘accommodation space’ comprises two components defined by sea-level rise,

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namely vertical wetland surface adjustment upwards and lateral habitat migration landwards

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(Phillips, 1986; Allen, 1990). The characterisation of lateral accommodation space within the

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DIVA\_WCM is built on an assessment of the impact of two controlling factors: i) coastal slope

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(Brinson et al., 1995); and ii) the presence or absence of dikes, which limits lateral accommodation

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space (Feagin et al., 2010). Lateral accommodation space, *aspace,* is calculated recursively in the

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model. The *aspace* value is initialised for each segment using the average topographic slope,

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derived from the ETOPO2 (NGDC, 2001) dataset. Model categorisation of coastal slope, and

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associated forcing scores, are given in the Supplementary Material Table A3. This initialized *aspace*

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score is then updated based on the estimated sea-dike height at each time-step. If appropriate,

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*aspace* is increased by 0.25 at each time step until the highest forcing score is obtained (Equation 2,

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Supplementary Material). Thus, the loss of lateral accommodation space is a progressive process in

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terms of stressing wetlands, being maintained at the highest vulnerability for the remainder of the

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model run once *aspace* reaches the maximum score of five. Importantly, the model does not

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simulate the impact of creating new lateral accommodation space.

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The DIVA model considers dike construction, and dike upgrading, as an adaptation response

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to coastal flooding (Hinkel et al., 2014). Three scenarios for dike construction are evaluated in this

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paper. Two bounding cases of ‘no dikes’ (in which no coastal floodplains are protected) and

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‘maximum dikes’ (in which all coastal floodplains are protected) are considered. In addition, dikes

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built according to a ‘demand-for-safety’ function, assuming the SSP2 socio-economic scenario

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(Hinkel et al., 2014), are also evaluated and termed ‘widespread dikes’. The driver *aspace* is not

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influenced by dikes under ‘no dikes’; is most affected under ‘maximum dikes’; and is subject to an

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intermediate effect under ‘widespread dikes’.

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* + 1. *Sediment supply*

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It has been strongly argued that sediment starvation at the coast, associated with the human

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management of river courses, deltas and erodible coastal cliffs, has had profound consequences for

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the maintenance of coastal sediment systems (e.g. Syvitski et al., 2005, 2009; Stralberg et al., 2011).

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The DIVA\_WCM also characterises the ability of a wetland to keep pace with relative sea-level rise

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through a third parameter, sediment supply *sedsup*. Following Stevenson et al. (1986), a widely

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adopted methodological approach has been to compare the rate of vertical accretion to relative

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sea-level rise and thus to calculate a wetland accretionary surplus or deficit. Such an approach

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assumes that accretion is equal to wetland surface elevation change. This is now known to be a

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simplification, as the relative balance between the *in situ* accumulation of organic sediments

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(Cahoon and Reed, 1995; Middleton and McKee, 2001; Rooth et al., 2003) or external, inorganic

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inputs (French and Spencer, 1993; Christiansen et al., 2000), or a combination of the two (Saintilan

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et al., 2013), can affect this balance, as can subsurface processes occurring within the soil column,

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including compaction, plant growth-decomposition and shrink-swell behaviour related to varying

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water storage (Cahoon et al., 2011; Krauss et al., 2014). However, given the scale of the

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DIVA\_WCM, these relationships must be simplified for the purposes of modelling, with a distinction

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between those settings and environmental histories that promote high sediment supply and those

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that favour low sediment supply. The model therefore considers a combination of six contextual

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physical and anthropogenic controlling factors – (1) tectonic context; (2) fluvial sediment inputs to

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the coastal zone; (3) sediment availability from Quaternary glacial sediments; (4) coastal

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geomorphic setting; (5) degree of coastal protection structures; and (6) timing of sediment supply

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from historical land use practices – in assessing the impact of varying sediment supply on wetland

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vulnerability. Each of these factors exhibits a range of values identified by a range in ‘forcing score’

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across different categories (see Supplementary Material, Table A4). Furthermore, each factor is

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multiplied by an internal weighting to reflect their relative significance within the *sedsup* parameter

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(Equation 3, Supplementary Material). The respective weights are based on expert judgement,

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derived from field experience and the published literature. It is recognised that sediment supply is

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the most difficult forcing factor to understand and parameterise in the model, due both to its

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localized and highly variable nature and to the lack of wetland datasets that specifically estimate

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this parameter.

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* 1. *Coastal Segment Vulnerability Score (CSVS)*

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The Coastal Segment Vulnerability Score (*CSVS*) reflects the integrated response of a

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wetland to relative sea level rise / tidal range, lateral accommodation space and sediment supply.

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The influence of each of these parameters is reflected through the weighted sum of the forcing

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factors, with the following weights: 0.5 for *rslr d*; 0.2 for *aspace*; and 0.3 for *sedsup* (and see

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Equation 4, Supplementary Material). These relative weightings indicate the importance of each

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parameter at the macro-scale, the values being derived from expert judgement, in turn based on

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field experience and published references. For gaining a better understanding of how these weights

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influence the model results, we performed a sensitivity analysis, comparing the model output of a

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series of different weight combinations.

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* 1. *Ecological Sensitivity Score (ESS)*

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Ecological systems are characterised by varying reaction and relaxation times to

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environmental perturbation. Many saltmarsh herbs, shrubs and grasses are very sensitive to

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landform change. Thus, for example, manipulative experiments in freshwater marsh systems,

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where inundation frequencies have been changed by transplanting marsh communities to lower (-

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10 cm) levels have shown responses in plant stem density and biomass over periods as short as a

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single growing season (McKee and Mendelssohn, 1989). By comparison, coastal forest trees show

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slower responses to changing environmental conditions. Thus, cypress forests on the Gulf of

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Mexico, USA have recorded 50% survival rates after 4 years of +120 cm and 18 years of + 60 – 300

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cm increases in water levels respectively. Modelling of bottomland forest succession is typically

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undertaken over 50 year timescales (summarised in Conner and Brody, 1989). However, resilience

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characteristics are quite different from herbs and shrubs, showing permanence (if not

364

regeneration) until a threshold point is reached when the system collapses catastrophically, as in

365

the case of hurricane-impacted mangrove forest (e.g. Cahoon et al., 2003). Once such a threshold

366

has been crossed, system re-establishment may be difficult and long-delayed (for review see

367

Spencer and Möller, 2013). Lag weights for current and previous 5-year time steps of a model run

368

were applied to parameterize these habitat-specific response lags to changes in the environmental

369

forcing factors (Table A5, Supplementary Material). For ‘freshwater marsh’, ‘salt marsh’,

370

‘unvegetated low’, and ‘unvegetated high’, a response time of 5 years was assumed and for ‘coastal

371

forest’ and ‘mangrove forest’, a response time of 10 years. The resulting modification of the Coastal

372

373

Segment Vulnerability Score (*CSVS*) is termed the Ecological Sensitivity Score (*ESS*) (Fig. 1).

374

* 1. *Habitat successional changes and wetland loss to open water*

375

Most existing large-scale models of wetland response to accelerated sea-level rise (e.g. the

376

GVA and its subsequent revisions (Nicholls et al., 1999)) assume the conversion of vegetated

377

surfaces to open water and thus simply generate statistics on total loss of wetland area. Such

378

models are most appropriate where local rates of relative sea-level rise are high, such as in

379

subsiding, sediment-starved deltaic environments. However, under more moderate rates of sea-

380

level rise, and with an adequate sediment supply, ecosystem change may be i) slower than

381

predicted; and ii) involve change stepped across wetland types rather than outright loss, as

382

ecological tolerances of particular plant communities are exceeded in turn. DIVA\_WCM assesses

383

both conversion to open water and transitions to other wetland types due to environmental change

384

through i) the construction of a series of wetland response curves, to define the proportion of

385

wetland expected to be lost; and ii) a model of wetland transitions, where losses are distributed

386

between different wetland types and open water.

387

During the first stage in the development of the habitat transition algorithm, a series of

388

habitat-specific response curves were estimated for total wetland loss as a function of Ecological

389

Sensitivity Score (ESS) (Fig. 1). These curves were approximated using the beta distribution (Fig. 2;

390

and see Equation 6, Supplementary Material) as this distribution can describe a wide range of

391

shapes within a constrained distribution (0% to 100% total wetland loss and 0.0 to 5.0 ESS value).

392

This is particularly useful for constructing habitat-specific response curves, reflecting different

393

resilience characteristics that were then calibrated using WARMER and SLAMM model outputs

394

(described in more detail below).

395

*Figure 2 near here*

396

Where there is wetland change or loss, Figure 3 outlines the model of transition that was

397

used within the DIVA\_WCM. Except in the case of coastal forest, with low to moderate

398

environmental forcing (CS*VS* value < 4) wetland types are transformed not only into open water

399

(50%) but also into other wetland types which are found lower in the tidal frame (50%), created as

400

a result of losses of wetland that occupy higher elevations. Thus, ‘low unvegetated’ wetland (i.e.

401

mudflat-sandflat) can be created where it did not previously exist within a geographical location.

402

Under high levels of environmental forcing (CS*VS* value >= 4) the model converts all wetland types

403

to open water.

404

405

*Figure 3 near here*

406

407

* 1. *Model calibration*

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In previous explorations of the DIVA\_WCM model structure (McFadden et al., 2007), the

409

model was calibrated qualitatively against model predictions of large-scale wetland type transitions

410

in the Barataria and Terrebonne sub-basins of the Mississippi Delta Plain (Reyes et al., 2000).

