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A Global Exploration of Tidal Wetland Creation for Nature-Based Flood Risk Mitigation in Coastal Cities

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Abstract

Coastal cities around the world are increasingly exposed to flood risks due to climate change, resulting sea level rise and more intense storm surges as well as due to growing coastal population densities and land subsidence. Nature-based risk mitigation, consisting of conservation or creation of coastal ecosystems that have the natural capacity to adapt to sea level rise and to mitigate storm surges, is increasingly proposed, but real-live implementation is yet limited to specific local cases. Our study presents a global scale analysis of the surface areas available for potential creation or restoration of tidal wetlands (salt marshes and mangrove forests) in front of 135 highly populated, flood-exposed coastal cities, as part of nature-based or hybrid strategies to buffer against coastal flood risks. Our results reveal that 34 % (4 600 km²) of the total land area within the influence zone of storm surge propagation between the sea and the cities is potentially available for tidal wetlands creation. Those areas mainly correspond to rural areas with a low population density such as croplands, paddy fields or vegetated areas and to water bodies. The key factors influencing the area potentially available for tidal wetlands creation are the geomorphology and the population density, as 60 % (8 300 km²) of the land area below mean high tide in front of the studied cities is urbanized or densely populated. Cities located along deltas or estuaries and in bays and lagoons (e.g. Hamburg, Guayaquil, Tianjin, Portland or San Jose) have generally larger low-lying coastal zones and consequently larger potentially available areas for salt marshes and mangrove forests restoration or creation for coastal flood risks mitigation. Our results contribute to increasing evidence and awareness of the possibilities of nature-based mitigation of coastal flood risks by restoring and creating tidal wetlands in front of flood-exposed coastal cities around the world.

Keywords: tidal wetlands, restoration/creation, cities, storm surge, nature-based strategies

1 Introduction

Global climate change and the related intensification of coastal hazards is threatening the coastal zone (de Sherbinin et al., 2007; Hallegatte et al., 2013; Neumann et al., 2015; Vitousek et al., 2017). This increasing risk of coastal hazard is resulting, among others, from sea level rise affecting the coasts by higher flood and erosion risk due to the action of wind and storm waves (Gedan et al., 2011; Storlazzi et al., 2011; Thampanya et al., 2006), but also from an increase in frequency of tropical cyclones and extra-tropical storms of high intensity, generating destructive storm surges when they reach the coastal area (Webster et al., 2005; Woodruff et al., 2013).

In addition to the increasing threats due to climate change, the coastal populations are facing socio-economic changes (Barbier, 2014; Hanson et al., 2011; Kron, 2013; Syvitski et al., 2009). In the Low Elevation Coastal Zone (LECZ, i.e. lower than 10 m above mean sea level), the population is expected to increase and reach by 2060 a global average density of 400 to 500 inhabitants per square kilometre (Hanson et al., 2011; McGranahan et al., 2006; Neumann et al., 2015), or two times the current global population density in the LECZ. This augmentation of the population pressure in the coastal zone also implies an increase of the assets exposed to coastal hazards (Barbier, 2014; Hallegatte et al., 2013; Hanson et al., 2011). Furthermore, the human influence on its natural environment, such as a reduced sediment supply to coastal zones by the trapping of sediments in upstream river dams (Auerbach et al., 2015; Syvitski et al., 2005) or the extraction of oil, gas or water from the substrate beneath coastal zones, is leading to the intensification of coastal land subsidence that further contributes to the increasing vulnerability of the coasts to flood and erosion risks (Balke and Friess, 2016; Kirwan and Megonigal, 2013; Syvitski, 2008).

Consequently, the observed decline in the world's tidal wetlands over the recent decades and the projections for the next decades are worrying (Duke et al., 2007; IPCC, 2007; McLeod et al., 2011; Pendleton et al., 2012). The disappearance of tidal wetlands is predominately an effect of the historical and present anthropogenic pressures on coastal areas, by conversion of mangroves and salt marshes into agriculture, aquaculture, urban and industrial areas. As such the loss of mangrove forests and salt marshes over the last century was estimated at 20 to 50 % of their total area (Food and Agriculture Organization (FAO) of the United Nations, 2007; McLeod et al., 2011; Spalding et al., 1997; Valiela et al., 2001). Furthermore, the degradation of the remaining mangrove forests and salt marshes through over-exploitation for timber, over-fishing, pollution or solid waste disposal is reducing their valuable ecosystem services, including their natural capacities to act as a barrier against wind and storm waves (FAO, 2007; Scott et al., 2014; Spalding et al., 2010). In the future, losses of mangrove forests and salt marsh areas are expected to be linked to amongst others continued land reclamation, wetlands degradation and relative sea level rise. Although, some studies estimate that the rate of degradation and loss of mangrove forests diminished since 2000 (Blankespoor et al., 2014; Duke et al., 2007; Hamilton and Casey, 2016; IPCC, 2007; Ma et al., 2014; Pendleton et al., 2012; Valiela et al., 2001). Important to notice in this respect, is that recent regional to global scale studies indicate that tidal wetlands have a high capacity to maintain surface area with future sea level rise, while the most important threats come from direct anthropogenic pressures, such as wetland conversion into human land use and human infrastructure that prevents inland wetland migration with sea level rise (Kirwan et al., 2016; Schuerch et al., 2018).

The need to develop strategic plans to mitigate coastal flood risks over the short and long term is increasing. In this respect, there is an increasing interest for so-called nature-based risk mitigation, i.e. the conservation, restoration and creation of natural habitats as a contribution to coastal protection against flood and erosion risks (Cheong et al., 2013; Costanza et al., 2008; Gedan et al., 2011; Sutton-Grier et al., 2015; Temmerman et al., 2013). Tidal wetlands, corresponding in this study to salt marshes in temperate and tropical climates and mangroves in (sub-)tropical areas, are widely considered as coastal habitats that provide a mitigating effect on flood and erosion risks in more landward located, human-occupied coastal plains (Barbier et al., 2011; Cheong et al., 2013; Duarte et al., 2013; Gedan et al., 2011; Temmerman et al., 2013). These ecosystems have a certain capacity to build up elevation with sea level rise through sediment accretion (Kirwan et al., 2016; Kirwan and Megonigal, 2013; Krauss et al., 2014; McIvor et al., 2013; Temmerman and Kirwan, 2015), to reduce wind waves and shoreline erosion, and to attenuate the landward propagation of storm surges due to friction provided by the wetland's vegetation and topography (Barbier et al., 2011, 2008; Gedan et al., 2011; Guannel et al., 2016; Narayan et al., 2016; Temmerman et al., 2013; van Wesenbeeck et al., 2017).

In the context of globally decreasing tidal wetland areas, the restoration or creation of tidal wetlands in combination with coastal engineering solutions like dikes or dams, is considered essential to allow populated coastal areas to mitigate and adapt to the increasing risks of coastal flooding and erosion (Barbier et al., 2008; Ma et al., 2014; Scott et al., 2014; Spalding et al., 2013, 2010; Valiela et al., 2009). Plans of such hybrid coastal protection schemes which combine tidal wetlands creation and hard engineering are already applied in various coastal areas over the world. The restoration of the Mississippi delta through the Louisiana Coastal Master Plan is one of the largest known examples of such combination of coastal marsh restoration or creation, and engineering of levees for coastal protection, with the expected restoration or maintenance of more than 2 000 km² of marshlands over the next 50 years (Boesch et al., 2006; Coastal Wetlands Planning Protection and Restoration

Act (CWPPRA), 1990; Day et al., 2007; RESTORE, 2017). Similarly, the San Francisco Bay Joint Venture works on the protection, restoration and enhancement of, among others, about 550 km² of tidal flats, marshes and lagoons and some 260 km² of seasonal wetland areas over the San Francisco Bay with the combined objective to gain benefits for wildlife and coastal protection (San Francisco Bay Joint Venture, 2018). At smaller scales, countries in Europe are applying managed realignment of their engineered flood defences (dikes), through landward relocation of dikes enabling the creation of tidal marshes on formerly embanked land. For example, in England and Wales, numerous projects are leading to the total realignment of about 660 km of coastline by 2030 with the aim to create 62 km² of intertidal areas (Esteves, 2014; Pendle, 2013); in Belgium, by 2030 about 40 km² of flood control areas are being realized on formerly embanked land (Meire et al., 2014; SigmaPlan, 2017). After the devastating 2004 tsunami in South East Asia and typhoon Haiyan in the Philippines in 2013, field observations indicated smaller damages in villages sheltered behind mangrove forests (Balke and Friess, 2016; Dahdouh-Guebas et al., 2005; Danielsen et al., 2005). In response to these flood disasters, associations and countries (i.e. Indonesia, India, Sri Lanka, Thailand and Malaysia) together instigated mangroves restoration projects (FAO, 2007; Schmitt, 2012). This resulted in, for example, the restoration of 20 km² of mangrove forest in Indonesia and the plantation of 310 000 seedlings over the Sri Lanka's coasts, of which about 60 % survived (Schmitt, 2012). In Pakistan, about 80 km² of mangrove forests are restored along the coasts, while development of restorations strategies in the Indus delta in collaboration with the IUCN (International Union for Conservation of Nature) is scoping to restore 100 km² of mangrove forest in the delta (Marois and Mitsch, 2015; MFF Pakistan, 2016; Schmitt, 2012; Spalding et al., 2014).

These examples of medium to large scale tidal wetland creation programs for mitigation of coastal flood and erosion risks exemplify the potential of nature-based mitigation programs on local to regional scales. Yet, on a global scale, there are no studies that explored to which extent major population centres, exposed to coastal flood risks, may benefit from nature-based risk mitigation by tidal wetland creation. Several studies have identified the number of people and assets exposed to coastal flood risks on a global scale (Hallegatte et al., 2013; McGranahan et al., 2007), showing that especially cities located in low-lying river deltas and coastal plains are major hotspots of coastal flood risks (Hallegatte et al., 2013; Hanson et al., 2011; Nicholls et al., 2007). However, until now, there are no global-scale studies yet that have assessed in which geographical settings there is potential for tidal wetland creation in front of flood-exposed coastal cities. Consequently, little is known on where in the world such restoration or creation of tidal wetlands could be realized both in terms of ecological and socio-economic feasibility. A global scale analysis of the potential for tidal wetland creation for coastal risk mitigation is lacking so far, but could contribute to promote the more widespread implementation of nature-based flood risk mitigation programs into policy on coastal zone management at several places around the world. Our study aims to provide a first global identification of the land surface areas where tidal wetlands, i.e. salt marshes and mangrove forests, could be restored or created in front of the world's most flood-exposed coastal cities; and we discuss geographical factors that influence the global-scale spatial variability in available surface areas for tidal wetland creation.

