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Future losses of ecosystem services due to coastal erosion in Europe

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Abstract

Coastal erosion is a major issue facing Europe that will only worsen under future climate change and the resulting sea level rise. One effect of erosion is the loss of ecosystem services, which are provided by coastal areas, such as provisioning, regulating, habitat, and cultural services. These services can be quantified in monetary terms. Here, we present comprehensive estimates of future decline in coastal ecosystem services due to the erosion of sandy coastlines. We used datasets derived from remote sensing products: a pan-European land cover/use dataset (Corine Land Cover) and new global probabilistic coastal erosion projections constrained by artificial and topographical barriers to erosion. The results include historical changes (2000–2018) and projections under two emission scenarios (RCP4.5 and RCP8.5) for 2050 and 2100 together with uncertainty bounds. We estimate that in 2018, the coastal zone (excluding open sea) included 579,700 km² of habitats generating 494 billion euros of services annually. The future sea-level rise could erode 1.0 % [90 % confidence interval 0.7–1.5 %] of the 2018 area under RCP4.5, and 1.2 % [0.7–2.2 %] under RCP8.5. The decline in services would be even greater: 4.2 % [3.0–6.1 %] under RCP4.5, and 5.1 % [3.3–8.5 %] under RCP8.5. The highest absolute losses would be sustained by salt marshes, while relative losses would be highest in beaches, sands, and dunes. The most affected countries in relative economic terms would be Denmark, Albania, Greece, Estonia, and Finland, but countries such as Germany, the Netherlands, and France would be among those losing the largest share of their coastal ecosystem services. Regional analysis using NUTS 3 regions shows high diversity of the impacts, with many regions along the North Sea and eastern Mediterranean Sea that are heavily affected by coastal erosion-induced loss of ecosystem services. The study highlights the urgency of undertaking mitigation actions.

Keywords: climate change, sea level rise, sandy coasts, land cover, GIS analysis, NUTS regions

1. Introduction

Erosion has been affecting European coastal zones with growing intensity due to climate change-induced increases in storm activity and rising sea levels (Deng et al., 2017; Terefenko et al., 2018a,b, 2019). Analyzing erosion dynamics, including meteorological, geological, and anthropogenic drivers, and its influence on coastal zones has long been a matter of widespread interest (Masselink et al., 2016; Vousdoukas et al., 2018b; Zelaya Wziątek et al., 2019). Among the various impacts of coastal erosion effects (Furmańczyk et al., 2014; Terefenko and Terefenko, 2014; de Sanjosé Blasco et al., 2018), the loss of ecosystem services provided by coastal areas, such as provisioning, regulating, habitat, and cultural services, will be substantial because of their high socioeconomic value (Roebeling et al., 2013). Therefore, there is a growing demand for a wider assessment and valuation of marine ecosystem services, including coastal planning for maritime planning (Börger et al., 2014; Saarikoski et al., 2018; Inácio et al., 2019). In some countries, ecosystem services have been identified as a priority research area (Marine Management Organisation, 2014). Further, there have been efforts toward national assessments of marine ecosystem services through the Platform on Biodiversity and Ecosystem Services supporting the Convention on Biological Diversity. The European Union Biodiversity Strategy until 2020 is also significant in this respect. Meanwhile, rising sea levels, increased storm wave frequency, reduced supply of sediments to the coast, anthropogenic degradation, and transformation of natural coastal areas could lead to substantial irretrievable loss of ecosystem services in the future (Barbier et al., 2011; Ranasinghe, 2016).

The concept of ecosystem services, as we now understand it, is relatively new and emerges from works published at the end of the 20th century (Costanza et al., 1997; Daily, 1997). Although some argue that the concept of ecosystem services represents a very instrumental or utilitarian view of nature by

assigning monetary value to particular narrow benefits of the environment to
30 humans (Thompson and Barton, 1994; McCauley, 2006), the new approach has
generated significant interest among researchers. Many publications have laid
the foundations not only for the theory (La Notte et al., 2017) but also for the
practice of estimating services (Martínez-Harms and Balvanera, 2012; Burkhard
et al., 2013; Milcu et al., 2013; Liqueste et al., 2013a; Martín-López et al., 2014).
35 The work of Costanza et al. (1997) remains the most common reference for
ecosystem service values, as it integrates many pre-existing production assess-
ments of each service for each type of ecosystem and approximated “prices”
of services based on “willingness to pay.” However, the valuation of marine
ecosystem services, including coastal ecosystems such as salt marshes, estuaries,
40 and lagoons, often lags behind the assessment of terrestrial ecosystem services
(Townsend et al., 2018). In effect, relatively few studies have assessed the loss of
value of ecosystem services related to coastal erosion (Ressurreição et al., 2012;
Hynes et al., 2013; Hattam et al., 2015b; Salcone et al., 2016; Mehvar et al.,
2018).

45 So far, there has been no pan-European analysis of potential losses in ecosys-
tem services due to coastal erosion under sea level rise (SLR). The most extensive
study to date by Roebeling et al. (2013) covered merely 10 European countries,
some of which were only partially evaluated, using deterministic projections of
erosion spanning only until the middle of the 21st century. The study indicated
50 that 4500 km² of land eroded between 1975 and 2006, with further erosion of
3700–5800 km² projected until 2050. The two values indicate two Special Re-
port on Energy Scenarios (SRES) climate change scenarios, which are no longer
used. This level of erosion corresponds to a loss of 0.7 billion euro of annual
ecosystem services (at a constant price level of year 2000) in 2006 compared
55 with 1975, and 1.5–2.2 billion euro in 2050 compared with 2006. However, no
information on impacts at the national and local scales was presented.

Vast improvements to spatial data and ecosystem service valuations in re-
cent years now enable a proper continental-scale analysis covering almost all
coasts of Europe. Recently developed probabilistic coastal erosion projections

60 (Vousdoukas et al., 2020) enable the quantification of uncertainty in shoreline
 retreat. The latest Corine Land Cover (CLC) datasets for 2018, created with
 finer satellite images, offer greater quality than previously published products
 (Copernicus Land Monitoring Service, 2020a), serving as an accurate baseline
 for the distribution of ecosystem services. Further, more comprehensive es-
 65 timates of the value of ecosystem services are now available (de Groot et al.,
 2012; Costanza et al., 2014), which improves the precision of assigning economic
 value to habitats. Finally, the regional economic accounts compiled by Euro-
 stat (2020a) allow computing the relative economic impact of ecosystem service
 loss at national and local scales. We hypothesize that coastal erosion will have
 70 highly uneven impacts on ecosystem services across Europe due to the uneven
 extent of shoreline retreat, but locally, those impacts will be very significant.

The aim of this study is to present the first pan-European analysis of ecosys-
 tem service loss that could occur by the end of the century due to SLR-induced
 coastal erosion. It is also the first application of the novel probabilistic coastal
 75 erosion projections from Vousdoukas et al. (2020). In this study we concetrate
 on projected impacts for years 2050 and 2100, including the uncertainty of the
 extent of coastal erosion under two future emission scenarios (RCP – Represen-
 tative Concentration Pathways): RCP4.5 (medium-low emissions) and RCP8.5
 (high emissions) (van Vuuren et al., 2011).

80 **2. Materials and Methods**

The methodology combines multiple datasets to obtain projections of ecosys-
 tem service loss. A flowchart in Fig. 1 presents the main components of the
 study: collection of land cover data (section 2.1), coastal erosion projections
 (section 2.2), ecosystem service valuations (section 2.3), and their combination
 85 into projections of losses and relative economic impacts (section 2.4).

Our study covers 33 countries and dependent territories (Figure 2, which
 cover most European coasts (including all coasts of Turkey and Cyprus). Rus-
 sia and Ukraine were excluded due to the lack of Corine Land Cover data.

Figure 1: Flowchart of the main elements of the methodology.