411

However, such calibration is problematic because these simulations begin in the period well before

412

the 1985 – 2005 baseline used in this study. More recent scenario modelling in the same region,

413

associated with Louisiana’s 2012 Coastal Master Plan (Couvillion et al., 2013) does provide detailed

414

forecasting of wetland loss on a sub-basin by sub-basin basis, but only until 2060. As an alternative

415

way forward in this study, therefore, DIVA\_WCM was calibrated against a set of six recent studies

416

of wetland change undertaken using the WARMER, and particularly, the SLAMM model (Table 2).

417

*Table 2 near here*

418

A number of difficulties were encountered in the inter-comparison of the DIVA\_WCM with

419

WARMER and SLAMM model outputs. Firstly, in many of the SLAMM studies it was difficult to

420

extract the necessary comparative information required on the sea-level rise scenario that had

421

been applied (i.e. starting year, end year, sea-level rise function used (such as SRES, RCP, linear

422

models), total sea-level rise within the simulation period (with reference to starting year)). Where

423

scenarios referred to a SRES-scenario (IPCC SRES, 2000), but with user-defined amplitude, these

424

were calculated, on a globally uniform basis, with the SRES-scenarios supplied with DIVA. Other

425

scenarios were identified as being driven by a linear rate of sea-level rise and constructed

426

accordingly. Secondly, the definition of wetland habitat types differed between the six calibration

427

studies and introduced some additional wetland types not present in the DIVA typology. It was thus

428

necessary to re-classify the wetland descriptions into their equivalent DIVA categories. Thirdly, the

429

effect of sea-dikes were included in SLAMM – DIVA\_WCM model comparisons where the SLAMM

430

studies made explicit reference that wetland loss had been affected by the presence of a dike.

431

However, it was not always very clear as to whether or not this had actually been the case. For

432

calibration, the DIVA\_WCM model was run against the relevant WARMER and SLAMM model

433

output for each of the sea-level rise scenarios reported in the respective study, including dikes

434

where applicable. The results were aggregated to allow inter-model comparison, the form of

435

aggregation depending on whether the WARMER / SLAMM study area integrated several DIVA

436

coastal segments or, alternatively, the one DIVA segment integrated several WARMER / SLAMM

437

sites. All loss rates are reported as percent total wetland loss per 5 years. The error measure used

438

to evaluate the model performance with reference to the reported data is the relative mean

439

difference (RMD). The ‘Nelder-Mead simplex direct search’ algorithm (Lagarias et al., 1998) was

440

applied to search for the RMD closest to zero by varying the habitat-specific beta values (Fig. 2,

441

Equation 6 in Supplementary Material). This exercise was conducted without constraints regarding

442

the relationships between the different beta values. The resilience/sensitivity estimation of a

443

specific habitat (Fig. 2) thus usefully emerged as a result of the calibration exercise. Running the

444

DIVA\_WCM with the optimized beta values (Equation 6, Supplementary Material) against the six

445

calibration studies produced a RMD value of 0.000227 with a mean difference of 0.00002% (loss per

446

5 years), indicating a close fit of model values with reported loss rates.

447

Fig. 4 shows the DIVA\_WCM model outputs against the SLAMM and WARMER calibration

448

studies. Interestingly, the WARMER model, which is not a derivative of SLAMM but a numerical

449

saltmarsh model that includes biophysical feedbacks, performed in a very similar manner to the

450

SLAMM models when it came to comparisons with the DIVA\_WCM model. Outliers relate to a small

451

number of particular DIVA segment comparisons across two SLAMM studies (Craft et al., 2009;

452

Geselbracht et al., 2011). It is clear that these examples of poor model fit are not related to

453

vegetated wetland habitats but rather to problems with estimating changes in ‘unvegetated low’

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habitat (i.e. mudflat/sandflat). This raises the need for an improvement in model formulation

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regarding the mechanisms in place when vegetated wetlands are drowning. Not considering the

456

site exposure to wave activity may partly explain the poor model representation of the

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‘unvegetated low’ at these sites. While this comparison illustrates the difficulty of model

458

calibration, these results are calibrated to these more detailed simulations, improving on the earlier

459

methods of Hoozemans et al. (1983), Nicholls et al. (1999) and McFadden et al. (2007). Further

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efforts at improved model calibration should receive high priority in future research efforts.

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463

*Figure 4 near here*

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465

466

* 1. *Model validation*

In order to undertake independent validation at a scale appropriate to the DIVA scale of

467

analysis, model outputs of the calibrated DIVA\_WCM were compared with two broad-scale coastal

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wetland vulnerability studies, one concerned with the modelled vulnerability of Indo-Pacific

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mangrove forests to sea-level rise and one a qualitative assessment of wetland stability along the

470

US mid-Atlantic coast.

471

Lovelock et al. (2015) developed a model (hereafter referred to as the ‘Lovelock model’) to

472

predict the time to submergence of mangrove ecosystems subject to accelerated sea level rise

473

based on the concept of the loss of ‘elevation capital’, the potential of a mangrove ecosystem to

474

remain within a suitable inundation regime (between Highest Astronomical Tide (HAT) and Mean

475

Sea Level (MSL)). The key controlling parameters are the rate of sea level rise, the tidal range and

476

suspended sediment supply. Sites with a tidal range of 10 m need to lose 5m of elevation capital to

477

bring them to the critical survival threshold of MSL whereas sites with a tidal range of 1 m only have

478

to lose 0.5 m of elevation to bring them to this threshold. Thus the Lovelock model predicts that

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sites with low tidal range are significantly more vulnerable to loss than those experiencing a high

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tidal range. Loss of elevation capital can be offset by elevation gains from vertical accretion as sea

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level rises. Thus mangrove forest sites with high sediment supply are less vulnerable to conversion

482

to open water than sites with low sediment supply. Total suspended matter in coastal waters was

483

acquired from remotely sensed imagery and converted to elevation gain through established

484

relationships between sea-level rise, suspended sediment concentrations and measured changes in

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surface elevation from SET sites. The Lovelock model excludes consideration of an accommodation

486

space term but it does identify the importance of the relations between rates of sea level rise, tidal

487

range and sediment supply in determining mangrove forest vulnerability to sea-level rise in a similar

488

manner to the DIVA\_WCM. The difference in model structure makes Lovelock et al. (2015) an

489

appropriate validation case for the DIVA\_WCM. The two models cannot be compared directly

490

because i) the Lovelock model provides a binary survival or loss indicator whereas the DIVA\_WCM

491

estimates percentage loss of mangrove forest over time; and ii) the Lovelock outputs are reported

492

on a 4 km resolution grid defined by the remotely sensed TSM data whereas the DIVA\_WCM results

493

are mapped onto the variable length DIVA coastal segments. However, a qualitative assessment is

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possible, for comparable sea level rise scenarios to 2100 (Fig. 5). The areas of mangrove

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submergence predicted by the Lovelock model (Fig. 5b) map well onto the areas of highest coastal

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wetland loss predicted by the DIVA\_WCM (Fig. 5a). Apart from Australia and Brunei, of the top ten

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areas of expected mangrove loss identified by the DIVA\_WCM in the region shown in Figure 5, eight

498

areas are also highlighted by the Lovelock model: Cambodia (55% coastal wetland loss at country

499

level by 2100 in the DIVA\_WCM); Philippines (50%); Sri Lanka (48%); Thailand (46%); Indonesia

500

501

(40%); Federated States of Micronesia (40%); Papua New Guinea (39%); and Solomon Islands (39%).

502

503

*Figure 5 near here*

504

A qualitative assessment of wetland stability on the eastern seaboard of the USA was

505

performed on behalf of the US Environmental Protection Agency (EPA) (Reed et al., 2008). As with

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the first validation exercise, a direct comparison between the EPA assessment and the DIVA\_WCM

507

output is not possible. This is partly because the DIVA\_WCM segments along the eastern seaboard

508

do not allow the level of disaggregation seen in the EPA assessment and partly because Reed et al.

509

(2008) rely on a more qualitative, expert judgement approach. Nevertheless, validation at the level

510

of aggregation associated with the DIVA\_WCM model was possible. Reed et al. (2008) assume a

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current rate of sea-level rise of 3 mm a-1 and provide estimates for future wetland development for

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three linear SLR scenarios: a continuation of the current rate of sea-level rise; the current rate plus

513

2 mm a-1 (i.e. 5 mm a-1); and the current rate plus 7 mm a-1 (i.e. 10 mm a-1) (Fig. 6a). The calibrated

514

DIVA\_WCM was run for each of these three scenarios for each DIVA\_WCM segment that falls

515

within this study area; it assumed that no dikes are present, since the primary driver analysed by

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Reed et al. (2008) was wetland drowning due to insufficient vertical wetland growth. Percentage

517

wetland losses as predicted by the DIVA\_WCM, and mapped into groups suggested by the Reed et

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al. (2008) categories, are shown in Fig. 6b-6d for each of the three sea-level rise scenarios. The

519

categorical comparison shows that the DIVA\_WCM reproduces the general patterns of increasing

520

wetland vulnerability with increasing rates of SLR. While Reed et al. (2008) conclude that most parts

521

along the US mid-Atlantic marshes are unlikely to be converted to open water under current rates

522

of SLR, modelled loss rates with DIVA\_WCM for the linear 3 mm a-1 SLR are <50% in most coastal

523

segments of the study area. Following Reed et al. (2008), with 5 and 10 mm a-1 SLR rates wetlands

524

are expected to survive to a marginal extent only or completely disappear respectively. Equivalent

525

conclusions can be drawn from the model validation runs, indicating wetland loss rates between 50

526

and 75% and >90%, respectively.