2 Method

The coastal cities considered in the analysis correspond to 135 cities studied by Nicholls et al. (2007) that have a population of more than 1 million people in 2005 (UN, 2005) and are exposed to coastal flood damages generated by storm surges and high winds without any consideration of coastal defences or adaptations. The city of Helsinki in Finland could not be included due to data availability. For each of these 135 cities, we identified the potential areas for tidal wetland creation that may contribute to nature-based flood risk mitigation. Below, in section 2.1, we first describe the different data sources that were used. Then, in section 2.2, we describe the procedure of how each of the data sources is used to identify potential areas for tidal wetland creation in front of the studies 135 cities.

2.1 Data

For the purpose of this study, we selected data sources that are globally available and that permit high spatial resolution and accuracy in comparison to other datasets. They are described below.

The *bathymetry* is coming from the General Bathymetric Chart of the Oceans (GEBCO) (British Oceanographic Data Centre, 2017) and represents a gridded bathymetry of the oceans at a 30 arc second resolution combined with the land topography defined by the National Aeronautics and Space Administration (NASA) Shuttle Radar Topography Mission (SRTM).

The NASA SRTM Global 30 arc second V003 dataset (NASA JPL., 2013) was used as *digital elevation model* (DEM), as it is the best known DEM available at global scale (Rodriguez et al., 2006; Sun et al., 2003).

Information on the *tidal amplitude* in front of every city was derived from the Finite Element Solution (2012) – Global Tide from AVISO. The Principal Lunar semi-diurnal component (M2) was used to define the averaged tidal amplitude in front of every city.

The location of the *urban areas* was determined from the Global Land Cover by National Mapping Organizations (GLCNMO) dataset that classifies the status of the world's land cover into 20 categories (category 18 corresponding to the urban areas, see below) based on the Land Cover Classification System (LCCS) developed by the FAO (Tateishi et al., 2014).

The *population distribution* originates from the LandScan 2013 Global Population Database (Bright et al., 2013). It represents the population over a 30 arc second grid resolution and integrates the diurnal movements and collective travelling behaviour of the population, i.e. the so-called "ambient population", averaged over 24 hours (Bright et al., 2013; Dobson et al., 2000). The dataset was adapted to deliver values of population density following the guidelines of the LandScan documentation (Bright et al., 2013; UT BATTELLE LLC., n.d.). This dataset was used to determine the population density.

The location of the *existing tidal wetlands*, defining the salt marshes and mangrove forests in this study, was determined based on the *Global distribution of Mangroves* (Giri et al., 2011) and the *Global distribution of Saltmarshes* (McOwen et al., 2017) from the United Nation Environmental Program – World Conservation Monitoring Centre (www.unep-wcmc.org).

The coastline in front of each city is extracted from the representation of the country boundaries as they exist in January 2015 and available through the ESRI platform (ESRI, DeLorme Publishing Company, Inc., 2015).

2.2 Procedure

To identify areas suitable for the development of mangroves and salt marshes that can contribute to flood risk mitigation, we need to delineate the likely area of storm surge propagation in front of the cities using the following procedure.

Firstly, the offshore starting point of the storm surge propagation was defined; it corresponds to the limit between the open sea where the storm is generated and the smaller coastal and estuarine water areas or emerged land, where the storm surge propagates as a long wave towards the city centre. The limit was created by splitting the datasets of bathymetry (GEBCO 14) and topography (SRTM) in two categories, namely the water bodies (elevation < 0 m) and emerged lands (elevation > 0 m). A focal statistic analysis was used to define the limit of the 'open sea', i.e. the water bodies with a width larger than about 3 km, by reclassifying the central pixel of a moving window of 10 by 10 pixels and granting it the most represented value in the window, i.e. water body or emerged land. The limit between those two areas is considered as the open sea limit, and corresponds to the line where the path of a storm surge will start to be considered in our analysis.

Secondly, an assumption is made that a storm surge will preferably propagate over water bodies (i.e. embayments, estuarine or deltaic channels) instead of over emerged land. As such, the most likely pathway a storm surge will follow from the open sea towards the city center is defined via a cost distance algorithm in which the emerged land areas were given a cost of 1000 and water bodies narrower than about 3 km a cost of 1. Some examples can be found in the Appendix A.

Finally, the likely area of storm surge propagation in front of each coastal city was defined as a buffer area of 20 km around the above defined probable storm surge pathway. We applied a buffer area of 20 km, based on the following argumentation. Tidal wetlands (salt marshes and mangroves) located along the likely storm surge pathways (i.e. mostly channels within deltas or estuaries) act as water storage areas during storm surges, and as such contribute to reduction of the peak storm surge level when storm surges propagate inland; this mechanism has been demonstrated by local field and hydrodynamic modelling studies (Loder et al., 2009; Smolders et al., 2015; Stark et al., 2016; Wamsley et al., 2010). Such studies also showed that propagation of storm surges over tidal wetlands is attenuated at rates (expressed as reduction of peak water level per distance travelled over the wetland) ranging in the order of 5 to 25 cm/km (see e.g.(Stark et al., 2015; Van Coppenolle et al., 2018; Wamsley et al., 2010) for reviews on such attenuation rates reported in the literature). Assuming that storm surge levels are typically in the order of 1 to 2 m above the surface level of tidal wetlands (e.g. Krauss et al., 2009), and assuming an average storm surge attenuation rate of 10 cm/km (see above), we consider that tidal wetlands within a buffer of 20 km from the likely storm surge pathway still contribute to storm surge attenuation. Wetlands farther away, outside of this 20 km buffer, can be considered to have a negligible effect on storm surge mitigation. Therefore a buffer of 20 km was applied.

Furthermore, we used this buffer approach instead of a delineation based on the elevation of the land compared to the height of an incoming storm surge for example, because the latter may introduce limitations related to the combination of the different global datasets with different resolutions and accuracies. For example, some mangrove or salt marsh areas may be erroneously located in areas defined by the global land elevation dataset at several meters above mean sea level. Moreover, as the cities have a certain area and might not be circular, the full extent of the city might not be accounted for, and hence the surface area of coastal ecosystems that may contribute to risk mitigation may be considered as a conservative estimate.

However, despite those limitations, the method gives an estimation of the possibilities of nature-based risk mitigation in front of globally distributed and highly populated coastal cities.

The four conditions retained to identify the suitable locations for tidal wetland creation in the likely area of storm surge propagation are (1) elevation below mean high tide, (2) absence of existing tidal wetlands, as defined by the Global distribution of mangroves (Giri et al., 2011) and the Global distribution of saltmarshes (McOwen et al., 2017) datasets, (3) location outside the urbanized area, as defined by the GLCNMO dataset and (4) a population density lower than 50 inhabitants per square kilometre, as defined by the LandScan 2013 Global population database. The logical steps are presented in Figure 1.

The value of 50 inhabitants per square kilometre is an arbitrary value based on the literature (McGranahan et al., 2006; Neumann et al., 2015). Scenarios with different population threshold values (20, 35 and 50 inhabitants/km²) were explored and are presented in Appendix A. The comparison of the results based on the three population density thresholds is showing a general increase of the average surface area available for tidal wetlands development of 9 ± 34 km² between the lowest and highest thresholds, while keeping the relative difference between the cities very similar. It is important to note that the considered threshold remains high and represent a theoretical exploration of the potential to restore tidal wetlands. In practice, the displacement of even one inhabitant may lead to the impossibility to restore or create tidal wetlands in the area.

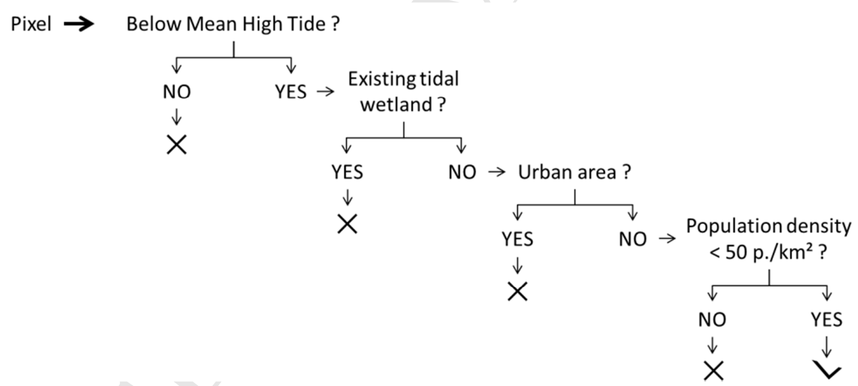

Figure 1 Representation of the logical steps for the selection of the pixels suitable for tidal wetlands restoration or creation

The areas available for tidal wetlands creation were compared to the current land cover based on the Global Land Cover by National Mapping Organization dataset. This was done in order to assess the present land use types in the areas identified as potentially available for tidal wetlands creation or restoration. This is relevant information, because, as we discuss in the discussion section, the feasibility and engineering measures needed for wetland creation depends on the present land use types. For example, tidal wetland creation on present agricultural fields or in present water bodies necessitates different types of ecological engineering measures. For each city, the different land covers in the areas potentially available for tidal wetlands creation or restoration were defined based on four categories, namely (1) the **vegetated** areas grouping the land cover classes of broadleaf and needle leaf evergreen and deciduous forest, mixed forests, tree open areas, shrub, herbaceous, herbaceous with sparse trees or shrub, sparse vegetation and wetlands areas; (2) the **cropland and paddy fields** areas corresponding to the cropland, cropland with other vegetation mosaic and paddy fields areas; (3) the **bare land** corresponding to the bare consolidated or unconsolidated land, and (4) the **water** areas corresponding to the water bodies (i.e. pounds or lakes and areas of coastal water considered inland following the delineation of the country's limits).

Due to the global scale of the different datasets, the vegetated areas include areas defined as wetlands and mangroves in the Global Land Cover dataset. This is because the use of different global-scale datasets implies limitations in local data accuracy or artefacts and overlap of features. In this analysis, the extent of salt marshes and mangrove forests is based on the *Global distribution of Mangroves* (Giri et al., 2011) and the *Global distribution of saltmarshes* (McOwen et al., 2017) *(see section 2.1)* that are considered as more accurate than the Global Land Cover dataset.

3 Results

3.1 Area below mean high tide for tidal wetlands restoration

The zone that sits below mean high tide was defined for every studied city and divided into three categories, (1) the existing tidal wetlands, (2) the area potentially available for tidal wetland restoration or creation (i.e. non-urban area with less than 50 inhabitants/km²) and (3) the area not available for tidal wetland creation (i.e. urban areas or with more than 50 inhabitants/km²) (Figure 2).