Additionally, Iceland, Faroe Islands, and northern Norway (Nordland, Troms,
 90 and Finnmark regions) were excluded because they were not covered by coastal
 erosion projections from Vousdoukas et al. (2020).

2.1. Land cover and use

The basis for computing losses to ecosystem services under coastal erosion
 comprised detailed land cover and use data. Here, we used Corine Land Cover
 95 (CLC) data, which is a European database containing information on the phys-
 ical and biological cover of the earth’s surface, including artificial surfaces,
 agricultural areas, forests, (semi-)natural areas, wetlands, and water bodies
 (Bossard et al., 2000). CLC has been used in environmental economics (Weber,
 2007) and for estimating losses of ecosystem services (Roebeling et al., 2013).
 100 The information provided by the CLC dataset is generally based on computer-
 assisted photo-interpretation of satellite images according to a standardized and
 hierarchical nomenclature distinguishing 44 land cover classes. The CLC inven-
 tory was generated for the reference years 1990, 2000, 2006, 2012, and 2018
 (Copernicus Land Monitoring Service, 2020a). The resolution of input satel-
 105 lite images is high and further improved over time from approximately 50 m in
 1990 to 25 m for 2000–2012 and reached 10 m in 2018. The minimum mapping
 unit of CLC is 25 ha, whereas linear features require a width of at least 100
 m to be mapped. The CLC dataset is complemented by CLC-Changes, which
 shows more detailed changes in land cover between reference years thanks to

Figure 2: Countries and territories included in the study and their coastal regions.

110 a smaller minimum mapping unit (5 ha). Supplementary Figure 1 presents the
 CLC inventory for an example coastal area in Germany. Land loss between 2012
 and 2018 due to erosion is clearly visible. The difference in resolution between
 the CLC and CLC-Changes is also noticeable. The transition from “moors and
 heathland” to “beaches, dunes, sands” is identified as CLC-Changes, but the
 115 extent of the change was too small to be mapped in CLC 2018 in contrast to
 loss of land to “sea and ocean.”

In this study, we used CLC 2018 as the baseline land cover/use for future
 projections. Historical changes were identified through CLC-Changes for 2000–
 2006, 2006–2012, and 2012–2018. This was due to the higher resolution of CLC-
 120 Changes and inclusion of only genuine changes, rather than changes involving
 improvements in classification made between different CLC datasets. All CLC
 data were used in vector format in the latest version available at the time of
 analysis (v20 dated May 1, 2019). We did not analyze changes between 1990

and 2000, as the earliest CLC inventory has lower geometric and thematic ac-
125 curacy, low time consistency, and does not cover all countries in the study area.
Consequently, we excluded CLC-Changes for 1975–1990, which was also used
by the aforementioned study. This dataset was generated in LaCoast and EU-
ROSION projects (European Environment Agency, 2002) and is not an official
130 part of the CLC inventory. It has very limited spatial coverage, low accuracy,
and very low time consistency; hence, we focused on the most accurate land
cover data for 2000–2018.

2.2. Coastal erosion projections

Probabilistic projections of sandy shoreline dynamics during the 21st century
were produced by Vousdoukas et al. (2020) combining the range of projected
135 SLR, spatial variations of active beach profile slope, ambient shoreline change
trends, and future changes in meteorological drivers such as storm surges and
waves. The long-term shoreline change D is expressed as the combined result
of two components:

$$D = R + A \quad (1)$$

where R is the shoreline response to the SLR and A is the ambient shoreline
140 change. A can be the result of different drivers, for example, geological and
hydrodynamic factors or human intervention. A is obtained by probabilistically
extrapolating historical shoreline change trends from two recent datasets based
on remote sensing data (Luijendijk et al., 2018; Mentaschi et al., 2018). These
historical trends were derived from the automated analysis of Landsat images
145 taken from 1984 to 2015 that allowed for the detection of surface-water presence
along coastal transects. Mentaschi et al. (2018) reported validation for 12 diverse
case studies, highlighting that the dataset on ambient change was able to capture
various natural and artificial effects, including coastal protection measures.

The SLR-driven retreat R was estimated using a modified Bruun rule (Bruun,
150 1962) and considering recent projections of future sea levels under RCP4.5 and
RCP8.5 emission scenarios (Jackson and Jevrejeva, 2016; Jevrejeva et al., 2014).

The Bruun rule requires the active beach slope as input, which is estimated considering the closure depth as offshore (Hallermeier, 1978; Nicholls et al., 1998). Here, the closure depth was estimated considering wave height projections from
155 a 6-member ensemble (Mentaschi et al., 2017), resulting in an equal number of estimates. Then, the slope of the active beach profile (Athanasίου et al., 2019) was estimated by combining topography from the MERIT digital elevation model (Yamazaki et al., 2017) with bathymetric data from the GEBCO dataset (Weatherall et al., 2015). Athanasίου et al. (2019) has shown using eight
160 case studies that the resolution and quality of the elevation model is adequate for the use in coastal erosion analyses. In addition, an empirical correction factor was applied. This factor was obtained from several local studies that compared the results of the Bruun rule against more elaborate methodologies (Le Cozannet et al., 2019; Ranasinghe et al., 2012a; Toimil et al., 2017). In-
165 formation about the location of sandy beaches along the global coastline was available from a recently published study (Luijendijk et al., 2018), and more detailed information about the methodology can be found in Voudoukas et al. (2020).

2.3. *Ecosystem service valuation*

170 The most commonly used resource for ecosystem service valuation is Costanza et al. (1997), which was created by integrating evidence from various case studies from different parts of the world and adjusting them to a single monetary benchmark. These global estimates have since been updated by de Groot et al. (2012) and further amended in 2014 by Costanza et al. (2014). In the process,
175 the total value of global annual services was revised upward more than threefold. This was caused by a vast improvement in available evidence from local case studies (also a threefold increase in the number of publications used) and inclusion of more types of services. The main types of services considered are (1) provisioning services (e.g., food, water, raw materials, and genetic resources),
180 (2) regulating services (e.g., regulation of water and air quality, waste treatment, erosion prevention), (3) habitat services (nursery service, genetic diversity), and

(4) cultural services (e.g., recreation and aesthetics) (de Groot et al., 2012).

In total, 16 land cover/use classes were presented in Costanza et al. (2014), and one further class is available from de Groot et al. (2012). We adapted
185 them to CLC classes in the same manner as Roebeling et al. (2013), with minor adjustments to account for changes in land cover/use classes in the new source datasets. In addition, we assigned zero value to CLC class “sea and ocean” (code 523), as we focused on the ecosystem services lost in coastal erosion without considering that some losses could be offset by services of the open
190 ocean. Nonetheless, these services are estimated to be much lower than those of provided by terrestrial or coastal ecosystem.

The original dataset provided valuation according to the purchasing power of the US dollar in the United States in 2007. Here, we converted these values into euros according to their purchasing power in the European Union (28 countries
195 before the departure of the United Kingdom) in 2018. Based on suggestions by de Groot et al. (2012), we first converted US dollars to euros using purchasing power parity for actual individual consumption in 2007 (Eurostat, 2020a) and then adjusted the result to the 2018 price level using the deflator for gross domestic product (GDP). The final valuation of services per CLC class is shown
200 in Supplementary Table 1. Apart from the total value of all services, we also calculated the value of each type of ecosystem service (17 types as presented in the supplement to Costanza et al. (2014)) for a more detailed analysis of the impact of coastal erosion. The issue of uncertainty in the valuation of services is addressed in the discussion.