527

While the general trend of modelled wetland loss rates compares well with the EPA

528

assessment, the spatial patterns in the area, as suggested by Reed et al. (2008), are poorly

529

represented in the model results. The most important reason for this is the relatively large length of

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coastal segments in estuarine environments, smoothing estuarine gradient and neglecting local

531

variations in tidal range and sediment supply. This in turn highlights the spatial scale at which the

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DIVA\_WCM results have to be interpreted and the conclusions that can be drawn from them. The

533

model is suitable for identifying hotspot regions (~200 km coastline length) of coastal wetland loss

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535

but is not applicable for sub-regional (< 100 km) scale analysis.

536

537

*Figure 6 near here*

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539

# Results

540

The DIVA database indicates a mapped total global coastal wetland stock (in 2011) of 756

541

x103 km2. This figure compares to the 302 x103 km2 reported by Hoozemans et al. (1993), for data

542

collected in the 1980s. Absolute and relative rates of global wetland loss between 1995 and 2100

543

are shown in Figure 6 for the high, medium and low scenarios of global sea-level rise and the three

544

dike scenarios, as outlined earlier (sections 2.1, 2.2.2. respectively). These combinations give

545

wetland loss by 2100 in the range from 281 to 592 x103 km2, or between 37 and 78 % of the total

546

stock of global coastal wetlands (Table 3). Total wetland loss from 1995 (mean sea level during

547

1985-2005 baseline period) to 2100 strongly varies with sea-level rise, with wetland losses being 27

548

- 31% lower for the lowest SLR scenario in comparison to the highest SLR scenario, independent of

549

550

the dike scenario (‘no dikes’, ‘widespread dikes’, ‘maximum dikes’) applied (Fig. 7, Table 3).

551

552

*Figure 7 near here*

553

554

*Table 3 near here*

555

In order to obtain a better understanding of which of the applied forcing factors are

556

primarily responsible for these wetland losses, we report the results of a sensitivity analysis for the

557

558

weights of the different forcing factors (see Supplementary Material Equation 4).

559

* 1. *Global coastal wetland loss rates by weighting of environmental forcing factors under high sea-*

560

561

*level rise*

562

Under the high sea-level rise scenario of 110 cm by 2100 (95% quantile, RCP8.5; Table 1)

563

and with no dike building, the DIVA\_WCM model predicts a loss of between 392 and 578 x103 km2

564

of coastal wetlands worldwide by 2100 (Fig. 8a), or 52 - 76% of the total global stock, depending

565

upon the comparative weighting of the three environmental forcing factors (Table 4). The loss of

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total global stock is 11 – 18% by the 2020s, 27 - 43% by the 2050s and 42 - 65% by the 2080s (Fig.

567

568

8a).

569

570

*Figure 8 near here*

571

A sensitivity analysis shows that, in the absence of dikes, the variation in loss rate is strongly

572

controlled by the influence of the accommodation space term. Where accommodation space has a

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relatively high weighting (and wetlands can migrate inland over low coastal slopes), loss rates are at

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the lower bound (Fig. 8a, Table 4(a)); where the influence of accommodation space is neglected by

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the model, all combinatorial weightings of sea-level rise and sediment supply give rise to high rates

576

577

of total wetland loss by 2100 (Table 4(b)).

578

*Table 4 near here*

579

580

The importance of accommodation space points to the critical importance of dike

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construction under the high sea-level rise scenario. Under the most extreme scenario of dike

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building tested, ‘maximum dikes’, the DIVA\_WCM is largely insensitive to any of the model

583

parameters (Fig. 8b, Table 5), as the benefit of accommodation space is lost (cf. Table 4(a)). Global

584

585

wetland loss rates are very high, at 574 – 619 x103 km2 of the total stock of 760 x103 km2 (Fig. 8b).

586

587

*Table 5 near here*

588

* 1. *Global coastal wetland loss rates by weighting of environmental forcing factors under low sea-*

589

590

*level rise*

591

Wetland loss rates are significantly less (Table 3) under the low sea-level rise scenario (5%

592

quantile, RCP2.6, Table 1) and there is less acceleration in the wetland loss rate towards 2100 (Fig.

593

7). With no dike building, the DIVA\_WCM model predicts a loss of between 222 – 356 x103 km2 of

594

coastal wetlands worldwide by 2100, or 29 – 47% of the total global stock, depending upon the

595

comparative weighting of the three environmental forcing factors (Fig. 8c). The accommodation

596

space term remains an important discriminator within this range but the overall range in loss rate is

597

a third less than under the high sea level scenario (Fig. 8c, Table 6(a)). When the accommodation

598

space term is removed, it is clear that the main control on wetland loss is the sea-level rise / tidal

599

range term, *rslr\_d*, (Table 6(b)) with lower loss rates where this term is high. Under the highest level

600

of dike construction (‘maximum dikes’), the envelope of loss rates narrows and rises but not

601

greatly, to between 312 – 418 x103 km2 of coastal wetlands worldwide by 2100, or 41 – 55% of the

602

total global stock (Fig. 8d). Similar to the results when accommodation space is neglected, the loss

603

rates are controlled by the slr / tidal weight, with lower loss rates when the slr / tidal weight is high

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605

(Table 7).

606

607

*Table 6 near here*

608

609

*Table 7 near here*

610

611

* 1. *Global patterns in predicted wetland loss rates*

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As well as the global estimates of wetland loss, the results can be disaggregated down to

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individual segment level where wetlands have been recorded. It is thus possible to view both the

614

global pattern of potential absolute wetland loss (Fig. 9a) and relative wetland loss (Fig. 9b). These

615

plots assume ‘widespread dikes’ and the medium sea-level rise scenario (median, RCP4.5; Table 1).

616

In these contexts, the wetlands that appear most at risk are those characterised by micro-tidal

617

618

settings. Regional hotspots include the Mediterranean Sea, the Caribbean Sea and the Baltic Sea.

619

*Figure 9a near here*

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621

*Figure 9b near here*

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# Discussion

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These results show that coastal wetlands are sensitive to sea level rise and, based on

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credible scenarios for the 21st century, there is a potential for considerable wetland loss at the

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global scale. This will be exacerbated by coastal squeeze caused by the construction, and upgrading,

627

of dikes which, whilst providing flood defence to coastal populations and infrastructure, prevent

628

the onshore and upslope migration of wetlands. This model-based conclusion is consistent with

629

other assessments in the scientific literature (e.g. Nicholls et al., 2007; Wong et al., 2014). It is also

630

consistent with earlier global assessments. While considering a smaller global wetland stock,

631

Hoozemans et al. (1993) concluded that a 1 m sea-level rise might cause coastal wetland loss of 154

632

– 180 x103 km2, or 51 – 60% of total global stock, depending upon assumptions about development

633

and dike construction. For a similar sea-level rise scenario, Nicholls et al. (1999) estimated wetland

634

losses of up to 46% of global coverage. The losses associated with a 38 cm rise in sea level by the

635

2080s were estimated at 0 – 2% by the 2020s, 2 – 11% by the 2050s and 6 – 22% by the 2080s. In

636

this analysis, evaluating the contribution of lateral accommodation space and sediment supply

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controls as well as sea-level rise, the most comparable sea-level rise scenario, the 5% quantile of

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RCP2.6 (29 cm by 2100), gives loss rates of 10-11% by the 2020s, 23-28% by the 2050s and 32-40%

639

by the 2080s. However, under the 95% quantile of the RCP8.5 sea-level rise scenario (110 cm by

640

2100), the wetland loss rates rise to 14-15% in 2020s, 33-40% in the 2050s and 53-66% in the

641

2080s. Hence, in its current form the DIVA\_WCM model shows higher sensitivity to sea-level rise

642

than these earlier analyses, and losses two or more times higher than these earlier estimates

643

appear possible.

644

However, this sensitivity may be reduced in future iterations if appropriate feedbacks from

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changing plant physiology and tidal hydrodynamics can be included in the model structure. Thus,

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for example, increased atmospheric CO2 and warmer temperatures, allied to mid-range rates of sea

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level rise, may lead to increases in the rates of plant productivity and wetland accretion (Langley et

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al., 2009; Cherry et al., 2009; Kirwan and Gutenspergen, 2012; Kirwan and Mudd, 2012), These

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dynamics might be further reinforced by increased sediment supply to wetland surfaces with

650

greater tidal energetics under higher sea levels, albeit with limits to ‘ecogeomorphic’ adaptability at

651

higher rates of sea level rise (Kirwan et al., 2010).

652

These rates also need to be seen in the context of wetland losses resulting from

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anthropogenic impacts. Thus, for example, Dodd and Ong (2008) have estimated that the coastal

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populations of nation states with mangroves will rise by 50%, from 1.8 billion to 2.7 billion, in the

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period between 2000 and 2025. Human pressures on mangroves include direct conversion to urban

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use for industry, port development, and housing; conversion for aquaculture and agriculture;

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timber extraction; and modification of hydrology and pollution, particularly oil pollution, nutrients

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associated with agricultural intensification, and heavy metals contamination. These pressures will

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be imposed upon mangrove systems (plus other wetlands) already suffering significant long-term

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declines in extent (Spencer and Möller, 2013). Whilst the exact figure for loss may be debateable,

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the general sentiment of Nicholls et al.’s (1999, S82) statement that ‘when combined with the

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direct loss scenarios due to direct human destruction, in the worst case 36 % to 70 % of the world’s

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wetlands (up to 210,000 km2) could be lost by the 2080s’ surely remains true (and with the total

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global wetland area reported here the 70% would equate to 529 x103 km2).