The cities with the largest areas below mean high tide are Hamburg in Germany (1 400 km²), Guangzhou in China (1 120 km²), Ho Chi Minh City in Vietnam (860 km²), Rotterdam in The Netherlands (780 km²) and Guayaquil in Ecuador (560 km²). At continental scale, the largest zones below mean high tide are found in European cities (Table 1) where the existing tidal wetlands are scarce (maximum of 12 km² in front of London in the UK). Asia and North America also present large areas below mean high tide with averages and standard deviations of 128 ± 201 km² and 80 ± 96 km² respectively. In general the areas located below mean high tide are smaller in South America and Oceania.

The cities that have the highest potentially available area for tidal wetlands creation are Hamburg in Germany (880 km²), Guayaquil in Ecuador (400 km²) and Tianjin in China (230 km²) (Figure 3), while eleven cities have no available space for tidal wetlands creation (Figure 4). Over the 135 studied cities, the available area for tidal wetlands is 34 ± 90 km² (average ± standard deviation), with large variations between the continents (Table 1). Oceania and Africa present the lowest available surface area for tidal wetlands development, followed by Asia (Table 1). Values for North and South America are slightly higher, while Europe has the largest averaged surface area available for tidal wetlands development (Table 1).

Table 1 Values of the minimum, maximum, average and standard deviation of surface areas below mean high tide and surface areas potentially available for tidal wetland restoration or creation for each continent.

	Area below mean high tide					**Area potentially available for tidal wetlands restoration or creation**				
Continent	Min (km^2)	Max (km^2)	City	Average (km^2)	Standard Deviation	Min (km^2)	Max (km^2)	City	Average (km^2)	Standard Deviation
Africa	1.9	490	Alexandria	44.4	106.1	0	100	Alexandria	11.7	22.2
Asia	1.4	1 120	Guangzhou	127.7	201.1	0	230	Tianjin	27.9	40.1
Europe	5.3	1 400	Hamburg	173.7	355.1	0.9	880	Hamburg	73.3	199.6
North America	4.7	340	New Orleans	80.4	95.5	0	180	Portland	39.2	55.8
South America	4.1	560	Guayaquil	67.1	130.0	1.0	400	Guayaquil	41.0	93.9
Oceania	9.7	42	Sydney	21.1	10.5	2.7	10	Adelaide	5.6	2.5

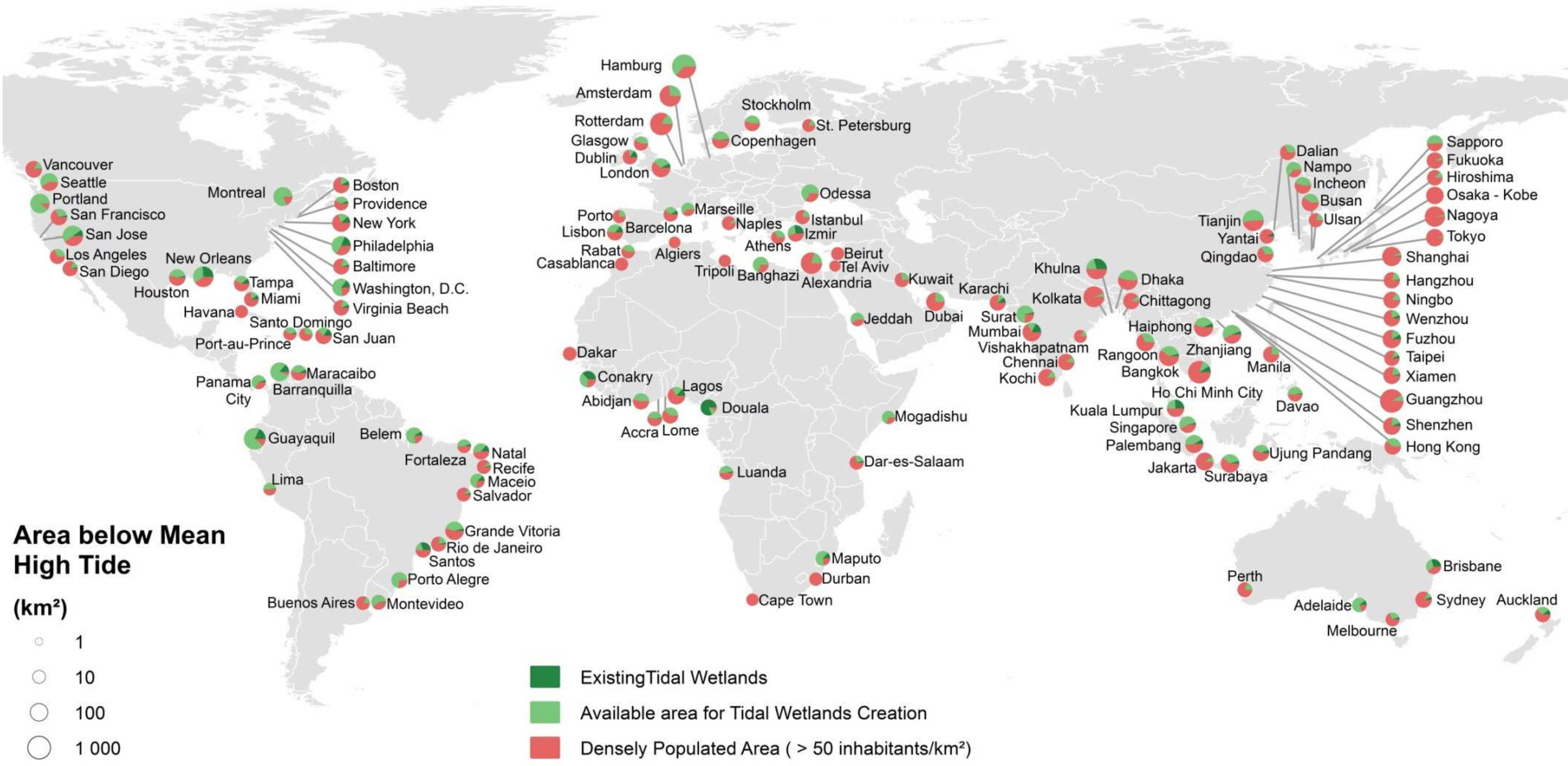

Figure 2 Surface area located below mean high tide in front of every city (size of the symbol), categorized to surface areas occupied by (1) the existing tidal wetlands, (2) the area potentially available for tidal wetlands creation (i.e. non-urban areas with less than 50 inhabitants/km²) and (3) the area not available for tidal wetlands creation (i.e. urban areas or with more than 50 inhabitants / km²) (colour of the symbol)

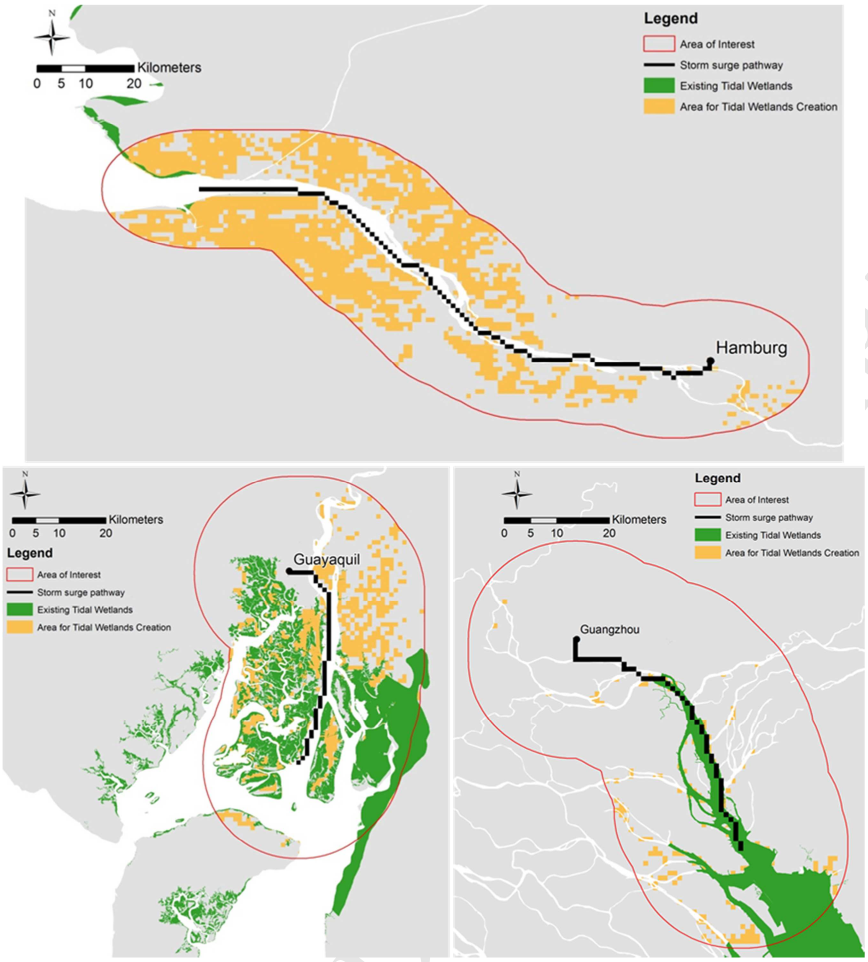

Figure 3 Representation of the three cities, i.e. Hamburg (880 km²), Guayaquil (400km²) and Guangzhou (97 km²), with the largest areas potentially available for tidal wetlands creation or restoration based on the analysis.

