Global hotspots of salt marsh change and carbon emissions

Salt marshes provide ecosystem services such as carbon sequestration\(^1\), coastal protection\(^2\), SLR adaptation\(^3\) and recreation\(^4\). Salt marshes, mangroves and seagrass are commonly called blue carbon ecosystems—coastal wetlands that store and sequester large amounts of carbon\(^1\). Globally, salt marshes occur across all continents, except Antarctica, in low-energy tidal environments. About 40% of the mapped salt marsh extent is found in North America and about 25% in Australia\(^6\). Globally, coastal wetland loss rates increased for much of the twentieth century before declining in the 1990s\(^21\). Drivers of salt marsh loss include drainage\(^7\), eutrophication\(^8\), sediment availability\(^9\) and SLR\(^5\). SLR-driven marine transgression can also cause gains\(^11,14\), an important process to offset loss globally\(^15\). Recent studies have addressed the difficulty of global mapping and change analysis in intertidal and subtidal systems with cloud computing including mangroves and tidal flats\(^16-18\), but salt marsh monitoring activities are limited to national efforts\(^20\) or included in generalized global estimates of coastal wetland change\(^21\). Moreover, the latest blue carbon accounting of stocks and fluxes still relies on dated estimates of salt marsh change (1–2% year\(^{-1}\)) derived from limited situe analyses of several estuaries and century-long time periods\(^22\). Here we create the first consistent spatial and temporal estimates of contemporary salt marsh change from 2000 to 2019.

We analyse the global distribution of salt marsh change rates, including loss, gain and recovery. We also assess the impact of these changes on salt marsh carbon stocks worldwide from the year 2000 onward. We constrained our analysis with the most comprehensive global salt marsh map available, based on a compilation of national and regional datasets\(^1\). We processed all Landsat 5, 7 and 8 imagery with Google Earth Engine within 1.8 km of the known extent\(^1\) by implementing a normalized difference vegetation index (NDVI)-based anomaly analysis\(^17,18\), comparing a reference period (1984–1999) to change in four 5-year epochs (2000–2019). We further conducted a rigorous accuracy assessment of our analyses with 12,600 validation points split evenly by epoch and used to calculate confidence intervals and threshold sensitivity. A panel regression analysis was also conducted by watershed for conterminous United States (excluding Alaska, Hawaii and Puerto Rico), subsequently referred to as the USA, to understand change drivers, including storm events, urbanization, change surrounding the salt marsh and local sea-level change (LSLC), defined as the 5-year local trend in sea level. Our global salt marsh change data are openly available.

From 2000 to 2019, there was a global net salt marsh loss of 1,452.84 km\(^2\) (733.1–2,172.07 km\(^2\); Fig. 1). This net salt marsh loss is equivalent to a quarter of net mangrove losses (3,807.1 km\(^2\)) from 1996 to 2016 (ref.\(^{23}\)), in a global study of mangrove carbon emissions with areal change calculated from Earth observation\(^24\). Between 2005 and 2009, North America experienced the largest net loss of any region in a single epoch (282.6 km\(^2\)). Here we found that watersheds affected by higher-category hurricanes lost more salt marsh. This highlights the climate dependence of these systems and expected increases in losses from climate change owing to increases in storm intensity and frequency. High uncertainty and continued net losses of salt marsh also highlight the need for continued global and local mapping and monitoring efforts at appropriate spatial and temporal resolutions to enable management, protection and restoration of these ecosystems.