665

This paper emphasises the importance of lateral accommodation space in mitigating high

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rates of wetland loss under high rates of sea-level rise. Such a finding gives support to those

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management strategies that aim to create or re-create space into which coastal wetlands can

668

retreat landwards under sea level forcing (e.g. UK: Rupp, 2010; Dawson et al., 2011; Canada: Djeza

669

et al., 2011; Australia: Abel et al., 2011). However, it is also clear that in many localities such set-

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back is not possible, either because of existing human occupation and development (e.g. McLeod et

671

al., 2011) or because natural topographic settings are often not conducive to such migration. Thus,

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for example, the rapid onshore steepening of coastal profiles inland from wetland fringes along

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most of the Californian coast severely limits migration sites along this coast (Committee on Sea

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Level Rise in California, Oregon and Washington, 2012).

675

The broad-scale nature of the model presents a major challenge to model calibration and

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validation and this in turn depends on the development of more systematic national to regional

677

scale assessments of wetland behaviour. A key model output has been the derivation of a series of

678

habitat-specific wetland response curves describing the transition between different wetland types.

679

Based on recently published estimates for habitat-specific regional wetland change, the calibration

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of these curves provides an important focus for the linking of regional scale assessments to global

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scale wetland modelling. Similarly, such regional scale assessments, either empirical or modelled,

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are necessary for model validation and, within this study, have been shown to give important

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information on the temporal and spatial accuracy of the DIVA\_WCM. An appropriate choice of

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calibration and validation data smooths over the fine scale variability in wetland response to

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environmental forcing which characterises the vegetation ‘mosaic’ of many wetlands and of which

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there are many studies. Model validation should be taken over long timescales so as not to give

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undue weight to the impacts of individual high-magnitude events or even long-term cycles in tidal

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flooding regimes. However, this remains a considerable challenge because of the lack of suitable

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large-scale data that explicitly address this question in a truly quantitative manner. Progress in

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better understanding wetland response to sea-level rise requires continued improvement of the

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underlying datasets, and studies across scales from local to global, with bridging regional

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assessments as utilised in this study.

# Conclusions

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The DIVA\_WCM has been developed to better identify the vulnerability of coastal wetlands

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at the macro-scale, over a timescale of up to 100 years and at global, continental and national

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scales spatial scales. The utility of the model is therefore directed towards decision-makers and

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analysts interpreting and evaluating wetland vulnerability to climate change on these scales. It gives

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a new and important perspective on coastal wetland behaviour at a spatial scale where existing

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models are limited and data is surprisingly poor. Here we focus on the global results as a diagnostic

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output.

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The modelling approach described in this paper, which considers three environmental

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factors, suggests that the potential rates of global coastal wetland loss over the coming decades to

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2100 are substantial. Countering these potential losses will require both climate mitigation (a global

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response) to minimise sea-level rise, and promotion of accommodation space and sediment supply

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(a regional response) to promote wetland survival. Collectively, these measures could greatly

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reduce losses if applied at a sufficient scale but some net loss appears inevitable given current

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trends and lock-in to some sea-level rise. Given the now clear ecosystem service value of coastal

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wetlands, and the magnitude of these long-term predicted losses, wetland management should

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become an environmental policy priority, even in areas where the existing threat from sea-level rise

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appears currently minimal. Results from DIVA\_WCM suggest that developing a greater

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understanding of the specific geomorphic natural slope settings which result in greatest levels of

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forcing on wetland loss would be useful in developing coastal wetland protection policy.

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Further development of the model is now needed to better assess the role of

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ecogeomorphic feedbacks to see if the incorporation of these terms fundamentally affects model

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outcomes (e.g. Kirwan et al., 2010; Shile et al., 2014). In addition, independent validation of the

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results predicted by the DIVA\_WCM, particularly across different geographical regions and

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timeframes, remains an important but difficult task. It is hoped that this broad-scale modelling of

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coastal wetlands will stimulate both the quantity and approach of field measurements, such that

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the data required to validate this type of model become more widely available. Changes to coastal

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725

wetlands need to be evaluated at multiple scales, including the macro-scale considered here.

726

# Acknowledgements

727

The authors gratefully acknowledge funding from the European Union under contract

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number EVK2-2000-22024. They thank all their partners in the DINAS-COAST project Dynamic and

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pioneering efforts of Mark Spalding and Carmen Lacambra.

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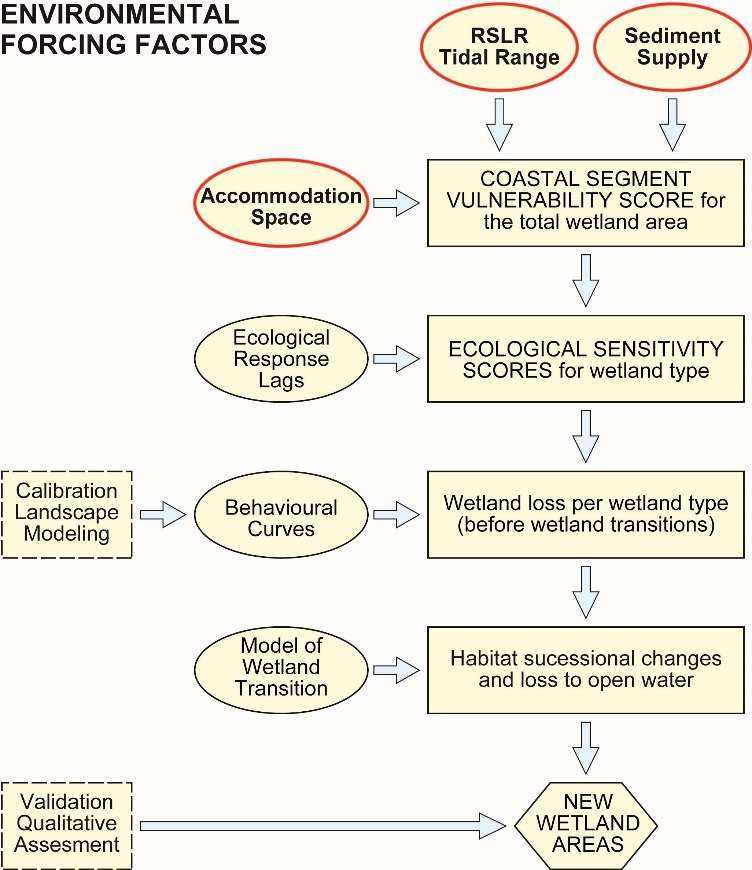
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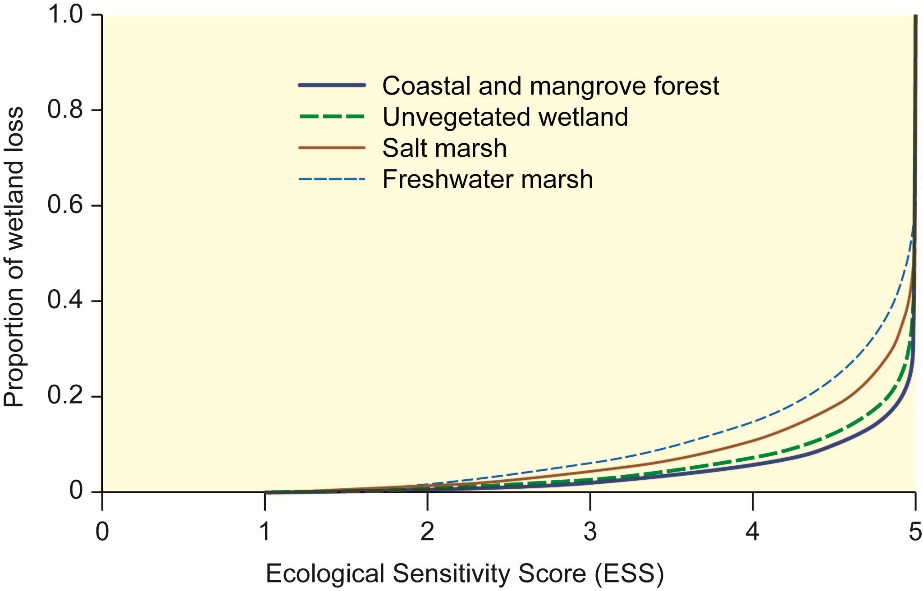
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**Fig. 1.** Model structure in the revised DIVA\_WCM wetland loss and transition algorithm.