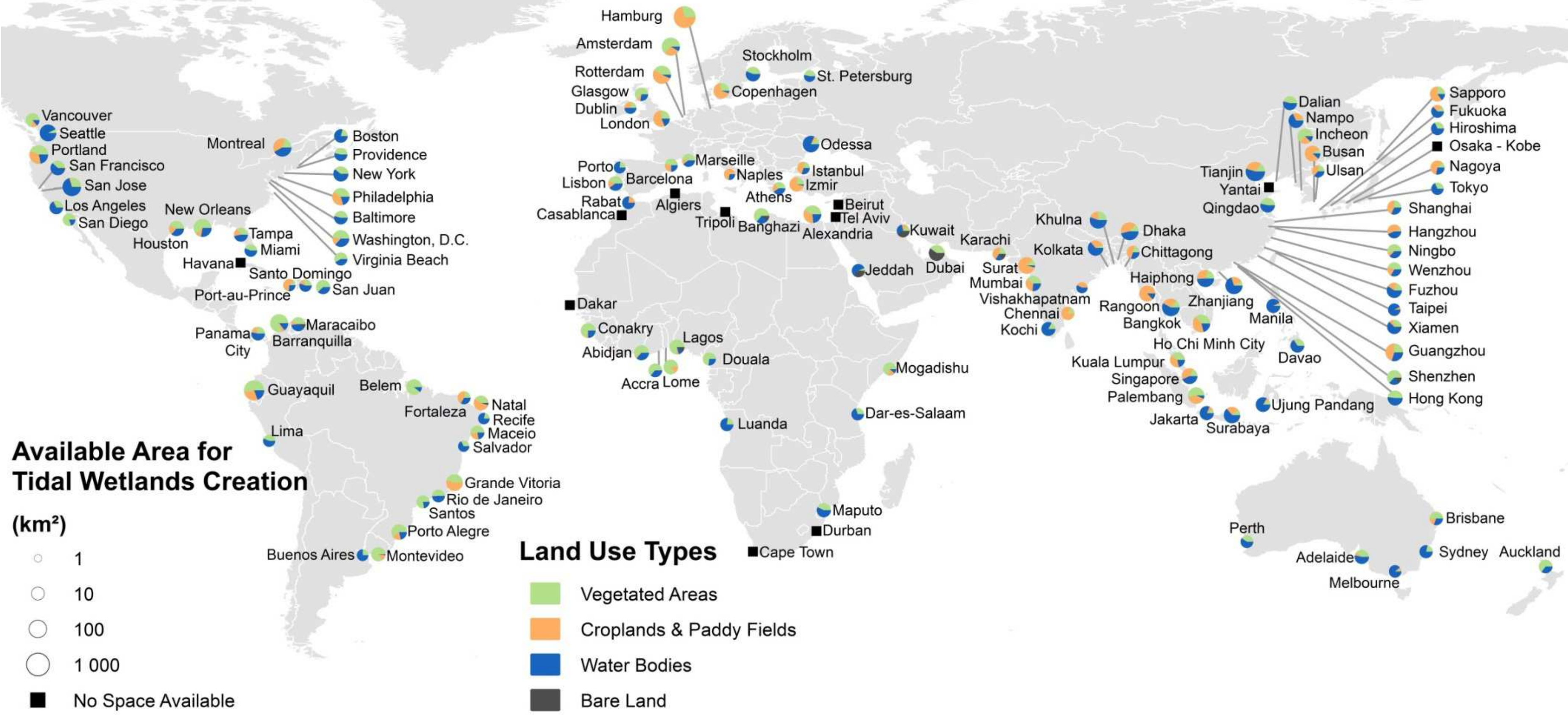

Figure 4 Surface area potentially available for tidal wetlands creation (size of symbols) and the current land use in those areas (colours of symbols).

When looking at the current land use types within the areas that are identified as potentially suitable for tidal wetlands restoration or creation, the most dominant land use types are cropland and paddy fields, mostly in European, Asian, South and North American cities (Figure 4 and Figure 5), in combination with vegetated areas (1 670 km² and 1 620 km² respectively, or 36 % and 35 % of the total potentially available area). The cities having the largest cropland and paddy fields areas below mean high tide are Hamburg in Germany (620 km² of cropland, 71 % of the potentially available surface area for tidal wetlands restoration or creation), Guayaquil in Ecuador (70 km² of croplands and 41 km² of paddy field, 28 %), Tianjin in China (67 km² of cropland, 30 %) or Rotterdam in The Netherlands (66 km² of cropland, 52 %) (Appendix B). Croplands are mainly found in North America and Europe, while paddy fields are mainly found in Asia, and occasionally on large surfaces in other continents such as in Guayaquil (Ecuador). For most of the cities, part of the potentially available area for tidal wetlands restoration or creation is currently defined as water bodies by the Land Cover Dataset; the largest areas are found in Asian and North American cities (Figure 5), as in Tianjin (China) with 126 km² and San Jose (USA) with 110 km². Bare land represents a really small share of the potentially available area for tidal wetlands restoration or creation for all the continents, with less than 1 km² on average (Figure 5).

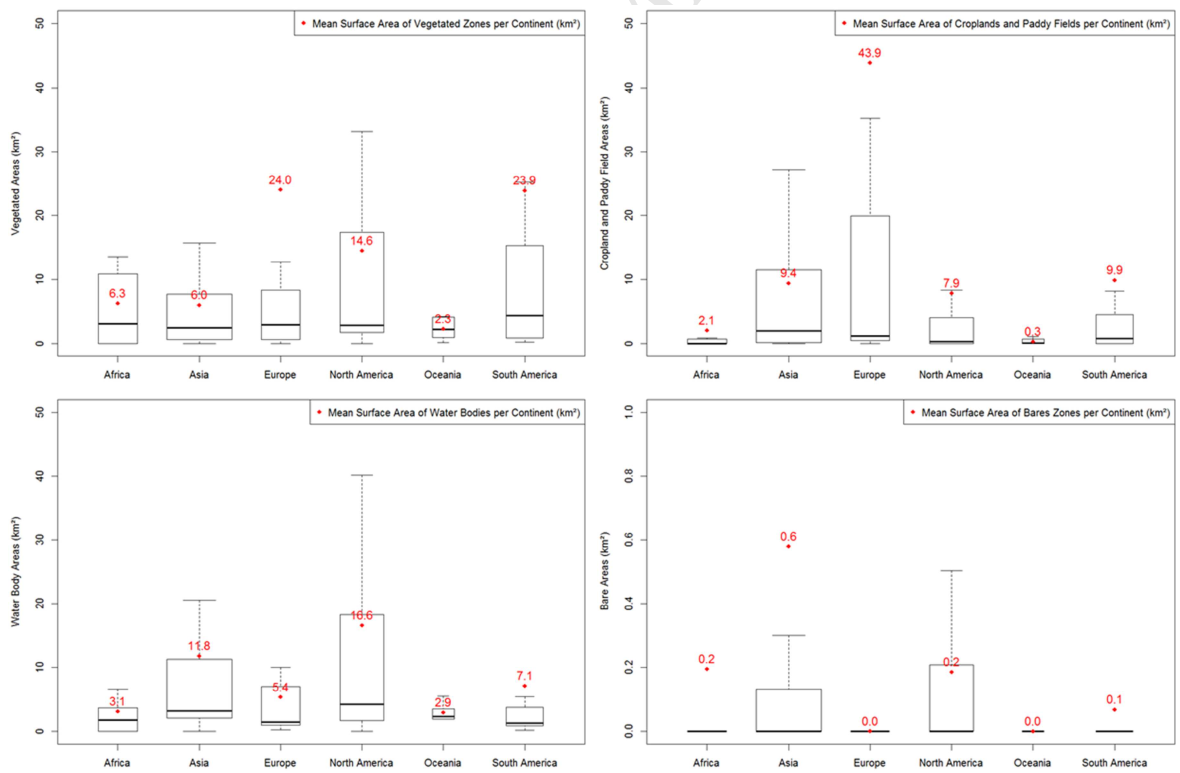

Figure 5 Comparison of the land use types in the potentially available areas for tidal wetlands creation in front of the 135 studied cities, categorized per continent. The width of the boxes corresponds to the square root of the number of cities per continent. Note that Y-axes have different scales.

4 Discussion

Nature-based mitigation of coastal flood risks, by conserving, restoring or creating coastal ecosystems that are known to attenuate the impacts of sea level rise, storm surges, wind waves and shoreline erosion, is increasingly proposed as a sustainable, cost-efficient strategy to mitigate and adapt to increasing coastal flood risks (Morris et al., 2018; Narayan et al., 2016; van Wesenbeeck et al., 2014; Vuik et al., 2016). Although this concept is increasingly adopted in scientific literature, so far there are no global-scale studies that explored the potentials of tidal wetland restoration or creation in front of flood-exposed coastal cities, while such an analysis may contribute to increase global interest in the implementation of nature-based risk mitigation programs into policy on coastal zone management. We presented here a methodology for a global-scale analysis which identified the potentially suitable and available areas for tidal wetlands creation within the likely area of storm surge propagation from the sea towards the 135 studied cities. The large variations found in this available area between the cities mainly reflect the differences in geomorphological setting, population settlement and land use history within the likely area of storm surge propagation.

Note that the areas identified as potentially available for tidal wetlands restoration or creation are theoretical areas based on parameters that do not include the socio-economic limitations of the implementation of nature-based strategies. In practice, socio-economic factors, such as governance capacity, political stability, and financial resources, highly influence the possibility and the success of tidal wetlands restoration or creation (Darwiche-Criado et al., 2017; Hartman and Cleveland, 2018; Perillo et al., 2009). Although, the socio-economic factors that facilitate or hinder the development of nature-based projects are poorly studied, the societal support and acceptance of tidal wetlands restoration or creation is at least as important as the financial and ecological considerations (Hartman and Cleveland, 2018; Perillo et al., 2009; Suman, 2019). In countries where the government's regulation capacities are weak for example, the conservation and restoration of natural areas is often highly difficult; the natural areas are freely accessible by the public with little monitoring of the different activities taking places. In general, the local communities are willing to support the development of such nature-based strategies, but not at their expense (e.g. livelihood reduction, land loss...), which can seriously hamper the possibilities of tidal wetlands restoration or creation (Perillo et al., 2009; Suman, 2019).

The geomorphic setting is a first factor that may explain the differences in potential areas for tidal wetland creation. Cities located along deltaic or estuarine channels or adjacent to bays and lagoons (i.e. having longer coastlines within their zone of likely storm surge propagation), have greater low-lying zones and subsequently larger potentially suitable areas for development of salt marshes and mangrove forests (Grobicki et al., 2016; McOwen et al., 2017; Pennings and Bertness, 2000; Scott et al., 2014; Spalding et al., 1997; Wolanski and Elliott, 2015). This can be illustrated, for instance, by the city of Hamburg (Germany) located adjacent to the Elbe estuary at 110 km from the estuary mouth, for which our analysis identified that the zone influencing the propagation of a storm surge includes a record area below mean high tide of 1 400 km² and a potentially available area for tidal wetlands development of 890 km² (see Appendix D, Figure D1). Another, tropical example, is Guayaquil (Ecuador), located alongside the main river channel within the large Guayas river delta at 60 km from the open sea, for which our analysis indicates an area below mean high tide of 560 km² and an area potentially suitable for tidal wetlands development of 400 km².