205 2.4. Data processing

The base coastline was extracted from CLC 2018 (in vector format) by taking the line adjacent to CLC classes “sea and ocean,” “coastal lagoons,” “estuaries,” or “intertidal flats” and any other class than the aforementioned four. Then, a 10 km buffer was generated around the coastline, forming the “coastal zone” that is
210 of interest for this study and was used by (European Environment Agency, 2002; Roebeling et al., 2013; Copernicus Land Monitoring Service, 2019). To analyze

the distribution of services by countries and regions, the coastal zone buffer was further intersected with a vector layer of NUTS 3 regions (Eurostat, 2020b). The Nomenclature of territorial units for statistics (NUTS) classification is a hierarchical system for dividing the territory of the European Union (EU) and some associated countries into smaller units, e.g., for socio-economic analyses and shaping EU regional policies. NUTS 3 is the most detailed level available. We applied the currently used 2016 version of the classification. Eurostat’s NUTS layer reproduces the coastline in less detail than the CLC layers and inconsistently omits certain areas on the interface of land and sea. Therefore, we adjusted the layer by “extrapolating” the NUTS regions to the full extent of our coastal zone buffer using a nearest-neighbor analysis. The historical development of ecosystem services was calculated by intersecting four CLC layers (2000, 2006, 2012, 2018) and three CLC-Changes layers (2000–2006, 2006–2012, 2012–2018) with the coastal zone layer. The values from Supplementary Table 1 were then multiplied by the area of CLC patches belonging to a particular NUTS region.

Coastal erosion projections (Vousdoukas et al., 2020) were obtained from an online repository, LISCOAST (European Commission Joint Research Centre, 2019). The dataset is in the form of a point layer with varying distances between the points, which represent the beach transects used in the study. The erosion values were assigned to CLC coastline segments according to the nearest-neighbor approach but only within a radius of 10 km. Only a line adjacent to CLC classes “sea and ocean” or “intertidal flats” (only if they are adjacent to “sea and ocean”) and any other class than the aforementioned two was considered here as the coastline. The coastline segments were additionally split into much smaller fragments (50 m on average) at each vertex to allow the assignment of erosion data to the coastline would be more precise. The coastline with erosion values was the basis for generating buffers around the segments corresponding to the predicted intensity of erosion. Twelve scenarios were considered: a combination of two time periods (2050, 2100), two emission scenarios (RCP4.5 and RCP8.5), and three percentiles of the uncertainty distribution of the projections

(5th, 50th, and 95th).

However, the erosion projections of Vousdoukas et al. (2020) assume an un-
245 limited backshore space for shoreline retreat. This is not realistic as erosion
could be slowed or stopped due to barriers such as hard protection structures,
build-up areas, or geological features (Doody, 2013; Pontee, 2013). We therefore
constrained the erosion by introducing two types of barriers into the analysis.
First, artificial barriers representing various anthropogenic structures were ex-
250 tracted from the Imperviousness Density 2015 dataset with a 100 m resolution
(Copernicus Land Monitoring Service, 2020b). This resource identifies areas in
which the soil is at least partially sealed by artificial surfaces and was gener-
ated using semi-automated classification of satellite-derived normalized differ-
ence vegetation index (NDVI). Second, topographic barriers representing high
255 ground that are unlikely to be easily eroded were obtained from a pan-European
elevation model EU-DEM (Eurostat, 2020b). The model is a blend of Shuttle
Radar Topography Mission (SRTM) and Advanced Spaceborne Thermal Emis-
sion and Reflection Radiometer Global Digital Elevation Map (ASTER GDEM)
data and has a relatively low bias (-0.56 m) and root mean-squared error of 2.9 m
260 (DHI GRAS, 2014). Although the original resolution of EU-DEM is 25 m, we
have bilinearly resampled it to 100 m resolution to match the imperviousness
dataset. We considered areas with an elevation of 10 m above sea level and
more as barriers. The exact threshold is difficult to define given the diversity of
coastal morphology and geology; however, the 10 m value is commonly used to
265 delineate the “low-elevation coastal zones” for SLR studies (McGranahan et al.,
2007; Neumann et al., 2015).

Further processing of the data is presented in Supplementary Figure 2. We
used the segmented coastline to generate transects of variable density at 50 m
apart on average. Each transect was intersected with the barriers, erasing areas
270 that are invulnerable to erosion. Parts of the transects that were no longer
connected directly to the coastline were removed. The remaining parts of the
transects were used to clip the erosion buffers to generate the final erosion layers
under the 12 scenarios. Through intersection with the CLC 2018 layer, the loss

in land and ecosystem services was obtained. The geospatial data processing
275 in the study was carried out using Python scripts executed in ArcGIS software,
while statistical analyses of the results were completed in MATLAB.

To analyze impacts at national and regional scales, the value of services lost
is presented not only in absolute terms or relative to their value in 2018, but
also in relative economic terms. For this, we used GDP data in purchasing
280 power parities from Eurostat (Eurostat, 2020a) at both the national (2018) and
regional (NUTS 3) levels (2018 or latest available, adjusted to 2018 price level).
Two regions of Ireland (mid-west and south-west) were merged for this analysis
due to the lack of separate regional GDP estimates. National-level GDP for
Bosnia and Herzegovina was used as the country has not yet implemented the
285 NUTS classification. In addition, whenever results for Norway as a whole are
presented in this paper, they exclude both ecosystem services and GDP from
three regions for which coastal erosion data were not available (Nordland, Troms,
and Finnmark). Five small countries and territories (Gibraltar, Guernsey, Isle
of Man, Jersey, and Monaco) were excluded from the economic analysis at the
290 national and regional levels due to the lack of GDP data.

Additionally, we analyzed some possible influences on the results from fac-
tors other than coastal erosion, namely (1) land use change, (2) uncertainty in
ecosystem service valuation, and (3) demographic growth. The transitions be-
tween land cover/use classes are investigated for year 2000–2018 used the more
295 detailed CLC-Changes layers. Then, to analyze the uncertainty of ecosystem
valuation, we used information from de Groot et al. (2012), who provided the
range of estimates of ecosystem services from various studies (mean, minimum,
maximum, median). We adjusted them to the year 2018 value of euro and
amended with values for croplands and urban areas (zero for minimum and me-
300 dian scenario, average from Costanza et al. (2014) for the maximum scenario).
Finally, to highlight possible changes in demand on services (see the discussion
in section 4.2.3), we computed additional results using probabilistic population
projections computed by United Nations (2019). This dataset provides annual
population numbers from 1950 to 2100, with projections starting in 2020 pub-

lished as five percentiles of the uncertainty distribution of demographic change (5th, 20th, 50th, 80th, and 95th). We then recalculated the results by increasing or decreasing the value of services proportionally to the population change.

3. Results

3.1. Pan-European overview

The coastal zone in the study area, excluding intertidal flats and open sea, covered almost 580,000 km² of mostly forests and semi-natural areas (41 %) as well as agricultural areas (41 %). Between 2000 and 2018, the area of both major land cover types declined, with a corresponding expansion of artificial surfaces (Table 1). A minimal decline in wetlands and a small increase in area covered by water (other than open sea) was observed. The overall area increased narrowly between 2000 and 2018 (by 109 km²). The largest share of sea-to-land transitions were observed for the Corine Land Cover class covering sea harbors. Altogether, the annual ecosystem service of the coastal zone in 2018 was 494 billion euros (Table 2), or 3 % of the combined GDP of the countries included in the study. Between 2000 and 2018, the value of ecosystem services declined by 323 million euros, entirely after 2012 (Table 2). The majority of the decline was caused by contraction of coniferous forests and intense agriculture, which was partially compensated for by the expansion of wetlands, mainly salt marshes. Some lost ecosystem services were also replaced by different types of services provided by urban areas.

Projections of the future indicate major losses due to erosion. By 2050, approximately 3000–3500 km² of coastal ecosystems would be destroyed by erosion, depending on the emission scenario (RCP4.5 or RCP8.5). The 90 % uncertainty interval of those estimates, combined for both emission scenarios, spans from 1700 to 6900 km² (Table 1). This constitutes 0.5–0.6 % [0.3–1.2 %] of the coastal zone’s area but 2.2–2.4 % [1.4–3.8 %] of the 2018 value of ecosystem services (Table 2). This is because while erosion particularly affects forests and semi-natural areas in terms of area (51–52 % of total area lost), loss of highly valuable wet-

Table 1: Historical (2000–2018) and projected (2050 and 2100) land cover/use in the coastal zone, in thousand km². The values for 2050 and 2100 are the median projections for RCP4.5 and RCP8.5 (first row) and the 90 % confidence interval combined for both emissions scenarios (second row). Water excludes the sea and ocean.