1219

**Fig. 2.** Proportion of wetland loss versus wetland sensitivity as measured by the Ecological

1220

Sensitivity Score (ESS; see text for definition). Values of beta: Unvegetated sediments (Sabka and

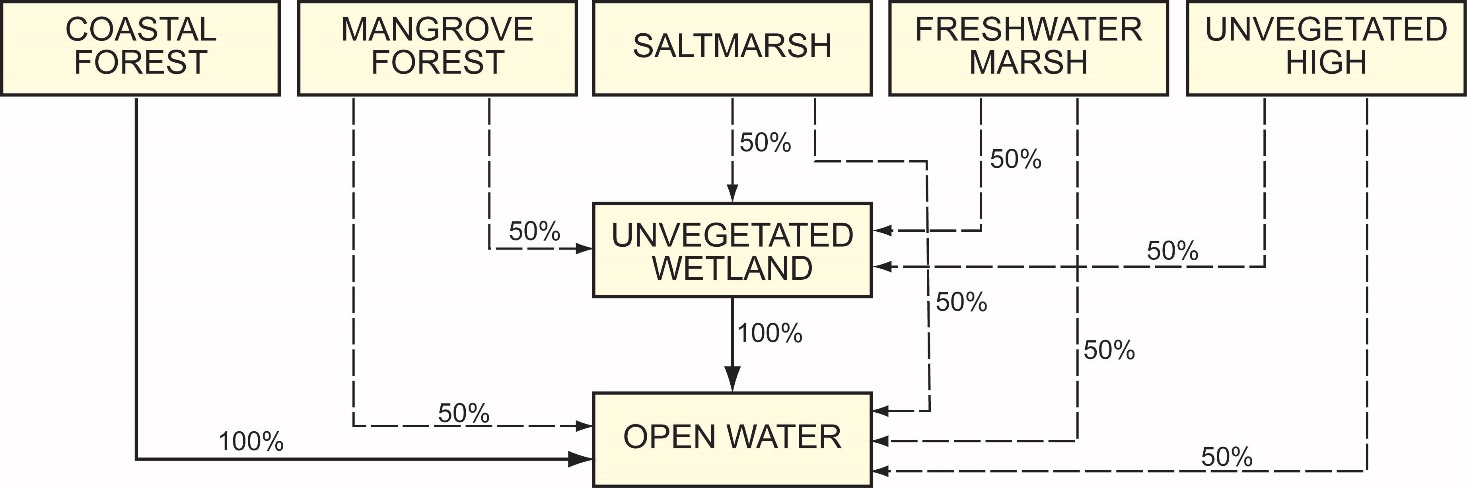
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Mudflat/Sandflat) (0.093); Freshwater Marsh (0.188); Saltmarsh (0.137); and Coastal Forest and

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Mangrove Forest (0.074).



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**Fig. 3.** Model of wetland loss and transition between wetland types and open water for low to

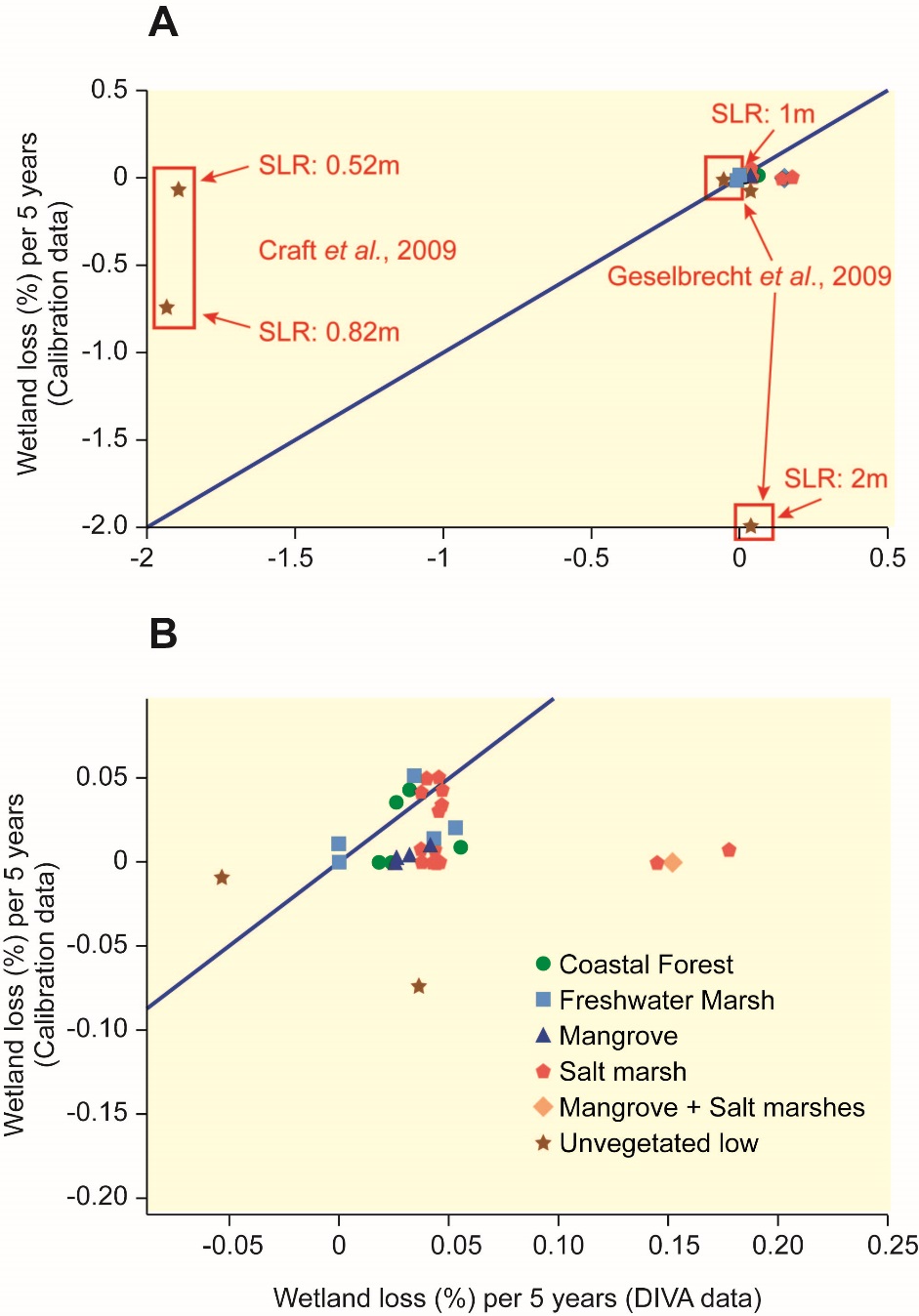
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moderate environmental forcing (Vulnerability Score < 4). For high environmental forcing

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(Vulnerability Score >= 4) all wetland areas are lost to open water.



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**Fig. 4**. DIVA\_WCM model outputs vs. wetland loss rates derived from the one WARMER and five

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SLAMM studies. In (a) all available data points are displayed (CSLID = DIVA segment number). In (b)

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only the central data points are shown. The blue line denotes the 1:1 line (perfect fit). Negative

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wetland losses (wetland gains), reported by the SLAMM studies were put to zero for all wetland

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types, where no gain is possible (all habitat types except unvegetated low).

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**Fig.5**. Coastal wetland loss in the Indo-Pacific region. A: Predicted relative coastal wetland loss by

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DIVA segment. DIVA\_WCM simulation with medium sea-level rise scenario (RCP4.5 median, 50 cm

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by year 2100) and SSP2 scenario with ‘widespread dikes’; B: Predicted loss of mangrove habitat

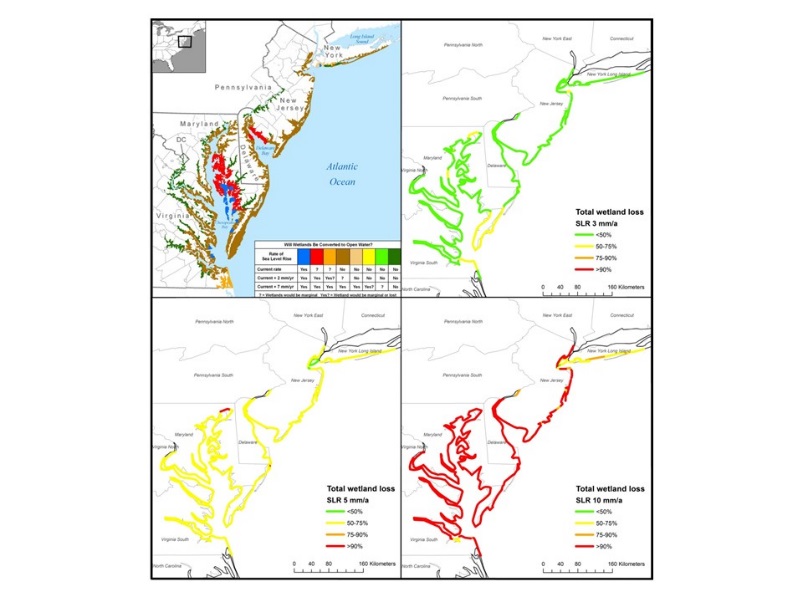
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under an RCP6 sea level rise scenario of 0.48 m by year 2100, as modelled by Lovelock et al. (2015).

Green indicates mangrove extent in 2011 (after Giri et al., (2011)) and red identifies areas of

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predicted mangrove habitat loss to sea level rise.



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**Fig. 6.** Comparison of the qualitative assessment by Reed et al. (2008) (a) with the outputs of the

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calibrated DIVA WCM for constant sea-level rise of 3 (b), 5 (c), and 10 (d) mm a-1. (a): Reed et al.

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(2008) assumes a current SLR rate of sea-level rise of 3 mm a-1. Future scenarios are a continuation

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of the current rate of sea-level rise; the current rate plus 2 mm a-1 (i.e. 5 mm a-1); and the current

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rate plus 7 mm a-1. Figures 6(b-d) can be compared to 6(a) as follows:

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%Total Loss < 50%: Wetlands will not be converted to open water (“No”)

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50%<%Total Loss < 75%: Wetlands will be marginal (“?”)