Besides geomorphology, the potentially available area for tidal wetland creation is influenced by the densely populated areas in front of the cities, as shown in Figure 2Figure 2. It highlights the fact that the population settlement in the low-lying zone can hamper the creation of new tidal wetlands. This can be illustrated by Guangzhou in China, for instance, where, on the 1 100 km² located below mean high tide (a huge area due to the location of Guangzhou in the large Pearl river delta), only 8 km² are at present occupied by existing tidal wetlands and 97 km² could be potentially available for new tidal wetlands creation, while 91 % of the area is occupied by densely populated urban zones (Figure D2). Indeed the city of Guangzhou is part of a huge agglomeration occupying most of the delta. The situation is similar for a number of cities as Nagoya, Kolkata or Shanghai (Figure D3), located especially in Asia (China, Japan, India), where several of such large deltaic agglomerations developed with little open space left for tidal wetlands restoration or creation (Figure 2).

The geomorphic and population factors together suggest that in the future, the reduction of area between the open sea and the city, by marine transgression through sea level rise and shoreline erosion, combined with an increasing population in the coastal zone, i.e. coastal squeeze, may limit the potential development of new tidal wetlands (Pontee, 2013; Rupp-Armstrong and Nicholls, 2007).

Land use history also plays a role. Over the world, the human influence on the coastal areas and particularly around coastal cities led to the degradation, destruction and conversion of hundreds of square kilometres of tidal wetlands, leaving cities with few or no remaining tidal wetlands (Airoldi and Beck, 2007; Alongi, 2008; Duke et al., 2007; He and Zhang, 2001). This is the case for many European cities in an estuarine or deltaic setting, which present the largest areas of croplands located below mean high tide level (Figures 4 & 5). Examples include Hamburg in Germany, London in the UK, or Rotterdam and Amsterdam in The Netherlands, where the embankment and drainage of coastal, estuarine and deltaic wetlands into so-called "polders", mostly for agriculture purposes, dates back to the Middle Ages and even earlier (Airoldi and Beck, 2007; Hansen, 2015; Hatvany, 2003; Hoeksema, 2007; Reise, 2005). As such, for instance almost 2 500 km² of salt marshes were reclaimed along the Elbe estuary between Hamburg and the sea (Figure D1) (de Haas et al., 2018; Hamburg Port Authority, 2006; Hansen, 2015; Pierik et al., 2017; Reise, 2005; Vos, 2015). Identified as a currently large “polder” (880 km²), especially consisting of cropland (620 km²), our analysis identified this area as potentially available for tidal wetland creation. In China, from the 1960s, the embankment of mangrove areas into rice fields, aquaculture ponds or areas for industrial and urban development, resulted in the loss of nearly 60 % of the Chinese mangroves; at a local scale, a city like Hong Kong lost 85 % of its original mangrove forests (Li and Lee, 1997; Meng et al., 2017). In other (sub-)tropical areas, historical land use changes from mangroves into human land use, often aquaculture ponds, is widespread (Chen et al., 2017; Deb and Ferreira, 2015; Meng et al., 2017; Scott et al., 2014; Valiela et al., 2001; Zhu et al., 2016). In Guayaquil (Ecuador), for instance, the potentially available area for tidal wetlands restoration coincides with present-day aquaculture ponds that were created over the past decades in former mangrove areas in the Guayas river delta (Delgado, 2013; Parés-Ramos et al., 2013).

The effectiveness of the creation or restoration of tidal wetlands strongly depends on the current land use both in terms of the success of the tidal wetlands development and of the future increase in coastal protection (Lewis and Brown, 2014; Zhao et al., 2016). Therefore, locations where we identified possible areas for tidal wetland creation may necessitate different measures and may experience different rates of success, depending on their present land use type (Figure 4). Firstly, the restoration or creation of tidal wetlands necessitates the presence of several hydro-geomorphic conditions. For example, establishment of tidal wetland vegetation may be limited when tidal flooding is too excessive and soil drainage during ebb is poor. Consequently tidal wetland vegetation is usually only able to grow in the upper portion of the intertidal zone, where soil and topographic conditions allow good drainage. As such, the establishment of tidal wetland vegetation where land use currently consists of water bodies (Figure 4), can be very difficult as the water bodies should be drained or elevated to create a tidal regime allowing the development of the wetland’s vegetation (Haltiner et al., 1997). Areas that are currently used as agricultural or paddy fields for food production (Figure 4), and that are currently protected from tidal flooding by structures such as dikes, dams or levees, may be converted to tidal wetlands by introducing a tidal regime, but also here, care should be taken that the elevation, tidal inundation regime and drainage is suitable to allow successful wetland vegetation establishment (Beauchard et al., 2011; Maris et al., 2007). Additionally, those agricultural areas may be polluted with fertilizer, leading to high concentrations of nitrate and phosphorus for example. The release of those nutrients during the tidal cycles can lead to severe problems, such as degradation of water quality (Ardón et al., 2017; Shoemaker et al., 2017). Similarly, the pollution in industrial or urbanized soils also implies limitations for the restoration or creation of tidal wetlands.

Secondly, the effectiveness of tidal wetland creation for nature-based storm surge mitigation also depends on the present land use type. Tidal wetlands reduce the height of storm surges due to their bed roughness and the friction exerted by their vegetation on the water column; the latter is dependent on amongst others the vegetation density, height and stiffness (Shepard et al., 2011; Sutton-Grier et al., 2015; Wamsley et al., 2009). For tidal wetlands, salt marsh vegetation (consisting of grasses, herbs, and low shrubs) exerts less friction than mangrove forests, yet they generate more friction on propagating storm surges than agricultural fields (i.e. croplands or paddy fields) or bare soil surfaces (Mattocks and Forbes, 2008; Passeri et al., 2018; Wamsley et al., 2009). The conversion of agricultural fields and bare soil surfaces to tidal wetlands will then increase the friction on landward propagating storm surges and hence will increase the attenuation rate of storm surges. On the other hand, forested areas have a friction comparable to mangrove forests (Mattocks and Forbes, 2008), making their conversion less interesting in terms of storm surge attenuation. Nonetheless, the propagation of storm surges through freshwater plants implies a salinity intrusion that is not well managed by a number of freshwater species (Carter et al., 2018; Middleton, 2016; Stanturf et al., 2007). Thus, keeping freshwater vegetation in order to protect the coastal population and areas from storm surges may be inefficient, as the resilience of the freshwater vegetation to salinity intrusion is uncertain. Subsequently, in areas with a high intensity and frequency of storm surges, the conversion of freshwater vegetation to saltwater vegetation (i.e. tidal wetlands) might be seen as a valuable nature-based strategy to increase the resilience of the vegetation to storm surges (Middleton, 2016).

Local knowledge on how and where to restore and recreate tidal wetlands is growing and highlights this unique character of each local setting and the importance of understanding amongst others the different hydrodynamic, geomorphic and ecological characteristics of the specific area that influence the success of wetland creation (Balke and Friess, 2016; Elliott et al., 2016; Oosterlee et al., 2018; Simenstad et al., 2006). Depending on the situation, the restoration or creation of tidal wetlands may necessitate active re-conversion of the area by restoring the natural hydrodynamic and subsurface hydrological flow patterns, reshaping the topography of the area, restoring the sediment supply or planting the appropriate vegetation for example, as in old aquaculture ponds, agriculture fields or in more urbanized areas (Garbutt et al., 2006; Lawrence et al., 2018; Lee et al., 2012; Lewis and Brown, 2014; Spalding et al., 2014), while in other places, wetland vegetation may spontaneously re-colonize the area without much intervention once the appropriate hydrodynamic and bio-geomorphic conditions are set (Eertman et al., 2002; Pethick, 2002). Although restoration projects can be successful, the restored area will often not recreate a pristine environment in terms of plant diversity, topography or hydrology (Bullock et al., 2011; Elliott et al., 2016; Hobbs et al., 2009; Lawrence et al., 2018; Spalding et al., 2014). However, restoration or creation of tidal wetlands can be successful in a large variety of environments, and are expected to be able to deliver ecosystem services such as water quality regulation, carbon sequestration and protection against wind waves and storm surges (Bullock et al., 2011; Hobbs et al., 2009; Spalding et al., 2014).

5 Conclusion

There is a pressing need for adaptation of the coastal zone to increasing threats due to climate change (increased frequency of storm surges, sea level rise...) and due to socio-economic changes (increasing coastal population density, coastal megacities...). The development of nature-based and hybrid protection structures for mitigation of coastal flood and erosion risks is increasingly regarded as a sustainable and cost-efficient strategy over the long term.

Our study reveals that on the 135 studied cities, 60 % (8 300 km²) of the area below mean high tide is urbanized or densely populated and 34 % (4 600 km², distributed over 124 cities) is potentially available for tidal wetlands restoration or creation. Key factors influencing this potentially available space are the geomorphology as well as the population density in the coastal area in front of the city. The land use in the potentially available area for tidal wetlands restoration or creation is mainly composed of croplands, paddy fields, water bodies and vegetated areas, and influences the effectiveness of tidal wetland creation for nature-based flood risk mitigation.

The analysis, which is based on global-scale datasets, is providing first estimations regarding the globally available areas for tidal wetlands restoration or creation. The development of specific successful restoration projects should necessitate further local- to regional-scale analyses including a combination of scientific, socio-economic, policy and management approaches. Local studies based on more high-resolution datasets are needed to identify the potentially available area for tidal wetlands restoration or creation at specific locations. It is anticipated that this study can promote global awareness of the possibility to restore or create tidal wetlands as nature-based risk mitigation in front of flood-exposed coastal cities around the world, and to lead local studies on the feasibility of such nature-based projects.