CLC class	2000	2006	2012	2018	2050	2100
Artificial surfaces	43.2	44.9	46.0	46.7	46.5 [46.3–46.6]	46.3–46.4 [46.0–46.5]
Agricultural areas	240.4	239.3	238.6	238.1	237.3–237.4 [236.4–237.7]	236.3–236.7 [234.5–237.2]
Forest and semi natural areas	240.9	240.3	240.1	239.8	238.0–238.3 [236.3–238.9]	236.2–237.1 [233.5–238.0]
Wetlands	37.8	37.7	37.7	37.7	37.2–37.3 [36.8–37.4]	36.7–36.9 [36.1–37.1]
Water	17.2	17.3	17.3	17.3	17.1–17.2 [16.9–17.2]	16.9–17.0 [16.6–17.1]
Total	579.6	579.6	579.6	579.7	576.2–576.6 [572.7–577.9]	572.4–574.1 [566.7–575.9]

lands would be the main driver of the decline in ecosystem service (71–72 % of
335 services lost).

In the second half of the 21st century, the uncertainty increases, and the two
emission scenarios diverge (Figure 3). Under RCP4.5, erosion reaches 5500 km²
[3800–8600 km²] by 2100, while emissions under RCP8.5 could lead to destruc-
tion of 7200 km² [4300–13,000 km²] of habitats. That is, 0.7 % [1.0–1.5 %] of
340 the 2018 area under RCP4.5 and 1.2 % [0.7–2.2 %] under RCP8.5. The decline
in services would be even greater: 4.2 % [3.0–6.1 %] under RCP4.5, and 5.1 %
[3.3–8.5 %] under RCP8.5. In absolute terms, this is a possible loss of up to 42
billion euros per year. Around half of the eroded area would be in forests and
semi-natural areas (48–50 %), but the services of wetlands would make up more
345 than 70 % of losses, up to 30 billion euros, or almost a quarter of the 2018 value.

The most severely affected type of CLC land cover in absolute terms would

Table 2: Historical (2000–2018) and projected (2050 and 2100) annual ecosystem services in the coastal zone, in billions of 2018 euros. The values for 2050 and 2100 are the median projections for RCP4.5 and RCP8.5 (first row) and the 90 % confidence interval combined for both emissions scenarios (second row). Water excludes the sea and ocean.

CLC class	2000	2006	2012	2018	2050	2100
Artificial surfaces	30.7	31.6	32.2	32.4	32.2–32.3 [32.1–32.3]	32.1–32.2 [31.9–32.3]
Agricultural areas	170.9	170.1	169.6	169.3	168.7–168.8 [168.1–169.0]	168.0–168.3 [166.7–168.6]
Forest and semi natural areas	114.1	113.4	113.3	112.6	110.5–110.8 [108.8–111.5]	108.7–109.5 [106.6–110.3]
Wetlands	127.6	128.2	128.4	128.7	119.9–120.7 [111.4–123.7]	110.0–113.4 [98.5–117.4]
Water	51.1	51.2	51.1	51.1	50.7 [49.9–50.9]	49.9–50.2 [48.5–50.5]
Total	494.4	494.5	494.6	494.1	482.0–483.2 [470.2–487.4]	468.7–473.4 [452.3–479.1]

be coniferous forests or beaches, sands, and dunes, depending on the scenario. However, beaches, sands, and dunes would be by far the endangered in relative terms. 17–18 % [10–33 %] is at risk by 2050 and 28–34 % [21–48 %] by 2100.

350 This translates into second-highest loss in annual services, up to 3.6 billion euros out of 7.5 billion generated in 2018. Relative losses to salt marshes are second-highest in that category (14–17 % [10–27 %] by 2100) but also by far the biggest in absolute terms, namely 14–17 [10–28] billion euro per annum by 2100. Salines, estuaries, inland marshes, and natural grasslands would also
355 be among the most affected in both relative and absolute terms. Agricultural areas associated with an intense mode of production (vineyards, plantations, orchards, permanently irrigated crops, etc.) would be the least affected.

The impacts of coastal erosion also varied between different ecosystem services. As shown in Table 3, the main benefits of the ecosystems are regulating

Figure 3: Historical (2000–2018) and projected (2050 and 2100) coastal ecosystem services per year, in billions of 2018 euros.

and provisioning services. However, regulating services would be far more affected, losing 7–9 % [5–15 %] by 2100, compared to a loss of only about 1 % [0.7–2.2 %] for provisioning. Habitat and cultural services are similarly valued at present, but losses of the former are twice as high as those of the latter. On a more detailed level, the most valuable service of the coastal ecosystem is that of waste treatment (115 billion euro). It is provided mostly by wetlands; hence, it would be the second-most affected in relative terms (11–14 % [8–22 %] by 2100). The highest relative losses would be caused by disturbance moderation (13–16 % [10–25 %] by 2100), which is also primarily done by wetlands. In contrast, very limited loss (1 % or less) would happen to services of climate regulation, soil formation, pollination, genetic resources, or food production.

3.2. Results by country

The scope and importance of coastal ecosystem services varied substantially between countries (Figure 4). The United Kingdom had the highest total in 2018, with more than 59 billion euros of annual service, followed by France
375 (54 billion euros) and Italy (50 billion euros). However, this only constitutes 2.5–2.8 % of national GDP, similar to the pan-European average. In contrast, services of the fourth-ranked Greece, totaling 41 billion euros, accounted for more than 18 % of GDP. This was the highest relative economic value in Europe, matched only by Norway and Albania (also about 18 % of GDP). Other countries
380 for which coastal ecosystem services were highly important include Cyprus, Denmark, Estonia, and Montenegro. On the other end of the scale, services in Poland and Germany did not exceed 1 % of GDP despite long coastlines. Lower importance was only recorded in countries with very short coastlines, namely Bosnia and Herzegovina, Belgium, and Slovenia.

385 Impacts of coastal erosion were similarly unevenly distributed. Denmark was projected to lose most services relative to GDP in all scenarios, reaching 1.2–1.6 % [0.8–2.4 %] by 2100 (Figure 5). Greece was the second-most affected country by the mid-century but surpassed by Albania in the latter part of the century. Losses of more than 0.3 % of GDP per annum are also likely in Estonia
390 and Finland, and possible (within 90 % confidence interval) in Norway, Latvia, Sweden, and the United Kingdom. On the other hand. In Bosnia and Herzegovina, Belgium, Poland, and Malta, the impacts would be almost negligible (less than 0.02 % of GDP). In addition, the highest absolute losses would occur in France (14–19 % share in Europe depending on the scenario and percentile),
395 before the United Kingdom (11–17 %), Denmark (12–15 %), and Germany (9–16 %).

Further, analyzing the loss as a percentage of 2018 services yielded a different picture. While Denmark again came on top, with projected losses of 9–12 % [7–19 %] by 2100, it was followed closely by Germany (Figure 6). The 95th percentile
400 of the uncertainty distribution indicated that Germany might even lose a greater share of its coastal ecosystem services than any other European country (21 %).

Figure 4: Coastal ecosystem services per year based on CLC 2018, in billions of euro and as % of 2018 GDP, by country. Notes: BH – Bosnia and Herzegovina, UK – United Kingdom.