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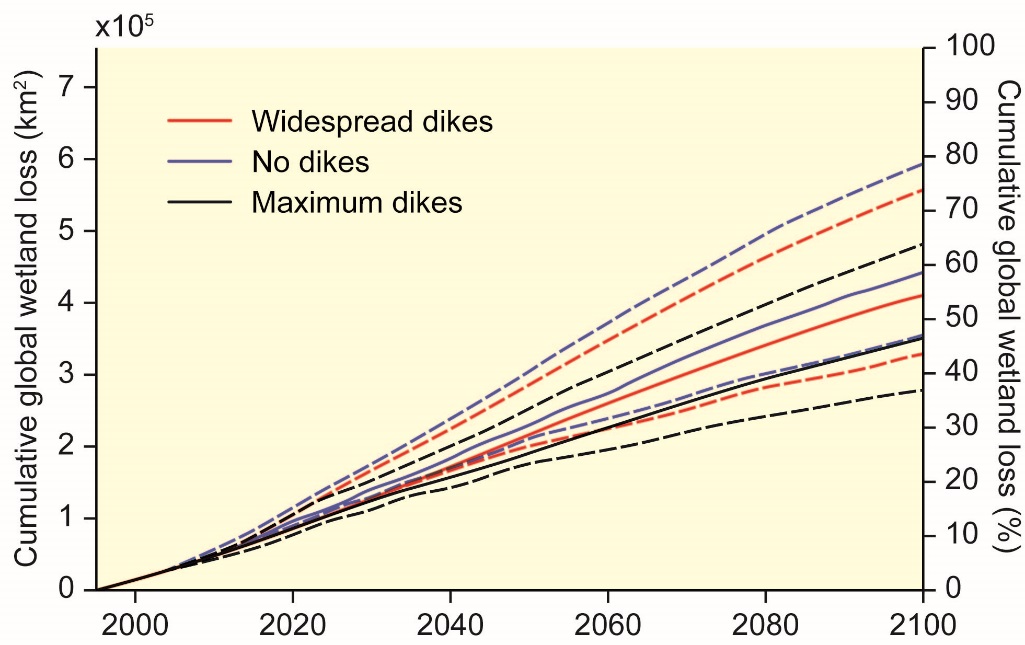
75%<%Total Loss < 90%: Wetland will be marginal or lost (“Yes?”)

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%Total Loss > 90%: Wetlands will be converted to open water (“Yes”)

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**Fig. 7.** Absolute and relative cumulative global coastal wetland loss from 1995 to 2100. Three

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scenarios describing the construction of sea dikes are considered: (i) ‘no dikes’; (ii) ‘widespread

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dikes’ built according to a ‘demand-for-safety function’, assuming an SSP2 scenario; (iii) ‘maximum

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dikes’. For each dike-scenario low (lower dashed), medium (full line) and high (upper dashed) sea-

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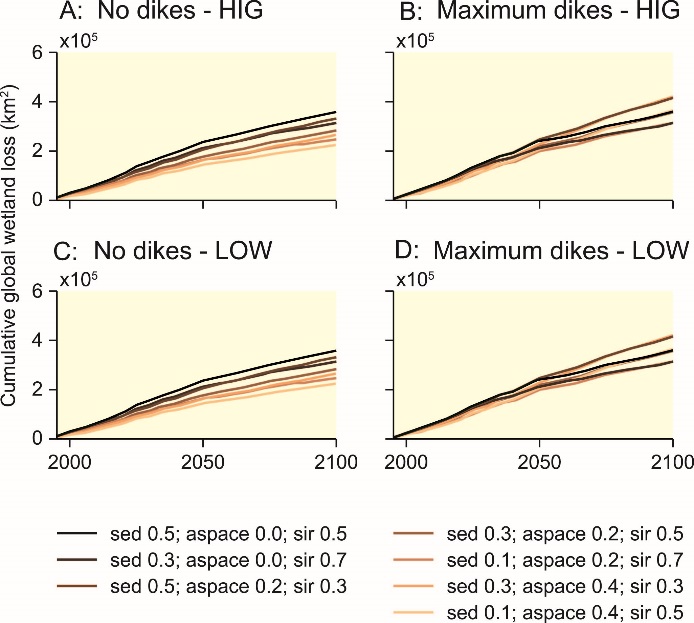
level rise scenarios are run: Low = RCP2.6 (5% quantile; 29 cm by 2100); medium = RCP4.5 (median;

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50 cm); high = RCP8.5 (95% quantile; 110 cm).

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**Fig. 8**. Absolute cumulative global wetland loss from 1995 to 2100 for different combinations of

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weights for the three environmental forcing factors (sea-level rise / tidal range (*rslr\_tidal\_score*);

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sediment supply (*sedsup*); accommodation space (*aspace*). See Tables 4 – 7 for more details. Dark

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line colours represent low weights for accommodation space, whereas bright colours indicate

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higher weights for accommodation space. Sensitivity runs were conducted assuming A) ‘no dikes’

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and high sea level rise (HIG = 110 cm by year 2100); B: ‘maximum dikes’ and high sea level rise; C:

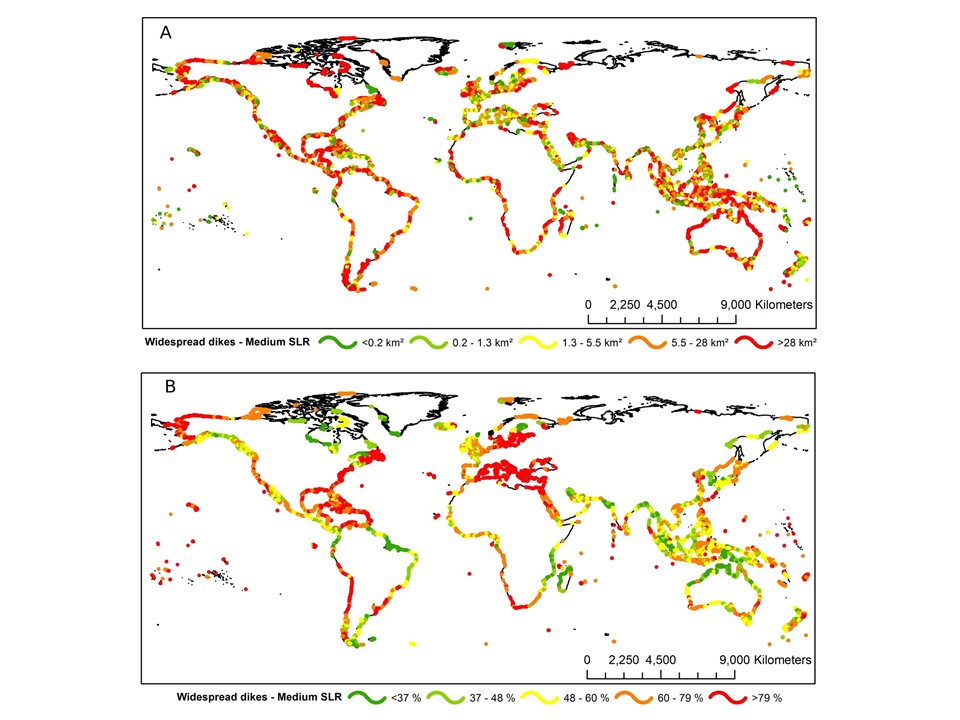
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‘no dikes’ and low sea level rise (LOW = 29 cm by year 2100); and D: ‘maximum dikes’ and low sea

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level rise.



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**Fig. 9.** Global wetland loss by DIVA segment between 1995 and 2100. A: absolute loss; B: relative

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loss. DIVA\_WCM simulation with medium sea-level rise scenario (RCP4.5 median, 50 cm by year

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2100) and SSP2 scenario with ‘widespread dikes’.

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# List of Tables

1275

**Table 1**. Global mean sea-level rise in 2100 with respect to 1985-2005. Median values, with 5% and

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95% quantiles in parentheses. After Hinkel et al. (2014). The DIVA\_WCM uses low sea-level rise =

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RCP2.6 (5% quantile; 29 cm by 2100); medium sea-level rise = RCP4.5 (median; 50 cm); high sea-

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level rise = RCP8.5 (95% quantile; 110 cm).

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| **Scenario** | **Model** | **Steric (cm)** | **Land-ice (cm)** | **Total (cm)** | **Acronym** |
| RCP2.6 | HadGEM2-ES | 14 | 21 (16, 39) | 35 (29,52) | LOW |
| RCP4.5 | HadGEM2-ES | 18 | 32 (23, 56) | 50 (41,75) | MED |
| RCP8.5 | HadGEM2-ES | 29 | 44 (31, 81) | 72 (60,110) | HIG |

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**Table 2.** Location, habitat, sea-level rise scenario and model characteristics for the six calibration

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studies used to calibrate the behavioural curves used in the DIVA\_WCM. Tidal range data: in

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reference or for USA sites from NOAA Tides and Currents

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(<https://tidesandcurrents.noaa.gov/stations.html?type=Datums>).