Hotspots of salt marsh change

Globally, an area of salt marsh approximately the size of two soccer fields (14,280 m\(^2\)) was lost hourly from 2000 to 2019, totalling 2,733.33 ± 355.06 km\(^2\). This loss was offset by 1,279.84 ± 255.34 km\(^2\) and mangrove encroachment\(^8\) are known drivers of salt marsh loss. However, the global magnitude and location of changes in salt marsh extent remains uncertain. Here we conduct a global and systematic change analysis of Landsat satellite imagery from the years 2000–2019 to quantify the loss, gain and recovery of salt marsh ecosystems and then estimate the impact of these changes on blue carbon stocks. We show a net salt marsh loss globally, equivalent to an area double the size of Singapore (719 km\(^2\)), with a loss rate of 0.28% year\(^{-1}\) from 2000 to 2019. Net global losses resulted in 16.3 (0.4–33.2, 90% confidence interval) Tg CO\(_2\)e year\(^{-1}\) emissions from 2000 to 2019 and a 0.045 (−0.14–0.115) Tg CO\(_2\)e year\(^{-1}\) reduction of carbon burial. Russia and the USA accounted for 64% of salt marsh losses, driven by hurricanes and coastal erosion. Our findings highlight the vulnerability of salt marsh systems to climatic changes such as SLR and intensification of storms and cyclones.
and 110.56 ± 20.05 km² of gain and recovery, respectively, for a net loss of 1,452.84 (733.1–2,172.07) km² (Fig. 2 and Supplementary Table 1). For comparison, from 1996 to 2016, there were 8,050.4 km² and 2,243.3 km² of mangrove loss and gain, respectively23. Our estimate of the global loss rate of salt marsh was 0.28% year⁻¹, a substantial reduction compared with loss rates of 1–2% year⁻¹ used in previous carbon emission estimates22. Net loss was slightly higher than mangrove net loss from 1996 to 2016 (0.2%)23.

Salt marsh losses were most prominent in Russia and the USA, which accounted for 64% of the total global salt marsh loss (Extended Data Table 1). The epoch with the greatest global loss rate was 2015–2019, when salt marsh extent decreased at a rate of 0.33% year⁻¹, mainly owing to the large losses in Russia and the USA. In fact, the magnitude of marsh losses in Russia and North America from 2015 to 2019 was similar, despite the extent of Russian salt marsh being less than half that of North America. Extreme erosion rates (up to 20 m year⁻¹)25, field survey methods, a starting survey year of 1973 and limited satellite data availability were probably causes of the high loss rates of Russia. For the USA, salt marsh changed at a rate of −0.35% year⁻¹ from 2005 to 2009, which closely agrees with the loss rate for 2004–2009 from national monitoring programmes (−0.46% year⁻¹)20. Epochs of elevated loss or gain were common globally. South America experienced elevated losses from 2000 to 2004 (Extended Data Table 1 and Supplementary Table 1). Oceania and Africa/Middle East were the only two regions in which marsh gains exceeded losses (Extended Data Table 1).

**Marsh recovery**

Recovery from disturbances and landward migration are two critical components that influence the persistence of salt marshes but are poorly understood at both regional and global scales. Globally, 4.7% of all salt marsh losses had recovered by 2019, with most of the recovery occurring in areas lost between 2005 and 2009. These 2005–2009 losses coincide with extreme weather events such as hurricanes Rita, Wilma and Katrina in 2005, which greatly affected the Gulf Coast of the USA and resulted in a conversion of 562 km² of land to water in Louisiana26. The 16.5% recovery rate for losses occurring from 2005 to 2009 in the Gulf of Mexico region provides further evidence that storm events had a higher recovery rate than other loss drivers. Recovery increased in each subsequent epoch, except for the losses
of Asia from 2000 to 2004 (Fig. 2). North American salt marsh comprised approximately 44% of the total salt marsh extent but 71% of all recovery. Still, salt marshes in the region were far from returning to the pre-epoch extents.

The location of loss and recovery times can provide essential insight into the process, type and amount of greenhouse gas emissions from blue carbon systems. In addition to natural regrowth, the recovery maps probably also captured restoration sites. In North America, storm events in 2005–2009 resulted in high losses (Extended Data Table 1) and subsequent high recovery rate owing to a combination of restoration and natural revegetation. In 2007, 2 years after Hurricane Katrina made landfall, the Louisiana legislature responded by commissioning the first Coastal Master Plan, which resulted in coastal restoration projects such as marsh creation, sediment pipelines, shoreline restoration and oyster reef restoration. The area of direct restoration is unclear given the potential indirect benefit of oyster reef restoration and sediment pipelines. Our results were able to quantify the long-term legacy of these recovery processes, which are critical for understanding salt marsh resilience.