The Netherlands was also projected to lose a considerable share of its relatively small amount of services (8–9 % [7–16 %]) by the end of the 21st century. France, Albania, Finland, and the United Kingdom could all lose more than 10 % of their 2018 services under certain scenarios. Conversely, Bosnia and Herzegovina, Belgium, Cyprus, Lithuania, and Montenegro would most likely lose no more than 1 %. Finally, out of minor territories for which GDP was not available, small impacts were projected in Monaco, Guernsey, and the Isle of Man, but major losses in ecosystem services are possible in Gibraltar and Jersey.

3.3. Results by region

The coastal zone in the study area intersects with 491 NUTS 3 regions, of which 405 would be affected by coastal erosion in at least one scenario. The remaining 86 regions mostly did not have direct access to the coastline. The uncertainty in local impacts was large, as can be seen in Figures 7 and 8, which

415 present the minimum and maximum estimates for the year 2100. In the best case, 9 NUTS 3 regions would lose ecosystem services equal to more than 5 % of their regional GDP, while 429 regions would lose services worth less than 1 % of regional GDP. However, in the worst case, the numbers would be 40 and 369, respectively. Similarly, only 24 regions would lose more than 10 % of 2018 services in the best case scenario but 95 would in the worst-case scenario.

Worst-affected regions are concentrated particularly along the southeastern coast of the North Sea, Bay of Biscay, and eastern Mediterranean Sea (Figures 7 and 8). The ranking of regions varied substantially depending on the method of measuring ecosystem loss. In absolute terms, the Danish, German, and French regions most frequently appeared on top. The Danish region of North Jutland (Nordjylland – DK050) and German district of Nordfriesland (DEF07) could lose about 2 billion euros of services. The Albanian county of Gjirokastrë (AL033) could incur the highest relative loss in economic terms (41 % of GDP in the most extreme case), although it could almost be matched by Nordfriesland under the worst-case scenario. Many Greek regions were projected to be among the most affected in relative economic terms, similar to the insular parts of Scotland. Losses relative to 2018 services would be particularly large in several districts of Lower Saxony in Germany, especially Friesland (DE94A) in the range of 59–65 % by 2100. Many regions in various parts of the United Kingdom would also be among those with the highest relative impacts, together with some Dutch and Greek regions. In the latter country, Imathia (EL521) could suffer the highest loss among European regions of 65 % (under the worst-case scenario). Conversely, the majority of the coasts of the western Mediterranean Sea would be only lightly affected, similar to most parts of the Adriatic Sea, Tyrrhenian Sea, southern Baltic Sea, and Black Sea.

3.4. Other factors

Other factors could importantly affect the projections of losses in ecosystem services, as discussed in Section 4.1. This section highlights the possible impacts of (1) land use change, (2) uncertainty in ecosystem service valuation,

445 and (3) demographic growth. First, table 4 indicates the role of the land use
 transitions, showing that land erosion (transition to “sea and ocean” or “inter-
 tidal flats”) was smaller than land accretion. However, 58 % of accreted land
 was in the form of artificial surfaces; hence, the loss of ecosystem services from
 erosion was narrowly higher than gain from land accretion. The largest transfers
 450 involved forests, particularly the decline from deforestation. The expansion of
 artificial surfaces (particularly urban fabric, industrial units, and construction
 sites) caused greater losses than the liquidation (revitalization) of such areas
 (mainly construction and mineral extraction sites). Changes related to agricul-
 tural production, though considerable, had a limited net effect on the value of
 455 ecosystem services. Overall, the land use transitions barely changed the overall
 value of ecosystem services in Europe between 2000 and 2018, but the local
 effect could still be considerable.

Second, (Table 5) presents the influence of uncertainty in ecosystem service
 valuation. The total value of ecosystem services in 2018 would have ranged
 460 from 50 billion euros if we had taken the minimum estimates per land cover
 class, or 1247 billion in the case of the maximums. Relative loss would also
 have varied but much less: under minimum valuation, losses until 2100 would be
 2.5–3.1 % of 2018 services, while under maximum valuation, they would increase
 6.8–8.3 %. Projections based on the mean values were in the middle of this
 465 range. The biggest difference between ecosystem valuation studies was related
 to estimating the value of wetlands (especially coastal) and forests. Using the
 median of estimates, the 2018 value of ecosystem services would be 131 billion
 euros, or 26 % of the baseline estimate analyzed in the result section. However,
 the projected losses in relative terms were close to the baseline, to a greater
 470 extent than when using mean, minima, or maxima of ecosystem service value
 estimates from local case studies (Table 5).

Finally, Figure 9 displays future losses modulated by projectinos of national-
 level population change. Aggregated at the European level, the absolute loss of
 ecosystem services would be 4–5 % lower in 2050 under the lowest percentile (low
 475 demographic growth) compared with the baseline scenario, or 8–10 % higher un-

der the highest percentile (high demographic growth). By 2100, the uncertainty increases and under low demographic growth for all countries, the ecosystem loss would be 25–28 % lower than in the baseline, or 7–16 % higher if demographic growth remains in the upper range of the projections. However, under
480 the median demographic scenario, the ecosystem service loss would be only 2–3 % higher in 2050 and 3–6 % lower in 2100 than the baseline scenario that does not consider demographic change. Still, some strong differences between countries exist, as southern and central Europe is projected to encounter significant demographic decline, in contrast to northern Europe, which is very likely to
485 grow between 2018 and 2100. The relative loss at that level of analysis would have been the same, as subnational demographic projections were not available for all of Europe at the time of writing. At the European level, relative loss was almost identical to the baseline under all demographic scenarios.

4. Discussion

490 4.1. Limitations

The study has focused on the negative effects of one coastal erosion process—. Notwithstanding the uncertainties and limitations of the datasets involved (land cover, erosion projections, ecosystem service valuation, and artificial and topographic barriers to erosion), which are discussed in the next
495 subsection, ecosystem services could decrease (or increase) due to factors that were not analyzed in this study (Barbier et al., 2011; Ranasinghe, 2016). For instance, coastal erosion could be compensated in some locations with land accretion (Ratliff and Murray, 2014). Although Vousdoukas et al. (2020) indicated accretion in many locations, it is difficult to assess what ecosystem would
500 develop and in what timeframe. The destruction of habitats could also have indirect effects. Salt marshes, the second-most affected land cover type according to this study, are particularly well-known for wave attenuation and their role in reducing erosion and flooding (Kirwan et al., 2010; Temmerman et al., 2012; Arkema et al., 2013; Vuik et al., 2016). Coastlines where salt marshes or sand

dunes are destroyed by erosion might be more endangered than others. Further, we did not consider episodic or permanent flooding of land that could occur due to mean SLR, increase in extreme sea levels, or the aforementioned loss of protective habitats (McGranahan et al., 2007; Ranasinghe, 2016; Vousdoukas et al., 2016, 2017). This requires a highly precise digital elevation model (DEM) that is difficult to obtain at the European level (Paprotny and Terefenko, 2017; Vousdoukas et al., 2018a; Paprotny et al., 2019). Coastal erosion could open new areas to inundations; for example, a coastal lagoon could be exposed by the loss of a lido, changing the lagoon’s characteristics, and losing some of its ecosystem services. Other habitats could also be negatively affected by saltwater intrusions (Barlow and Reichard, 2010; Hussain et al., 2019). Finally, the effect of the destruction of coastal ecosystems is assumed to be linear (proportional to the area lost), but this might not be the case because of the complexity of natural environment (Barbier et al., 2008). Such non-linear associations between ecological features and ecosystem services have only been investigated within local scales (Aburto-Oropeza et al., 2008).

Another factor not considered here is the possibility of habitat migration. In particular, coastal wetlands are expected to migrate landwards in response to SLR (Borchert et al., 2018). However, this could happen only if there is enough accommodation space. Fragmentation and ” coastal squeezes caused by topographic or artificial barriers strongly inhibit this process (Martínez-López et al., 2019). Schuerch et al. (2018) found strong sensitivity of projections of wetland loss to assumptions about the availability of accommodation space. Europe’s coasts are heavily built-up and densely populated with many managed land-use types (especially urban and agricultural); hence, the aforementioned study found strong ecosystem loss for the continent, notwithstanding the assumption of habitat migration).