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Suspended sediment concentrations: (1) 2011 annual mean of MERIS geophysical product Total

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Suspended Matter (TSM) in 0.017 degree resolution, northern NSW, Australia

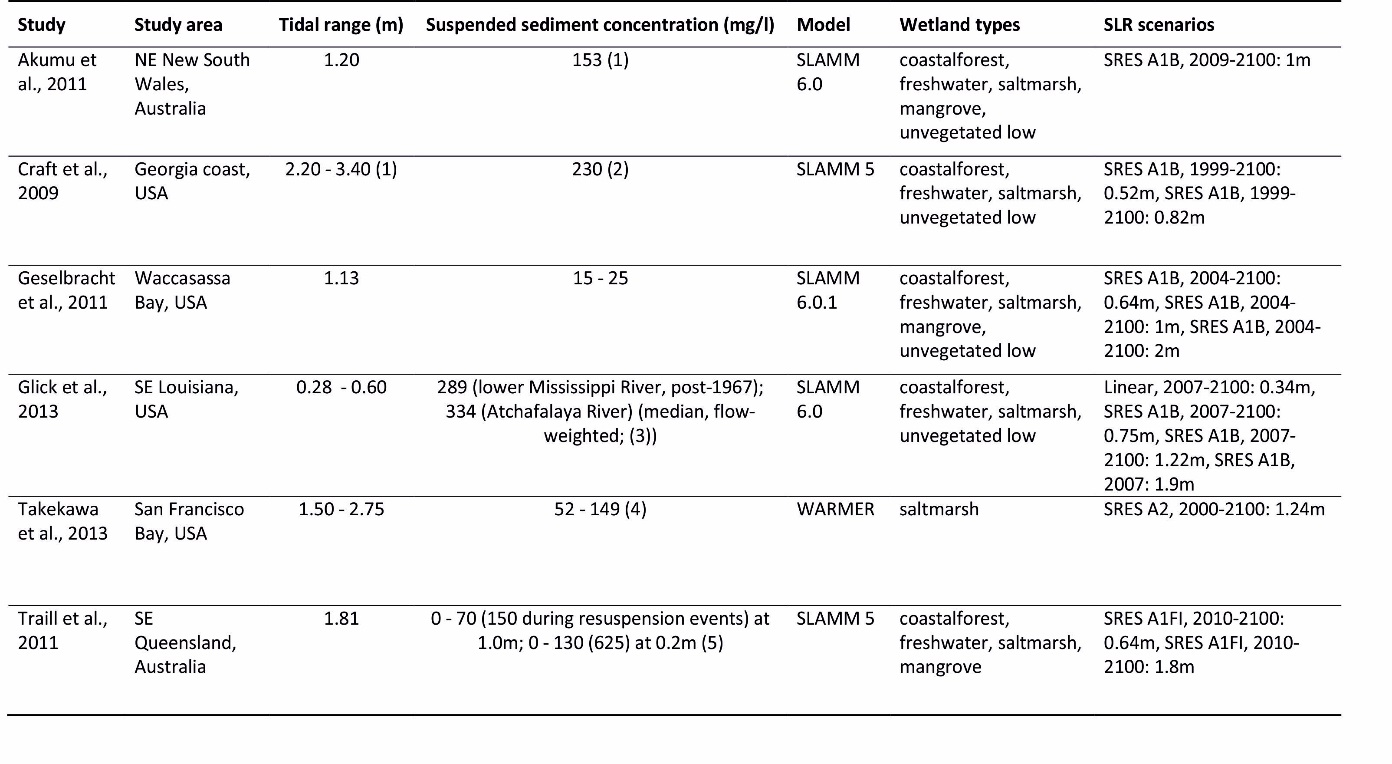
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([http://hermes.acri.fr/);](http://hermes.acri.fr/)%3B) (2) Howard and Frey (1985); (3) Heimann et al., (2011); (4) Buchanan and

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Morgan (2014); (5) You (2005).



1289**Table 3**. Absolute global wetland loss (x103 km2) and relative loss of total global wetland stock (%)

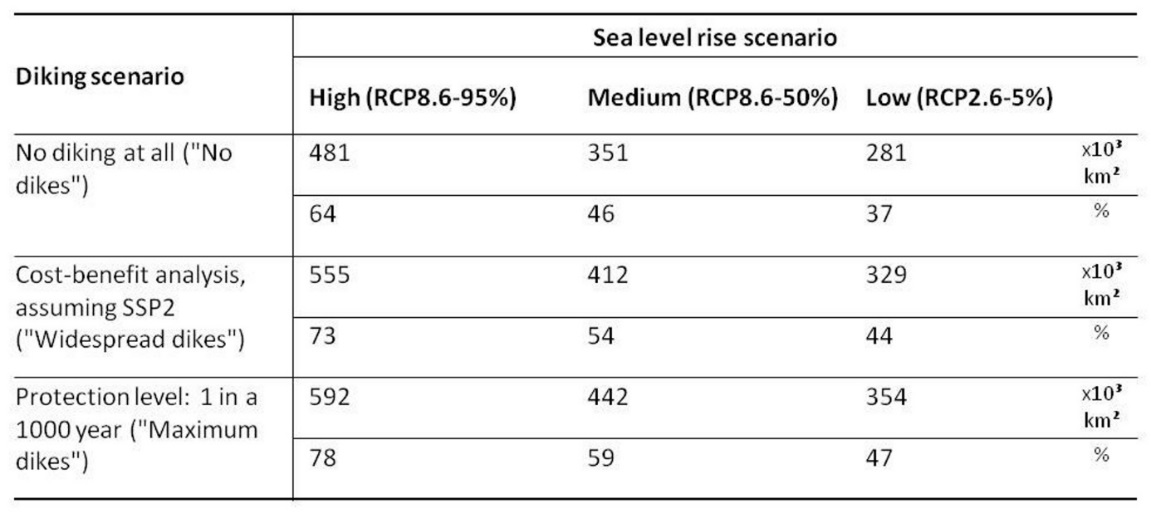
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by 2100 under the diking and sea-level rise scenarios (see Table 1 and text for details on the

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scenarios employed).



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**Table 4.** Percentage of total wetland area loss at 2100 under different weighting combinations of

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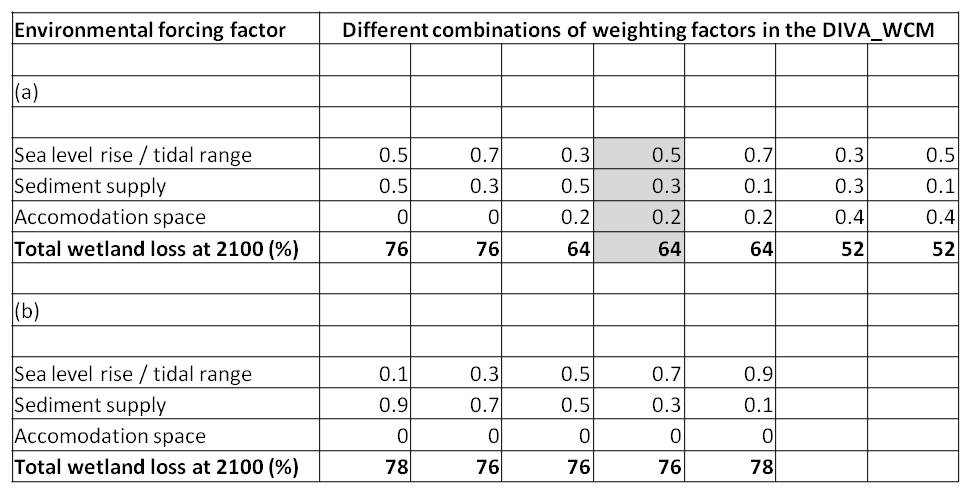
(a) three environmental forcing factors and (b) sea-level rise / tidal range and sediment supply only,

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given high sea-level rise scenario of 110 cm by 2100 (95% quantile, RCP8.5; Table 1) and ‘no dikes’.

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Shaded area = ‘standard’ DIVA\_WCM output (see Equation 4, Supplementary Material for details).



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**Table 5.** Percentage of total wetland area loss at 2100 under different weighting combinations of

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the three environmental forcing factors given high sea-level rise scenario of 110 cm by 2100 (95%

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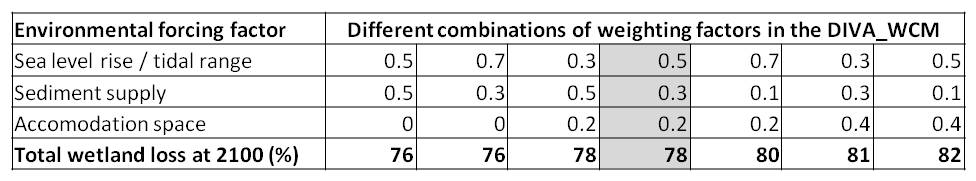
quantile, RCP8.5; Table 1) and ‘maximum dikes’. Shaded area = ‘standard’ DIVA\_WCM output (see

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Equation 4, Supplementary Material for details).



**Table 6.** Percentage of total wetland area loss at 2100 under different weighting combinations of

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(a) three environmental forcing factors and (b) sea-level rise / tidal range and sediment supply only,

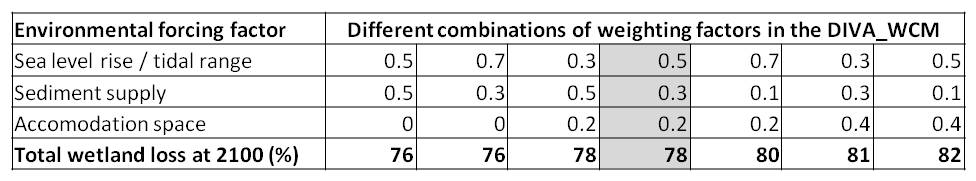
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given low sea-level rise scenario of 29 cm by 2100 (5% quantile, RCP2.6; Table 1) and ‘no dikes’.

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Shaded area = ‘standard’ DIVA\_WCM output (see Equation 4, Supplementary Material for details).