Salt marsh change drivers

We analysed salt marsh change within the USA in relation to LSLC, urbanization, change surrounding the salt marsh and hurricane landfall and intensity. We found that urbanization was not a detectable twenty-first century loss driver, suggesting that protections for salt marshes effectively limited conversion from drainage, and indirect effects related to urbanization such as increased nutrients and changes to the sediment supply were not considerable drivers of change. Similarly, findings in mangrove ecosystems showed that settlement accounted for only minimal recent losses. The largest increase in loss was related to hurricane landfall and intensity, which increased salt marsh losses but had no notable effect on salt marsh gains.

In both panel regression models, loss and gain in the 100 m surrounding salt marshes were important predictors of salt marsh change (Supplementary Table 2). Losses surrounding the marsh were probably because of edge erosion and wetter tidal flats. Gain anomalies, increases in NDVI, in the 100 m surrounding the salt marsh, resulted in more changes within the salt marsh in terms of both losses as well as gains. For salt marsh loss, the observed relationship with vegetative greening (gain in NDVI) adjacent to a salt marsh could correspond to accretionary coasts (Fig. 3a–d). Higher LSLC was substantially related to reduced salt marsh gains (Supplementary Table 2).

The significance of change near the salt marsh in the watershed-scale panel analysis demonstrates the importance of gradual local change that our anomaly analysis observes. For example, SLR probably caused extensive losses within this section of Maryland’s Eastern Shore (Fig. 3) supported by the nearby Ocean City tidal gauge with a long-term SLR trend of 6.05 ± 0.73 mm year⁻¹ (ref. 29). Gradual loss is also evident, such as erosion along the barrier island (Fig. 3d). In the panel analysis, loss was not substantially affected by LSLC. As inundation increases in the region, loss anomalies surrounding the marsh increase, therefore these losses surrounding the marsh better reflect the impact of SLR than the short-term trends of LSLC.

Marsh erosion is linearly related to water body size and, although not directly included in the analysis, we expect erosion rates to relate to losses of vegetation surrounding the marsh. Drainage and direct anthropogenic conversion were relatively limited in the USA. By contrast, on a global scale, salt marsh trends were complicated by anthropogenic change, which we believe is underrepresented in this analysis owing to the limitations of the baseline salt marsh extent dataset. For example, a recent study of salt marsh change in China demonstrated a net loss of 359.27 km² from 1985 to 2019 but only 22.02 km² of loss from 2000 to 2019 (ref. 31). Despite our analysis including only approximately half the salt marsh extent in China for 2000 (514 km² compared with 1,176 km²), we found a small net loss rate of 0.006 (−0.45 to 0.47)% year⁻¹ and Chen et al. found a loss rate of 0.0009% year⁻¹ (ref. 31). Similarly, a regional analysis of European salt marsh change found a net increase of 127.5 km² of salt marsh from 1986 to 2010 (ref. 31); whereas our analysis found a 135.9 (38.7–235.8) km² loss of salt marsh extent from 2000 to 2019 (ref. 31). Despite these differences in results between our maps and more localized studies, our results allow, for the first time, to evaluate global patterns of salt marsh change with a consistent dataset, reproducible methodology and rigorous uncertainty analysis.