In the future, ecosystem service value could increase or decrease through changes in land cover and use caused by human activities, especially the expansion of urban, industrial, and infrastructure-related land. Existing projections are very coarse and oriented toward changes in agricultural and forested lands

based on expected changes in food and wood demand (van Vuuren et al., 2011; Meiyappan et al., 2014; Popp et al., 2017). Historical changes are shown in sections 3.1 and 3.4. They highlight that although the net effect of various land use transitions is almost zero, transfers between individual land use types are
540 considerable. This changes the composition of services as different ecosystems provide a different set of services (Table 3).

Mitigation action could prevent many of the losses indicated in the study but are not considered beyond their past influence on coastal erosion intensity. At the same time, coastal erosion mitigation is not always compatible with pro-
545 tecting ecosystem services. Conventional hard measures, which are short-lived and expensive, alter the coastal environment (Bugajny and Furmańczyk, 2017; Brière et al., 2018). This can be prevented with a variety of soft site-specific human interventions, such as beach nourishment (Hegde, 2010; Bigongiari et al., 2015). From a global perspective, even moderate greenhouse gas emission mit-
550 igation could prevent 40 % of losses in areas covered by sandy beaches (Voudoukas et al., 2020).

The study included only European or closely related countries. However, the methodology (Figure 1) can be reproduced in other areas as well. The coastal erosion projections are global datasets, while the ecosystem service valuation
555 was adapted from a dataset with a global scope. Other datasets—land cover, imperviousness, digital elevation model, and regional GDP—were specific for Europe, but their equivalents are available for the whole world, even with a reduced resolution. However, due to the magnitude of expected coastal erosion, land cover or terrain data should have as high a resolution as possible to be
560 comparable. Otherwise, the transposition of raw coastal erosion data into land cover change will not be accurate.

4.2. *Uncertainties*

4.2.1. *Land cover and use*

CLC is a high-resolution product; however, the minimum size of features
565 is often too low to capture important aspects of the coastal zones, especially

beaches, as they are often less than 100 m in width. Otherwise, CLC is too coarse to record all but the largest instances of coastal erosion. The CLC-changes have greater resolution but are still mostly inadequate to record historical changes. Further, CLC was designed to fit the full range of European regions and was not explicitly designed to account for the characteristics of any particular country or environment. Some mixed classes (e.g., heterogeneous agricultural land) could be considered ambiguous and overlap in their description and content (Environmental Protection Agency, 2018; Lavalle et al., 2016; Villa et al., 2008). Several studies refer to the need for the detection and quantification of errors in land cover and land use maps (Bach et al., 2006; Catalá Mateo et al., 2008; Ovejero-Campos et al., 2019), which even officially should only pass the threshold of 85 % thematic accuracy. However, CLC is the most comprehensive dataset of its kind available for the European continent at present. We should also note that vast improvements to land cover data availability for the coastal zone are expected in the near future. A new CLC dataset is in preparation specifically for the 10-km coastal zone in Europe (the same as considered in this study). It will have a minimum mapping unit of 0.5 ha (compared to 25 ha in the standard product), an expanded classification of land cover and use, and is expected to become available by the end of 2020 (Copernicus Land Monitoring Service, 2019).

4.2.2. Coastal erosion

The most important source of uncertainty is the projections of coastal erosion. It is challenging on a continental scale and implies some limitations due to the lack of data, predictive tools, and the availability of computational resources. Most significantly, the dataset used here (Vousdoukas et al., 2020) is limited to coasts with sandy beaches. Further, an important component of the projections is ambient shoreline trends that were obtained by extrapolating historical trends to the future, a commonly used approach (Le Cozannet et al., 2019; Vitousek et al., 2017; Wainwright et al., 2015). However, this assumes that the processes that introduce, displace, and remove sediment along the coast will not change

in the future. Furthermore, the resolution of the remote sensing data (30 m) may be too coarse for parts of the European coastline.

The erosion projections partially use the Bruun rule to estimate shoreline retreat due to SLR (Bruun, 1962), an approach that has been the subject of criticism (Cooper and Pilkey, 2004; Pilkey and Dixon, 1996; Pilkey et al., 1993), and improvements (Anderson et al., 2015; Bray and Hooke, 1997; Dai et al., 2007; Ranasinghe et al., 2012b; Romańczyk et al., 2005; Wang and Dean, 2007). The approach is adequate for first-take, continental-scale assessments (Hinkel et al., 2013; Ranasinghe and Stive, 2009) in the absence of any alternatives. The Bruun rule was used by Vousdoukas et al. (2020) in combination with the ambient change, within the constraints of its main assumptions (longshore averaged beach response to SLR only), while ambient change expresses all other aspects (sediment budget, large-scale, long-term longshore processes). This factor is a probabilistic extrapolation of historical trends (1984–2015), which captures changes from sediment budget imbalances and geological or anthropogenic factors with good accuracy (Mentaschi et al., 2018). In addition, the projections include improvements compared to previous ones in several aspects: changes in the wave climate are being considered (Mentaschi et al., 2017), along with variations in the active beach slope (Athanasίου et al., 2019), while an empirical correction factor is applied to compensate for the overprediction of shoreline retreat by the Bruun rule found in previous studies (Le Cozannet et al., 2019; Ranasinghe et al., 2012a; Toimil et al., 2017).

Despite some regional efforts (Monioudi et al., 2017), to date, there is no European dataset of sandy beach width. As a result, the baseline projections from Vousdoukas et al. (2020) assume unlimited backshore space for shoreline retreat; an assumption that may result in an overestimation of shoreline retreat as, at many sites, the latter will be constrained by human development or physical barriers. Due to limited data on such barriers, we introduced two proxies: artificial barriers based on soil sealing data and topographic barriers based on the elevation data. The exact interaction of the coastline with these barriers is uncertain, but it still provides an improvement in terms of constraining ero-

sion to areas where it is most likely to occur. Depending on the scenario, the “constrained” erosion is 38–58 % lower compared to assuming no restrictions on shoreline retreat. However, the most valuable habitats such as wetlands
630 were rarely sheltered from erosion by the barriers; hence, the loss of ecosystem services is only 11–24 % lower compared to assuming unlimited space for erosion.

4.2.3. *Ecosystem service valuation*

Further, there is uncertainty in the economic analysis. Ecosystem services have a specific value and are measurable, but there is great diversity in methods
635 used to estimate their actual value. The approach of Costanza et al. (1997), an updated version of which is used here, has been criticized for the calculation methods used, low reliability of the output data, and the incorrect aggregation of ecosystem types and service types (Bockstael et al., 2000). Different studies used various approaches deviating from the willingness to pay concept utilized
640 by Costanza et al. (1997), such as market prices, replacement costs, production costs, hedonic behavior prices, travel costs, travel cost changes, border prices, selection modeling, replacement costs, opinion polls, and health value assessment (Brenner-Guillermo, 2007; DEFRA, 2007). Consequently, researchers use at least four different systems for the classification of ecosystem services proposed by (1) Costanza et al. (1997), (2) Millennium Ecosystem Assessment
645 (2005), (3) TEEB (2010), and (4) CICES V5.1 (Haines-Young and Potschin, 2018). Despite calls for the integration of ecological, socio-cultural, and economic methods (de Groot et al., 2012; Lopes and Videira, 2013), no compromise has been reached on this issue and it is rare for synthesis to result from different
650 approaches (Pascual et al., 2012), although mixed methods are used in some cases (Szabó, 2011; Kenter et al., 2013). The result is often habitat-specific or scattered individual information rather than comparative results across ecosystem services. Although models are increasingly being developed to fill gaps, they are often based on non-organic data, with many limitations and simplifications (Hattam et al., 2015a; Atkins et al., 2015). In addition, we mostly used
655 the mean value of estimates in (de Groot et al., 2012), while Pendleton et al.