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**Table 7.** Percentage of total wetland area loss at 2100 under different weighting combinations of

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the three environmental forcing factors given low sea-level rise scenario of 29 cm by 2100 (5%

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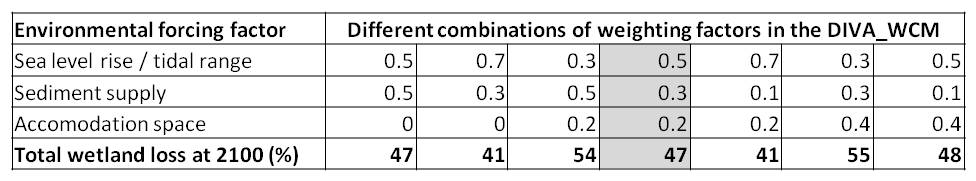
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quantile, RCP2.6; Table 1) and ‘maximum dikes’.



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# Supplementary Material

**Background tables and equations used in the DIVA\_WCM algorithm**

1. **Relative sea-level rise and tidal range forcing**

We first compute rslr\_tidal as:

rslr\_d = rslr\_annual^1.4 / htidal **Eq.1**

where rslr\_annual is the annual rise in relative sea level in metres, htidal is the tidal range in metres derived from the LOICZ typology (Maxwell and Buddemeier, 2002) as shown in Table A1.

|  |  |  |
| --- | --- | --- |
| **Tidal Range classes from LOICZ typology** | **Tidal Range (metres), LOICZ typology** | **htidal (Tidal forcing score within the DIVA Wetland Change Model)** |
| <2 | 0-2.5 | 0.25 |
| 2 | 2.5-3.5 | 1.25 |
| 3 | 3.5-5.0 | 3 |
| 4 | 5.0-6.5 | 6 |
| 5 | >6.5 | 9 |

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Table A1 Derivation of tidal forcing scores (representing tidal range) based on the tidal range classes from the LOICZ typology.

Then we convert rsrl\_d into an *rslr\_tidal\_score* between 1 and 5 based on the 95, 84, 50, 16 percentiles of all rslr\_d values where wetlands are reported (assuming a current global SLR of 3 mm a-1). Resulting class values are reported in the following table:

|  |  |
| --- | --- |
| **rslr\_d** | **rslr\_tidal\_score** |
| >=0.001121 | 5 |
| >=0.000402 | 4 |
| >=0.000178 | 3 |
| >=0.000044 | 2 |
| >0 | 1 |

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Table A2: Assigning a forcing value to the impact of relative sea-level rise and tidal range on wetland vulnerability.

# Lateral accommodation space

We initialize the forcing score for lateral accommodation space based on the coastal slope (degrees) using Table A3.

|  |  |
| --- | --- |
| **Average slope (slopecst, degrees)** | **Forcing score for lateral accommodation space (aspace)** |
| >4.5 | 5 |
| >1.5 < 4.5 | 4 |
| > 0.5 < 1.5 | 3 |
| > 0.25 < 0.5 | 2 |
| <0.25 | 1 |

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Table A3: Forcing score used to represent the impact of coastal slope on wetland vulnerability. slopecst is the average topographic slope (in degrees) along the segments.

*aspace* value is initialised using the average topographic slope, derived from the ETOPO2 (NGDC, 2001) dataset. This initialized *aspace* score is then updated based on the actual computed dike height within each time-step using Equation 2. DIVA\_WCM builds sea dikes along the entire coastal segment given sea level and socio-economic forcing following the demand function of safety given in Hinkel et al. (2014).

if (sdikehght > htidal/2 and *aspace* < 5): *aspace* = *aspace* + 0.25 **Eq. 2**

Where htidal/2 is a critical value of sdikehght defining its functioning as barrier to landward movement of wetland and to flooding. This threshold value is based on expert judgement.

# Sediment supply

External of DIVA we calculate a constant sediment supply factor *sedsup* based on a variety of biophysical coastal properties.

*sedsup* = (t \* tw) + ((dis + d\_dis)/2) \* fw) + (gl \* glw) + (geo \* gew) + (man \* mw) + (his \*hw)

# Eq. 3

where:

t = Tectonic control parameter tw = Tectonic control weighting

dis = Annual river discharge parameter

d\_dis = Distance from point of discharge parameter fw = Fluvial weighting

gl = Glacial limit parameter glw = Glacial limit weighting

geo = Geomorphic setting parameter gew = Geomorphic setting weighting

man = Management parameter (presence or absence of sea dikes) mw = Management weighting

his = History of resource exploitation parameter hw = History of resource exploitation weighting

In the DIVA database we have values of *sedsup* between 1.7 and 4.9.

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| --- | --- | --- | --- | --- | --- |
| **Sediment Supply Factor** | **Description** | **Key data reference/source** | **Category** | **Forcing Score** | **Weighting Factor** |
| Tectonic Context | Global tectonic setting | Inman and Nordstrom (1971) | Passive Margins | 1 | 0.07 |
| Marginal Seas | 3 |
| Active Margins | 5 |
| Fluvial Context | Annual river discharge | Ludwig and Probst (1998) | >500 FTSS (1012 g/yr) | 1 | 0.2 |
| 100-500 FTSS (1012 g/yr) | 2 |
| 50-100 FTSS (1012 g/yr) | 3 |
| 5-10 FTSS (1012 g/yr) | 4 |
| <5 FTSS (1012 g/yr) | 5 |
| Distance to the point of fluvial discharge | Calculated by GIS | 0-30km | 1 |
| 30-70km | 2 |
| 70-120km | 3 |
| 120-180km | 4 |
| >180km | 5 |
| Glacial Context | Location relevant to maximum extent of last glaciation | Williams et al. (1991) | 100km-300km | 1 | 0.1 |
| >300km | 3 |
| <100km | 5 |
| Geomorphic Context | Coastal geomorphic setting | McGill (1958) | Sheltered coast (Inlet/delta/estuary) | 1 | 0.03 |
| Open coast | 5 |
| Management Context | Degree of coastal protection | DIVA adaptation algorithm or user inputs (Tol et al., 2005) | Sea dike absent (< 0.5 m high) | 1 | 0.3 |
| Sea dike present (> 0.5 m high) | 5 |
| Historical Context | Timing of peak resource exploitation | Expert judgment | Classical | 1 | 0.3 |
| Medieval | 2 |
| Colonial | 3 |
| 20th Century | 5 |

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Table A4 Factors influencing sediment supply and their incorporation into the DIVA\_WCM via forcing scores and weighting factors.

# Coastal segment vulnerability score (csvs)

The above calculated three forcing scores are then combined into the coastal segment vulnerability score (csvs) following Equation (4)

csvs = *rslr\_tidal\_score* \* 0.5 + *aspace* \* 0.2 + *sedsup* \* 0.3 **Eq. 4**

where *rslr\_tidal score* is the relative sea-level rise and tidal range forcing (Equation 1), *aspace* the lateral accommodation space forcing score (Equation 2) and *sedsup* the sediment supply forcing score (Equation 3).

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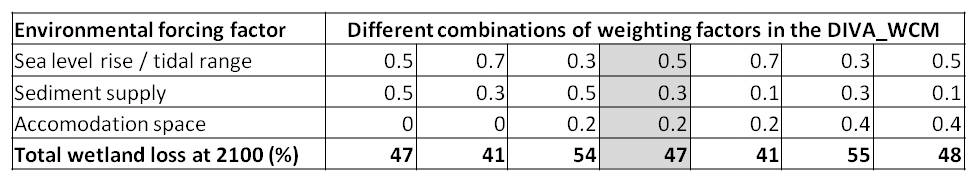
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# Ecological Sensitivity Score (ESS) by wetland type



Finally, we compute the ecological sensitivity score (ess) by combining the csvs values of the current and last time step.

# Eq. 5

where ESStype is the ecological sensitivity score for the given wetland type, csvscurrent is the coastal segment vulnerability score and csvslast is the value of this variable calculated within the previous time step, weight\_currenttype is the lag weight associated to the current time step and weight\_lasttype is the lag weight associated to the csvs value from the previous time step. The weights used are given in Table 4.

|  |  |  |  |
| --- | --- | --- | --- |
| **Wetland Type (type)** | **Previous 5 year lag weight (weight\_lasttype)** | **Current 5 year lag weight (weight\_currenttype)** | **Response time (yrs)** |
| Coastal forest | 1 | 0 | 10 |
| Freshwater marsh | 0 | 1 | <5 |
| Saltmarsh | 0 | 1 | <5 |
| Mangrove forest | 1 | 0 | 10 |
| Unvegetated wetland | 0 | 1 | <5 |
| Mudflat and sand flat | 0 | 1 | <5 |

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Table A5: Response to environmental change by wetland type as modelled by relative importance of previous and current ecological state. Response time = 5 / Current 5 year lag weight.

# Wetland response (Annual wetland loss rate)

The ess values are then translated into relative 5-years wetland loss rates (RLR5), which is the proportion of wetlands lost for a specific wetland type during a 5-year time step.

RLR5 = 1- (β +1) \* (1 – ESStype/5) ^ β + β \* (1 – ESStype/5) ^ (β + 1) **Eq. 6**

Values of beta:

|  |  |  |
| --- | --- | --- |
| 1449 | 1. | Unvegetated high and low: 0.093 |
| 1450 | 2. | Freshwater Marsh: 0.188 |
| 1451 | 3. | Saltmarsh: 0.137 |
| 1452 | 4. | Coastal Forest and Mangrove Forest: 0.074. |

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