Uncertainty and future analysis

Our work focused on improving salt marsh change estimates only. However, overlap in blue carbon ecosystem extent can introduce some uncertainty. Globally, there is overlap between the global extent of mangroves and the mapped extent of salt marshes. Most of this overlap, 80%, occurs within Australia (1,590 km²). To complicate matters, the overlap between the two ecosystems is a source of double-counting in existing blue carbon budgets, and overlap should be accounted for in uncertainty estimates. Mangrove encroachment is probably an

![Figure 2](https://example.com/fig2.png) **Regional salt marsh recovery.** Salt marsh recovery for each study region by epoch in which the recovery occurred. Colour denotes the year in which a loss occurred. Error bars represent the standard error of the recovery area.
important regional change driver in Oceania, in which the salt marsh–mangrove ecotone is altered as mangroves migrate poleward with increasingly warm conditions. In this study, mangrove encroachment or misclassification corresponded with more than a third of gains in Australia. Mangrove encroachment increases carbon sequestration and the value of ecosystem services, but the full environmental and ecological impact is unclear. In the case of SLR adaptation, for example, low marshes were aggrading with 10 mm year$^{-1}$ of regional SLR, greater than a proposed 7 mm year$^{-1}$ threshold for mangroves. In the case of restoration, salt marsh provides ecological structure quicker than mangroves. Mapping the salt marsh–mangrove ecotone is challenging at the 30-m Landsat spatial resolution, and a combination of higher resolution and new methods are necessary to improve carbon and ecosystem monitoring in this ecotone. Our estimates of mangrove encroachment and misclassification were key for constraining estimates of salt marsh change in Oceania and illustrated the need to consider coastal systems as a whole to understand both blue carbon and changes to coastal resilience.

This study is a comprehensive salt marsh change analysis. Our change rates and associated uncertainties can improve carbon monitoring estimates in salt marsh ecosystems. Our revised salt marsh maps clarify that these systems experienced a decline in net loss rates from 2000 to 2019. WorldView-2 image collected on 11 September 2019. An area of the salt marsh before losses. Orthoimage collected on 12 April 1989. Northeast area of complete loss and, to the southeast, an area of interior die-off, both identified as losses by the algorithm. WorldView-2 image collected on 11 September 2019. All images NIR, G, B in RGB. Created using ArcMap 10.8. © 2005, 2010, 2015, 2019 Maxar, NextView License.

**Fig. 3** Detailed salt marsh change for a section of the eastern shore of Maryland, USA. **a**, Salt marsh change from 2000 to 2004. QuickBird image collected on 17 May 2005. **b**, Salt marsh change from 2000 to 2009. WorldView-2 image collected on 7 December 2010. **c**, Salt marsh change from 2000 to 2014. WorldView-2 image collected on 18 April 2015. Areas of increase were visible in this epoch along the north of the barrier island. **d**, Salt marsh change from 2000 to 2019. WorldView-2 image collected on 11 September 2019. **e**, An area of the salt marsh before losses. Orthoimage collected on 12 April 1989. **f**, Northeast area of complete loss and, to the southeast, an area of interior die-off, both identified as losses by the algorithm. WorldView-2 image collected on 11 September 2019. All images NIR, G, B in RGB. Created using ArcMap 10.8. © 2005, 2010, 2015, 2019 Maxar, NextView License.
Salt marsh carbon emissions

Globally, soil organic carbon stock (SOCS) losses were relatively consistent between epochs, despite the regional and temporal fluctuations of marsh loss. Conversely, gain was estimated with carbon burial rates, which increased linearly owing to the process being cumulative (Fig. 4). Total salt marsh losses from 2000 to 2019 represented a reduction of 4.88 (3.16–6.81) Tg C of aboveground biomass. Net loss represented a reduction of 0.045 (−0.14–0.115) Tg CO₂e year⁻¹ of carbon burial. The lower bound of net carbon burial represents the potential that current change already represents a net sink when considering only anthropogenic-driven loss, (4) salt marsh regions affected by storm events and high rates of SLR, and (5) repeated globally for the new tidal marsh extent map. This study provides sophisticated methods for monitoring these ecosystems and improves our understanding of carbon stocks and change. Our work is the most comprehensive effort to monitor salt marsh carbon globally, resulting in salt marsh change estimates, improved salt marsh carbon budget and an estimate of salt marsh recovery. We identified salt marsh change spatially across two decades, provided country-level estimates of carbon stocks informed by monitoring these ecosystems and improves our understanding of salt marsh carbon dynamics.
by local estimates of SOCS and identified hurricanes as a key driver of loss in the USA. Our results justify further monitoring to facilitate inclusion of the ecosystem in preservation-based carbon protocols and Nationally Determined Contributions to incentivize restoration and protection.