(2016) suggested using median values because of the often large variance of limited data points. We analyzed the influence of different ecosystem service value aggregation methods in section 3.4, indicating that the uncertainty of this factor
660 on relative loss (i.e., as % of 2018 services) is lower than for the uncertainty in coastal erosion projections.

In the long term, the value of ecosystem services could increase or decrease due to changes in supply and demand. With ongoing erosion and land-use change, the supply is unlikely to increase in the European coastal zone, as was
665 highlighted by our results. However, demand could change due to factors such as evolution in preferences or willingness to pay (Costanza, 2000; Voudoukas et al., 2009; Uehara et al., 2018). Although such preferences were shown to change from a multiannual perspective, they are complex and very uncertain (Skourtos et al., 2010). This would have only a limited impact on the relative
670 losses in ecosystem services, as any changes to the value of existing services would change the value of future losses considered in the same reference prices. The absolute value of losses could be strongly affected, but this effect also comes from price inflation in the general economic shifts or movements in exchange rates between countries. For this reason, the study uses a constant price level of
675 2018, as it is a standard procedure to use constant prices in economic analysis. Further, future economic growth is also highly uncertain (Dellink et al., 2017); hence, losses in relative economic terms are presented here based on most recent historical regional GDP data. This allows identifying areas that would be most severely affected, although the actual loss would likely be lower in 2100 due
680 to an increase in GDP by the end of the century. However, GDP growth will partially depend on population growth, which is a factor influencing demand for ecosystem services. The possible influence of population growth on the results is shown in Section 3.4. On the aggregated European level, this factor has minimal influence on the relative loss, although it may still be locally important.

685 *4.3. Differences in regional-scale impacts*

Impacts of coastal erosion on the value of ecosystem services are unevenly distributed. The shoreline retreat rate varies substantially between coasts, both at present (Mentaschi et al., 2018) and in future projections (Vousdoukas et al., 2020). Most affected areas, such as south-eastern coast of the North Sea; east-
690 ern coasts of the Bay of Biscay, the Bay of Cádiz and the Irish Sea; Danube, Po and Thames river estuaries; and western Estonia have some common features. They combine rather low protection capacity in terms of topography, geomorphology, and protection structures (Liquete et al., 2013b) and more extreme storm surges compared to surrounding areas (Paprotny et al., 2016). This con-
695 tributes strongly to the high impacts recorded in Denmark, Estonia, France, Germany, and the United Kingdom. Conversely, lower extreme water levels and wave heights in the Mediterranean Sea and favorable geomorphological conditions reduce ecosystem service losses in many parts of Greece, Italy, and Spain. Glacial isostatic adjustment will result in uneven SLR (Slangen et al., 2014);
700 hence, Estonia, Finland, and Sweden will experience lower impacts owing to this effect. Further, not all coasts include sandy beaches (Luijendijk et al., 2018), which reduces the possibility of significant erosion, particularly in Spain (especially on the northern coast), Ireland, the United Kingdom, Norway, Sweden, and Turkey.

705 Nonetheless, exposure (the value of existing ecosystem services) strongly determines impacts. Future losses relative to GDP are correlated with the relative value of services in 2018 (Supplementary Figure 3) more strongly ($R^2 = 0.29$ – 0.46) at regional level than the relative loss of land (compared with the size of the coastal zone in 2018), which indicates R^2 of 0.21–0.28. Therefore, accurate
710 valuation of ecosystem services is no less important than precise assessment of coastal hazards. Similarly, the contribution of services to local economies needs to be evaluated. In our study, we only analyzed impacts relative to the present-day economic situation, as future economic development at the regional level is highly uncertain. Moreover, large losses to ecosystem services could hamper eco-
715 nomic growth, especially since the ecosystem services are recreation and tourism

(Martínez et al., 2007). Both are highly important to many coastal economies, generating gross value added of almost 200 billion euros per year and employing more than 3 million people (European Commission, 2020). Otherwise, local farming and forestry could be adversely affected. However, the relative loss of
720 services is uncorrelated with the existing size of regional economies (measured by total GDP), as their economic structures are highly diverse and not all GDP is generated in the coastal zone.

Finally, we briefly mention the differences in results compared to the only study that covered several European countries Roebeling et al. (2013). It indicated significant historical erosion for 1975–2006, whereas the newer CLC
725 inventory for 2000–2018 does not show any. The methodological differences between older CLC layers could be responsible for this (see section 2.1). Overall, stronger erosion was indicated by 2050 (1.7–2.7 % of the 2006 area, depending on the SRES climate change scenario) for 10 countries included in the older
730 study compared to our analysis with (0.9–1.2 % [0.7–2.0 %] of the 2018 area). Similarly, a greater loss of ecosystem services was projected: 6.9–10.1 % compared to 5.0–6.0 % [3.7–9.5 %] in this study. However, Roebeling et al. (2013) estimated that the existing services were worth only 30 billion euros in 2006 at the 2018 price level, 10 times less than this study for a comparable set of coun-
735 tries. We therefore predict far greater potential losses than previously indicated. A comparison of national or regional impacts is not possible, as the older study does not contain any information on the spatial distribution of losses.

5. Conclusions

This paper analyzed the effects of coastal erosion on the value of ecosystem
740 services that are continuously provided by Europe’s coastal habitats. Despite the large uncertainty in the speed of erosion, there is a clear downward trend in the projections presented here. Historical data for 2000–2018 indicate a decline in coastal ecosystem service. In the future, the process will continue to accelerate, if not mitigated. The spatial variability of both existing services and

745 their future losses is very high. Some regions and countries face considerable losses compared to the size of their local economies. Despite limitations in the scope of the study and many uncertainties highlighted in the discussion, the trend is clear and losses almost certain to amount to billions of euros if not mitigated.

750 Appropriate management, closely linked to balanced development of littoral areas, is becoming a major challenge for the research community, which is involved in knowledge-based shaping of environmental policies. Economic limitations make it impossible to design and build schemes to protect vulnerable coastal areas from all anticipated events. Although the processes that generate
755 coastal erosion are beyond human control, ecosystem services can be saved in the future if adequate mitigation procedures are developed. This study quantifies possible future losses in detail and can therefore support coastal protection planning. We relied entirely on publicly available data, which can be downloaded from the sources cited in the paper. The main results of the paper are publicly
760 available on figshare (<http://dx.doi.org/10.6084/m9.figshare.12318653>).

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Table 3: Historical (2018) annual ecosystem services in the coastal zone, in billions of 2018 euros, and projected (2050 and 2100) change in percent of 2018 services. The values for 2050 and 2100 are the median projections for RCP4.5 and RCP8.5 (first row) and the 90 % confidence interval combined for both emission scenarios (in brackets).