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Appendix A

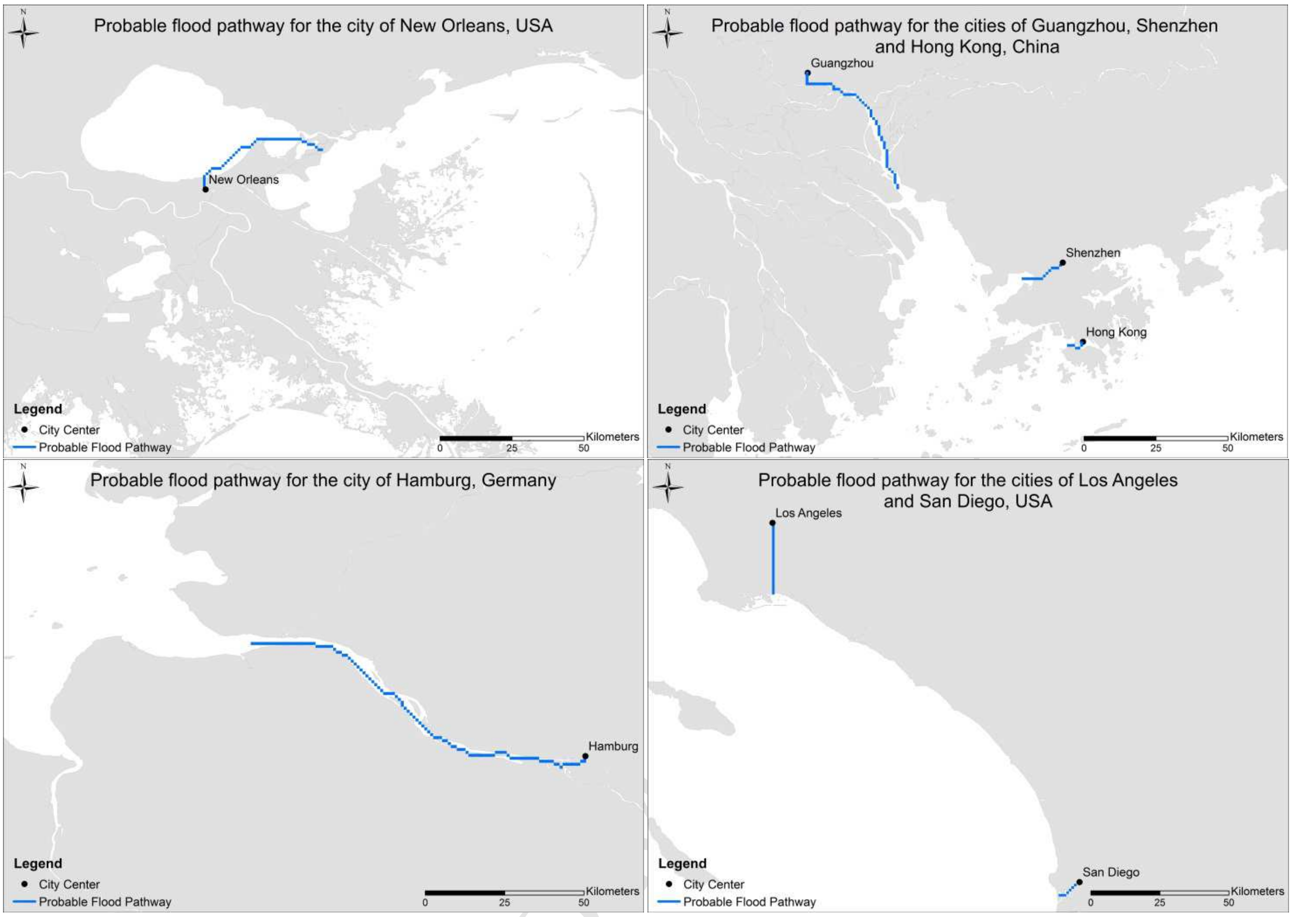


Figure A1 Representation of the probable flood pathways, i.e. the path preferably followed by the storm surge between the open sea and the city, accounting for the friction exerted by the land area and the water bodies, as calculated in the analysis for seven cities located at the end of long and small river channels (Guangzhou, Hamburg and Shenzhen), in bays (New Orleans), or at the coast (Hong Kong, Los Angeles, San Diego).

Appendix B

Table B1 Values of the surface area available for tidal wetlands creation or restoration based on the three population threshold tested (20, 35 and 50 inhabitants/ km²)

Cities	Continent	Area for tidal wetlands development (km²) according to the population thresholds		
		< 20 inhab/km²	< 35 inhab/km²	< 50 inhab/km²
Abidjan	Africa	14.96	16.75	18.07
Accra	Africa	6.92	6.92	6.92
Adelaide	Oceania	9.95	9.95	10.32
Alexandria	Africa	86.78	96.33	101.37
Algiers	Africa	0.00	0.00	0.00
Amsterdam	Europe	71.10	97.35	123.75
Athens	Europe	2.74	2.74	2.98
Auckland	Oceania	5.02	6.51	6.54
Baltimore	North America	3.81	4.86	5.47
Banghazi	Africa	19.63	21.08	21.08
Bangkok	Asia	86.32	89.51	95.83
Barcelona	Europe	2.88	3.10	4.30
Barranquilla	South America	105.84	113.79	118.37
Beirut	Asia	0.00	0.00	0.00
Belem	South America	19.73	25.89	27.83
Boston	North America	4.84	5.30	6.05
Brisbane	Oceania	2.09	5.47	6.14
Buenos Aires	South America	0.41	0.65	1.40
Busan	Asia	22.59	25.47	27.26
Cape Town	Africa	0.00	0.00	0.00
Casablanca	Africa	0.00	0.00	0.00
Chennai	Asia	4.91	4.91	4.91
Chittagong	Asia	3.86	4.21	4.21
Conakry	Africa	15.09	15.67	15.67
Dakar	Africa	0.00	0.00	0.00
Dalian	Asia	5.95	6.96	8.01
Dar-es-Salaam	Africa	2.45	2.71	2.71
Davao	Asia	6.21	6.38	7.07
Dhaka	Asia	97.28	102.44	107.20
Douala	Africa	4.31	4.38	4.38
Dubai	Asia	22.88	29.08	31.20

Dublin	Europe	1.23	1.23	1.85
Durban	Africa	0.00	0.00	0.00
Fortaleza	South America	3.58	3.82	3.82
Fukuoka	Asia	2.19	2.73	3.25
Fuzhou	Asia	13.98	15.46	19.51
Glasgow	Europe	3.40	4.51	5.13
Guangzhou	Asia	46.91	70.93	97.43
Guayaquil	South America	323.94	372.53	399.43
Haiphong	Asia	49.50	65.04	74.72
Hamburg	Europe	505.26	747.58	880.84
Hangzhou	Asia	2.70	4.20	7.37
Havana	North America	0.00	0.00	0.00
Hiroshima	Asia	3.15	3.35	3.83
Ho Chi Minh City	Asia	49.61	68.84	98.08
Hong Kong	Asia	14.49	17.42	17.59
Houston	North America	16.25	17.51	19.19
Incheon	Asia	17.41	20.84	21.85
Istanbul	Europe	1.00	1.28	2.76
Izmir	Europe	8.35	10.36	11.14
Jakarta	Asia	5.61	8.99	8.99
Jeddah	Asia	3.32	3.74	4.54
Karachi	Asia	4.04	4.04	4.04
Copenhagen	Europe	22.96	29.30	29.78
Khulna	Asia	50.55	63.30	73.15
Kochi	Asia	4.66	6.79	8.82
Kolkata	Asia	15.10	20.24	23.72
Kuala Lumpur	Asia	12.27	15.93	17.49
Kuwait	Asia	2.73	3.70	3.70
Lagos	Africa	13.66	14.16	15.58
Lima	South America	1.64	2.04	2.04
Lisbon	Europe	9.77	10.33	11.85
Lomé	Africa	10.47	12.17	13.92
London	Europe	21.84	35.54	48.39
Los Angeles	North America	3.92	4.60	6.28
Luanda	Africa	4.14	4.15	4.51
Maceio	South America	6.95	7.97	8.28
Manila	Asia	5.61	5.81	8.61

Maputo	Africa	10.05	10.58	10.58
Maracaibo	South America	8.49	9.08	9.45
Marseille	Europe	3.38	3.94	3.94
Melbourne	Oceania	2.16	2.60	2.67
Miami	North America	2.22	2.85	2.85
Mogadishu	Africa	4.35	4.35	4.35
Montevideo	South America	9.35	10.06	10.76
Montreal	North America	132.72	140.14	143.97
Mumbai	Asia	12.75	16.23	20.71
Nagoya	Asia	6.50	7.62	9.18
Nampo	Asia	11.67	14.94	16.49
Naples	Europe	0.84	0.90	0.90
Natal	South America	8.82	13.81	14.78
New Orleans	North America	107.50	112.85	116.02
New York	North America	13.27	17.13	18.23
Ningbo	Asia	5.61	8.16	8.16
Odessa	Europe	49.85	50.45	50.63
Osaka	Asia	0.00	0.00	0.00
Palembang	Asia	37.98	39.92	45.86
Panama City	North America	6.94	6.94	6.94
Perth	Oceania	2.85	3.24	3.24
Philadelphia	North America	71.27	77.58	80.68
Port-au-Prince	North America	2.27	2.27	2.27
Portland	North America	165.62	177.76	184.20
Porto	Europe	0.00	1.33	1.33
Porto Alegre	South America	20.88	23.43	25.26
Providence	North America	2.86	3.08	3.65
Qingdao	Asia	11.94	12.25	13.21
Rabat	Africa	0.00	0.00	3.14
Rangoon	Asia	4.81	10.28	33.63
Recife	South America	1.07	1.07	1.07
Rio de Janeiro	South America	3.05	3.81	3.81
Rotterdam	Europe	66.38	93.96	126.62
Salvador	South America	0.57	0.96	0.96
San Diego	North America	2.39	2.95	2.95
San Francisco	North America	8.47	11.85	13.26
San Jose	North America	142.42	152.62	159.03

San Juan	South America	8.69	9.43	9.93
Santo Domingo	North America	2.09	2.10	2.10
Santos	South America	3.39	4.18	4.45
Sapporo	Asia	10.35	13.13	18.15
Seattle	North America	49.40	51.48	53.46
Shanghai	Asia	6.91	8.30	9.68
Shenzhen	Asia	6.94	12.07	13.74
Singapore	Asia	25.23	27.48	28.52
Stockholm	Europe	10.65	11.86	12.55
St Petersburg	Europe	0.22	0.42	0.92
Surabaya	Asia	30.56	44.14	46.19
Surat	Asia	50.42	53.29	53.29
Sydney	Oceania	2.16	4.11	4.40
Taipei	Asia	2.14	2.60	2.60
Tampa	North America	7.71	8.24	8.24
Tel Aviv-Yafo	Asia	0.00	0.00	0.00
Tianjin	Asia	174.07	211.63	233.37
Tokyo	Asia	2.10	2.19	2.31
Tripoli	Africa	0.00	0.00	0.00
Ujung-Pandang	Asia	9.69	13.57	15.07
Ulsan	Asia	1.04	2.04	2.28
Vancouver	North America	1.41	5.10	8.19
Virginia Beach	North America	3.77	4.29	4.34
Vishakhapatnam	Asia	0.00	0.19	0.80
Grande Vitoria	South America	45.10	49.92	55.71
Washington D.C.	North America	48.53	51.70	54.70
Wenzhou	Asia	5.56	5.79	8.36
Xiamen	Asia	9.31	9.38	10.97
Yantai	Asia	0.00	0.00	0.00
Zhanjiang	Asia	60.07	66.27	73.93

Appendix C

Table C1 Surface areas (km²) of the land use types within the potentially available area for tidal wetlands creation or restoration in front of the 135 coastal cities, with the distinction between croplands and paddy fields

City	Continent	Vegetated Areas (km²)	Croplands (km²)	Paddy Fields (km²)	Bare Areas (km²)	Water Bodies (km²)
Abidjan	Africa	10.65	0.86			6.56
Accra	Africa	4.37				2.55
Adelaide	Oceania	4.10	0.70			5.51
Alexandria	Africa	43.87	33.68			23.82
Algiers	Africa					
Amsterdam	Europe	78.51	35.22			10.01
Athens	Europe	1.16	0.59			1.22
Auckland	Oceania	4.24				2.30
Baltimore	North America	2.43	0.12			2.92
Banghazi	Africa	13.57			2.18	5.34
Bangkok	Asia	13.19	21.41	5.75		55.48
Barcelona	Europe	2.09	1.13			1.07
Barranquilla	South America	93.14	8.03			17.21
Beirut	Asia					
Belem	South America	24.73				3.10
Boston	North America	1.28				4.77
Brisbane	Oceania	3.13	1.14			1.87
Buenos Aires	South America	0.34				1.06
Busan	Asia	2.37	14.08	7.59		3.22
Cape Town	Africa					
Casablanca	Africa					
Chennai	Asia	0.83	0.74	3.33		
Chittagong	Asia	0.87	1.75	0.32		1.26
Conakry	Africa	11.09	0.70			3.88
Dakar	Africa					
Dalian	Asia	3.48		0.19		4.34
Dar-es-Salaam	Africa	0.82				1.89
Davao	Asia	2.26	0.19			4.62
Dhaka	Asia	14.06	8.74	34.76	0.79	48.86
Douala	Africa	3.19				1.18
Dubai	Asia	12.48			18.29	0.43

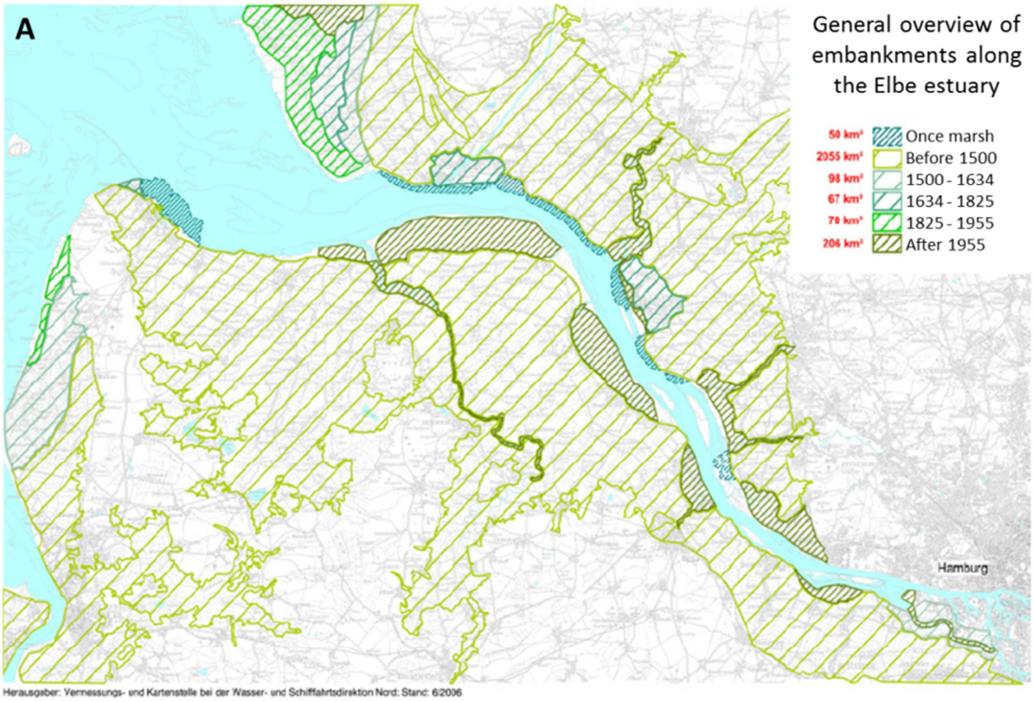
Dublin	Europe	0.42	0.51			0.92
Durban	Africa					
Fortaleza	South America	0.85	1.71			1.26
Fukuoka	Asia	0.38	0.31	0.52		2.03
Fuzhou	Asia	3.85	2.25	1.88		11.52
Glasgow	Europe	3.22	0.48			1.43
Grande Vitoria	South America	25.22	30.36			0.13
Guangzhou	Asia	24.25	28.02	16.11	1.51	27.54
Guayaquil	South America	210.44	70.38	41.33		77.28
Haiphong	Asia	15.71	9.21	13.04		36.76
Hamburg	Europe	251.46	623.92			5.46
Hangzhou	Asia	0.29	1.99	1.88		3.20
Havana	North America					
Hiroshima	Asia	1.22				2.61
Ho Chi Minh City	Asia	35.61	14.67	27.19		20.60
Hong Kong	Asia	8.46				9.13
Houston	North America	8.50	2.95	0.74	0.63	6.37
Incheon	Asia	13.86	3.59	1.65	0.30	2.45
Istanbul	Europe	0.65	1.29			0.82
Izmir	Europe	2.61	7.51	0.67		0.35
Jakarta	Asia	0.83	0.84			7.33
Jeddah	Asia	0.27			1.59	2.68
Karachi	Asia	0.90	0.80	0.96	0.69	0.69
Copenhagen	Europe	8.36	19.93			1.50
Khulna	Asia	18.69	6.36	7.68		40.42
Kochi	Asia	1.52				7.30
Kolkata	Asia	1.47	4.61	2.52		15.12
Kuala Lumpur	Asia	7.06	6.08	0.40	0.17	3.78
Kuwait	Asia	0.57	0.48		1.50	1.15
Lagos	Africa	11.63	0.66		1.53	1.76
Lima	South America	0.89				1.15
Lisbon	Europe	5.25	1.88			4.71
Lome	Africa	11.86	2.01			0.05
London	Europe	12.78	26.43			9.19
Los Angeles	North America	1.86	0.19			4.23
Luanda	Africa	1.02				3.50
Maceio	South America	4.35	2.09			1.84

Manila	Asia	0.34	0.32			7.95
Maputo	Africa	4.55				6.03
Maracaibo	South America	4.09		0.79	0.79	3.78
Marseille	Europe	1.90	0.47			1.57
Melbourne	Oceania	0.13	0.18			2.36
Miami	North America	1.27				1.58
Mogadishu	Africa	3.06	0.78			0.51
Montevideo	South America	9.12	1.41			0.23
Montreal	North America	29.31	54.19			60.47
Mumbai	Asia	8.36	5.44	1.42		5.50
Nagoya	Asia	1.14	2.84	2.60		2.61
Nampo	Asia	0.69	3.80	0.79	0.10	11.12
Naples	Europe		0.65			0.25
Natal	South America	5.89	8.17		0.36	0.36
New Orleans	North America	72.15	10.24	1.32		32.31
New York	North America	7.22			0.36	10.65
Ningbo	Asia	4.41		0.74		3.01
Odessa	Europe	5.92	3.41			41.30
Osaka	Asia					
Palembang	Asia	24.30	16.51	2.00		3.04
Panama City	North America	2.10	0.65	0.51		3.68
Perth	Oceania	1.32				1.93
Philadelphia	North America	33.12	23.29	7.47	0.21	16.59
Port-au-Prince	North America	0.29	1.14	0.25		0.60
Portland	North America	81.38	52.88	9.59	0.20	40.15
Porto	Europe	0.31				1.02
Porto Alegre	South America	15.32	0.74	3.77		5.43
Providence	North America	1.70				1.95
Qingdao	Asia	6.14				7.08
Rabat	Africa		0.71			2.43
Rangoon	Asia	0.47	23.81	5.01	0.45	3.89
Recife	South America	0.23				0.85
Rio de Janeiro	South America	1.94				1.87
Rotterdam	Europe	52.15	66.38			8.09
Salvador	South America	0.31				0.65
San Diego	North America	2.37			0.05	0.53
San Francisco	North America	5.05			0.50	7.71

San Jose	North America	42.79	4.33		1.89	110.02
San Juan	South America	6.18				3.75
Santo Domingo	North America	0.62	0.30			1.17
Santos	South America	3.45				1.00
Sapporo	Asia	4.44	7.71	3.17		2.83
Seattle	North America	3.81			0.42	49.23
Shanghai	Asia	2.48	2.93	1.18		3.09
Shenzhen	Asia	8.65			1.99	3.10
Singapore	Asia	4.75	11.48			12.29
St. Petersburg	Europe	0.42				0.49
Stockholm	Europe	5.52				7.03
Surabaya	Asia	5.19	3.41	8.09		29.50
Surat	Asia	6.27	40.05	4.49	0.38	2.11
Sydney	Oceania	0.94				3.45
Taipei	Asia		0.19			2.40
Tampa	North America	2.87	0.76	0.76		3.85
Tel Aviv-Yafo	Asia					
Tianjin	Asia	35.81	67.22	2.00	2.00	126.34
Tokyo	Asia	0.74				1.57
Tripoli	Africa					
Ujung Pandang	Asia	1.69	0.83			12.56
Ulsan	Asia	0.57	0.63	0.24		0.84
Vancouver	North America	5.92	1.12			1.15
Virginia Beach	North America	2.55				1.79
Vishakhapatnam	Asia		0.34			0.46
Washington D.C.	North America	26.31	7.99	0.35		20.05
Wenzhou	Asia	3.76	1.08	0.74		2.78
Xiamen	Asia	3.04	1.12		0.36	6.45
Yantai	Asia					
Zhanjiang	Asia	5.92	10.68	5.70		51.63

Appendix D

Historical land reclamation from the mouth of the Elbe estuary to the city of Hamburg in Germany and comparison of the estimated land reclamation from our analysis.



Adapted from Hansen, K., 2015. *Ecosystem functions of tidal marsh soils of the Elbe estuary*. University of Hamburg.