Type of ecosystem service	Value in 2018 (bn euro)	% lost by 2050	% lost by 2100
Regulating services	217.3	3.9–4.3 [2.4–8.5]	7.4–9.1 [5.4–14.8]
Air quality regulation	0.2	0.6–0.7 [0.4–1.4]	1.1–1.4 [0.8–2.4]
Climate regulation	19.2	0.5–0.6 [0.3–1.1]	0.9–1.2 [0.6–2.2]
Disturbance moderation	5.4	7.0–7.7 [4.4–15.0]	13.0–15.8 [9.6–24.7]
Water regulation	11.5	2.7–3.0 [1.7–5.7]	4.9–6.0 [3.6–9.1]
Erosion control	41.8	1.6–1.7 [0.9–4.1]	3.3–4.1 [2.3–7.8]
Soil formation	13.8	0.3 [0.1–0.6]	0.5–0.7 [0.3–1.4]
Nutrient cycling	2.8	3.5–3.9 [2.2–7.5]	6.3–7.7 [4.6–11.9]
Waste treatment	115.4	5.9–6.5 [3.7–12.7]	11.2–13.6 [8.2–21.9]
Pollination	1.2	0.4–0.5 [0.2–1.0]	0.8–1.0 [0.5–1.8]
Biological control	6.1	1.2–1.4 [0.7–2.6]	2.1–2.7 [1.5–4.5]
Habitat services	52.6	2.0–2.2 [1.2–4.3]	3.7–4.5 [2.7–7.4]
Nursery service and genetic diversity	52.6	2.0–2.2 [1.2–4.3]	3.7–4.5 [2.7–7.4]
Provisioning services	166.8	0.5–0.6 [0.3–1.2]	1.0–1.2 [0.7–2.2]
Food production	90.0	0.5 [0.3–1.1]	0.9–1.1 [0.6–2.1]
Water supply	17.3	0.7–0.8 [0.4–1.6]	1.3–1.7 [0.9–2.9]
Raw materials (incl. ornamental resources)	10.9	0.7–0.8 [0.4–1.6]	1.3–1.7 [0.9–3.0]
Genetic and medicinal resources	48.6	0.5 [0.3–1.0]	0.9–1.1 [0.6–2.0]
Cultural services	57.3	0.9–1.1 [0.5–2.0]	1.7–2.1 [1.2–3.5]
Recreation	52.9	0.8–0.9 [0.4–1.7]	1.4–1.8 [1.0–3.0]
Spiritual, aesthetic, and other cultural	4.4	2.8–3.1 [1.7–6.0]	5.0–6.1 [3.7–9.4]
Total	494.1	2.2–2.4 [1.4–4.8]	4.2–5.1 [3.0–8.5]

Figure 5: Projected (2050 and 2100) loss of coastal ecosystem services per year, % of 2018 GDP, by country and scenario. The error bars indicate the 90 % confidence interval. Notes: BH – Bosnia and Herzegovina, UK – United Kingdom.

Figure 6: Projected (2050 and 2100) loss of coastal ecosystem services per year, % of 2018 services, by country and scenario. The error bars indicate the 90 % confidence interval. Notes: BH – Bosnia and Herzegovina, UK – United Kingdom.

Figure 7: Projected (2100) loss of coastal ecosystem services per year, % of 2018 (or latest available) GDP, by NUTS 3 region, minimum estimate (5th percentile of the coastal erosion projection under RCP4.5).

Figure 8: Projected (2100) loss of coastal ecosystem services per year, % of 2018 (or latest available) GDP, by NUTS 3 region, maximum estimate (95th percentile of the coastal erosion projection under RCP8.5).

Table 4: Historical (2000–2018) change in coastal ecosystem services by year, in millions of 2018 euros. The different processes do not add up to the total as they do not form an exhaustive list, and they overlap to some extent.

Process	Balance of ecosystem services (million euro)			
	2000–2006	2006–2012	2012–2018	2000–2018
Land accretion	+106	+137	+77	+321
Land erosion	-166	-57	-122	-344
Urbanization	-330	-237	-286	-853
Revitalization	+203	+330	+202	+735
Agriculture expansion	+10	-1	-88	-79
Agriculture contraction	-110	-236	+98	-248
Reforestation	+289	+429	+254	+971
Deforestation	-586	-645	-730	-1961
Total	+116	+67	-506	-323

Table 5: Projected (2050 and 2100) loss of coastal ecosystem services per year, as % of 2018 services, including two sources of uncertainty: (1) different ecosystem value aggregation methods from local case studies in de Groot et al. (2012); (2) from the effect of future population change at national level, using different percentiles of the United Nations (2019) probabilistic population projections

Demographic scenario	Loss of ecosystem services (% of 2018 services)			
	2050 RCP4.5	2050 RCP8.5	2100 RCP4.5	2100 RCP8.5
Baseline	2.2 [1.4–3.8]	2.4 [1.3–4.8]	4.2 [3.0–6.1]	5.1 [3.3–8.5]
Mean of estimates	3.5 [2.2–6.1]	3.9 [2.1–7.8]	6.8 [4.9–9.8]	8.3 [5.3–13.5]
Median of estimates	1.6 [1.0–3.0]	1.8 [0.9–3.8]	3.1 [2.2–4.8]	3.9 [2.4–6.6]
Minimum of estimates	1.2 [0.6–2.5]	1.3 [0.6–3.1]	2.5 [1.7–4.3]	3.1 [1.9–6.1]
Maximum of estimates	3.6 [2.2–6.1]	3.9 [2.1–7.8]	6.8 [4.9–9.7]	8.3 [5.3–13.4]
Low demographic growth (5 th percentile)	2.2 [1.4–3.8]	2.4 [1.3–4.9]	4.4 [3.1–6.6]	5.5 [3.4–9.1]
Medium demographic growth (50 th percentile)	2.2 [1.4–3.8]	2.4 [1.3–4.9]	4.4 [3.1–6.5]	5.5 [3.4–9.0]
High demographic growth (95 th percentile)	2.2 [1.4–3.8]	2.4 [1.3–4.9]	4.0 [2.9–5.9]	4.9 [3.4–8.1]

Figure 9: Projected (2050 and 2100) loss of coastal ecosystem services per year, % of 2018 GDP, by country and scenario, including the effect of future population change at national level, using the expectancy of the distribution of the United Nations (2019) probabilistic population projections. The error bars indicate the 90 % confidence interval of future coastal erosion. Notes: BH – Bosnia and Herzegovina, UK – United Kingdom.

Supplementary Information to Paprotny *et al.*, “Future losses of ecosystem services due to coastal erosion in Europe”

Supplementary Figure 1. CLC inventory over an example area (southern part of Sylt island, Germany): **(a)** original satellite image (IRS P6 LISS III/RapidEye from 2011–2012), **(b)** classified CLC 2012; **(c)** classified CLC 2018, **(d)** changes between 2012 and 2018 shown in CLC-Changes (Copernicus Land Monitoring Service, 2020a).

Supplementary Table 1. Value of coastal ecosystem services by CLC land cover/use class. Adapted from Roebeling et al. (2013), with ecosystem service values updated from Costanza et al. (2014), except class 324 updated from de Groot et al. (2012) and converted to euro at the 2018 price level and purchasing power parity for the European Union (28 countries before departure of the United Kingdom).

CLC classification				Value per ha (euro)
Level 1	Level 2	Level 3	Codes	
Artificial surfaces	Urban fabric	<i>All categories</i>	111, 112	8971
	Artificial, non-agricultural vegetated areas	<i>All categories</i>	141, 142	8971
	<i>All other categories</i>	<i>All categories</i>	121–133	0
Agricultural areas	Arable land	<i>All categories</i>	211–213	7497
	Permanent crops	<i>All categories</i>	221–223	7497
	Pastures	Pastures	231	5611
	Heterogeneous agricultural areas	<i>All categories</i>	241–243	7497
Forest and semi natural areas	Forests	<i>All categories</i>	311–313	4225
	Scrub and/or herbaceous vegetation associations	Natural grasslands; moors and heathland; sclerophyllous vegetation	321–323	5611
		Transitional woodland-shrub	324	2139
	Open spaces with little or no vegetation	Beaches, dunes, sands	331	34,586
		Bare rocks; burnt areas	332, 334	0
		Sparsely vegetated areas	333	5611
Wetlands	Inland wetlands	Inland marshes	411	34,586
		Peat bogs	412	0
	Maritime wetlands	Salt marshes; salines	421, 422	261,062
		Intertidal flats	423	0
Water bodies	Inland waters	<i>All categories</i>	511, 512	16,851
	Marine waters	Coastal lagoons; estuaries	521, 522	38,943
		Sea and ocean	523	0

Supplementary Figure 2. Calculation of constrained erosion extent. Barriers (topographical and artificial) protect the areas behind them (analyzed at the level of transects). The protected area (light green) is removed from the unconstrained erosion extent (green line) to obtain the final extent (dark green).

Supplementary Figure 3. Value of coastal ecosystem services by NUTS 3 region, 2018.