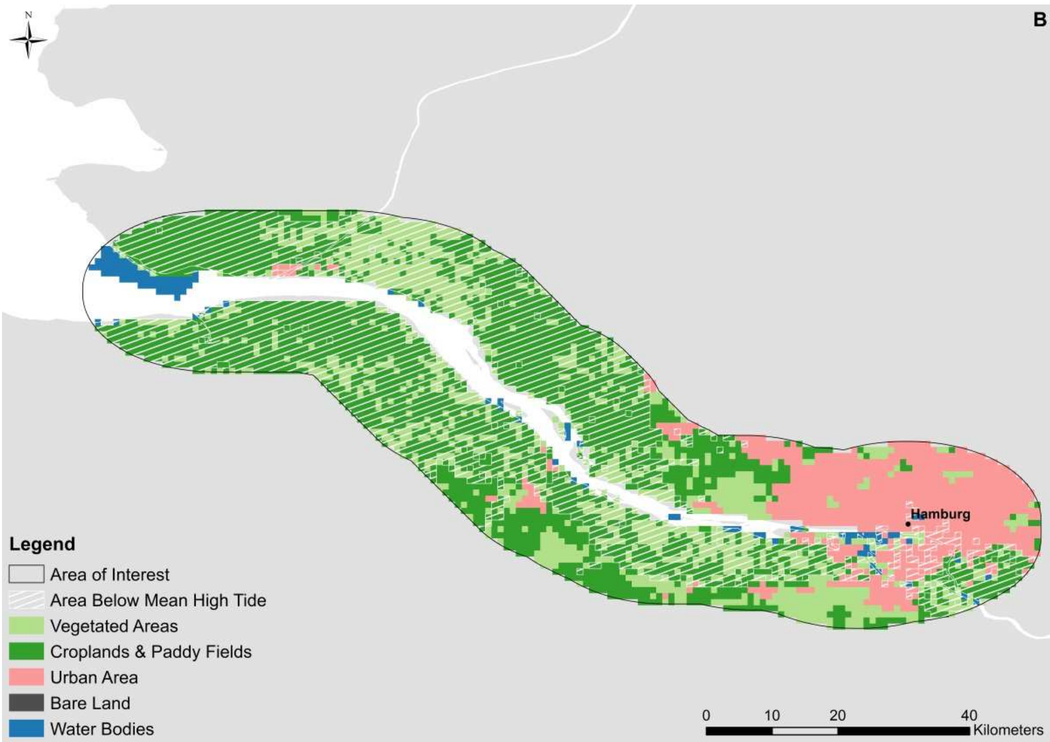


Figure D1 Overview of the extent and period of construction of the embanked areas along the Elbe estuary adapted from Hansen (2015) and (B) representation of the urban, below mean high tide and potentially available for tidal wetlands creation areas in the likely pathway of storm surge propagation towards Hamburg according to our analysis

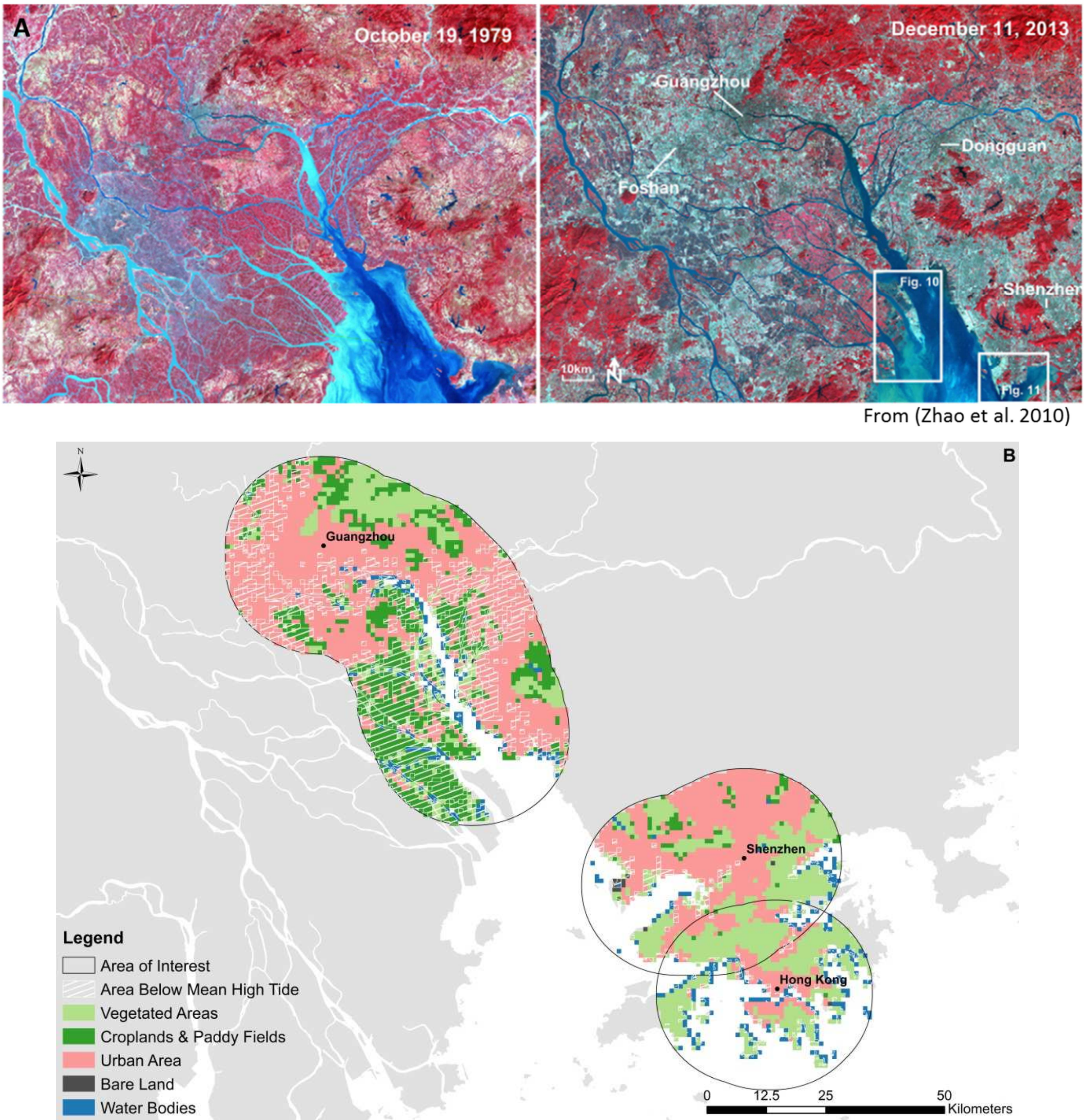


Figure D2 **(A)** Representation of the urban expansion in the Pearl River delta and the city of Guangzhou via infrared-enhanced satellite images for the years 1979 and 2013. The red areas correspond to the delta vegetation, the blue areas to the water bodies and the grey areas to the urbanized land. From (Zhao et al., 2010) **(B)** representation of the areas urbanized, below mean high tide and potentially available for tidal wetlands creation in the likely pathway of storm surge propagation of the cities of Guangzhou, Shenzhen and Hong Kong according to our analysis.

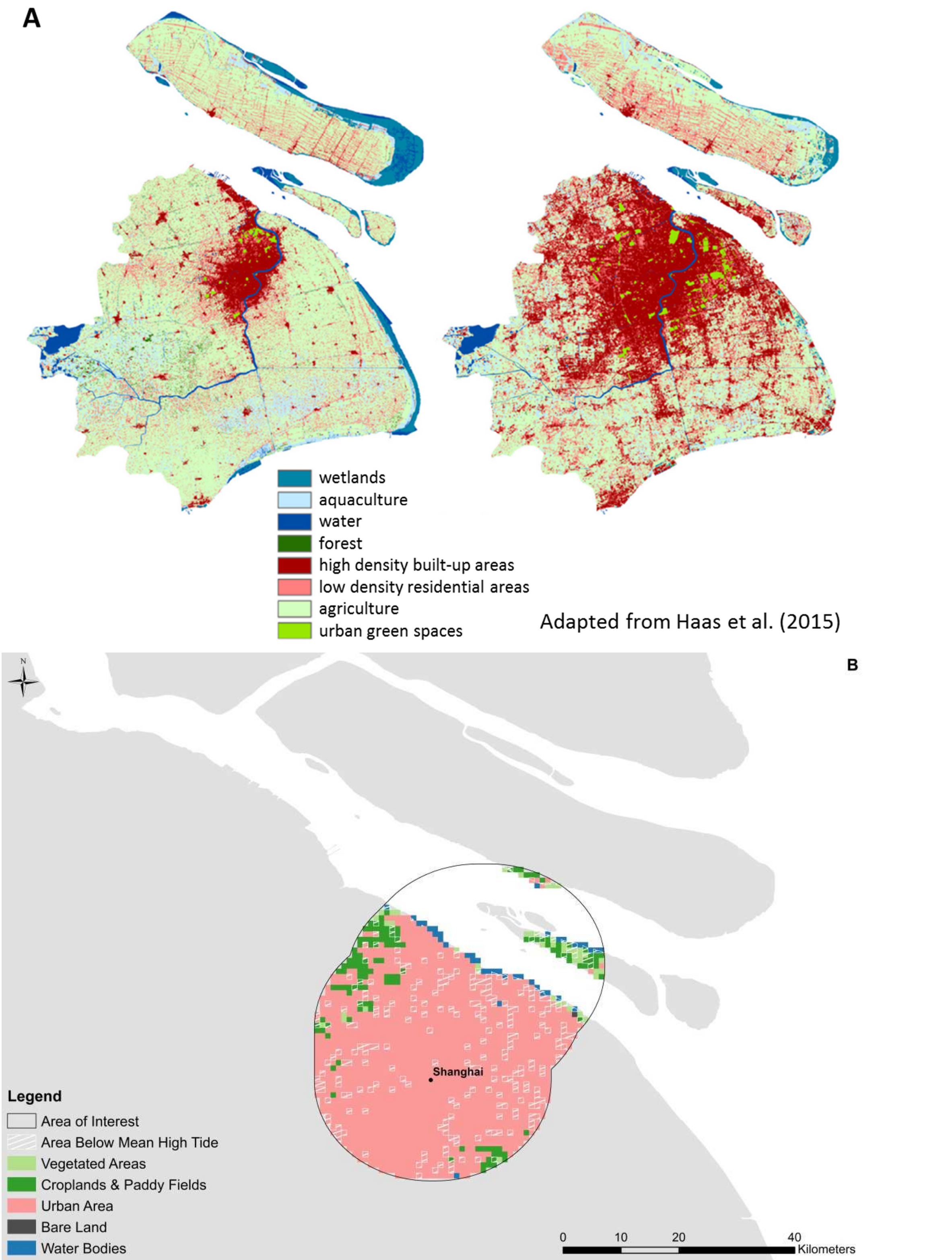


Figure D3 **(A)** Representation of the land uses for the city of Shanghai and the adjacent Yangtze river delta adapted from Haas et al. (2015) **(B)** representation of the areas urbanized, below mean high tide and potentially available for tidal wetlands creation in the likely pathway of storm surge propagation of the city of Shanghai according to our analysis.