Online content

Any methods, additional references, Nature Research reporting summaries, source data, extended data, supplementary information, acknowledgements, peer review information; details of author contributions and competing interests; and statements of data and code availability are available at https://doi.org/10.1038/s41586-022-05355-z.
Methods

Change anomaly analysis
We conducted a global change analysis using the best available salt marsh extent maps. When previous work compared salt marsh trends using a starting extent derived from Mcowen and ancillary data with higher-resolution regional mapping, a high correlation was found. Although acknowledged as incomplete, those maps were the most comprehensive global salt marsh map available. We used the National Wetlands Inventory (NWI) extent in the USA owing to regional updates since the aggregation of the global extents. The salt marsh extent for each region was converted to a 30-m raster and uploaded to Google Earth Engine. We filtered the Landsat 5, 7 and 8 collections by 50% cloud cover, geometric accuracy, image quality and pixel quality to remove low-quality scenes and pixels. Our time series approach of comparing all quality images within an epoch mitigates the effect of any single image on the change outcome. All these process steps should mitigate the effect of the Landsat 7 shutter synchronization anomalies. We used Google Earth Engine to compare the salt marsh NDVI baseline from 1984 to 1999 and four 5-year periods from 2000 to 2019. These 5-year epochs of 2000–2004, 2005–2009, 2010–2014 and 2015–2019 allowed for a per-pixel analysis of change to minimize the effect of tidal stage. The baseline for Siberian watersheds was adjusted to 1984–2004 owing to the limited Landsat images in the region before 2000, resulting in three 5-year analysis periods (2005–2019).

NDVI has frequently been used to analyse salt marsh change and estimate salt marsh biomass. The link between remote sensing and ecological change is nonlinear and complicated but well established from the Arctic to wetlands and marshes. Tidal marsh monitoring has also been conducted with other indices and several indices simultaneously. Machine learning kernel approaches (kNDVI) has been proposed as a method to address saturation of NDVI. However, many vegetation indices still use the relationship between red and near infrared. An intercomparison of the linear relationship of these indices, excluding kNDVI, and salt marsh aboveground biomass found that NDVI performed best. An important concern in regional and global salt marsh change analysis is variability in leaf structure in these environments. Regardless of the spectral index used, this would be an issue and something that future analyses can address with more in situ data and remote-sensing approaches.

To overcome some of these challenges, we applied a time-series NDVI anomaly approach to minimize the effect of seasonal and tidal variability and conducted a robust accuracy assessment protocol. We define anomalies as pixels that experienced greater than an absolute 0.2 magnitude of NDVI change, a threshold used in mangrove change analyses. These methods and threshold have recently been used to understand storm impacts and loss in mangrove environments. Our accuracy and threshold assessments further confirmed that this threshold had high user accuracy and successfully identified change with minimal inclusion of stable areas (Supplementary Tables 3–9). Salt marsh NDVI was compared during peak biomass months of August and September for the Northern Hemisphere and February and March for the Southern Hemisphere. The analysis unit was the watershed, HUC 6 watersheds within the USA and the World Wildlife Foundation (WWF) Basin 6 watersheds globally.

We calculated salt marsh anomaly metrics for four 5-year epochs from 2000 to 2019 using a combination of Python 3.8.10 and R 3.6.2. Salt marsh change metrics included recovery, salt marsh loss and gain anomalies, and loss, as well as gain anomalies within 100 m of the salt marsh. For a pixel to be considered recovered, it had to return to at least its reference period NDVI in a subsequent epoch. Salt marshes are unlikely to recover from certain loss processes such as herbivory, mangrove encroachment, eutrophication and SLR. However, recovery following losses from overwash, ice scour and saltwater intrusion into freshwater environments are all documented.

Accuracy assessment
We conducted an accuracy assessment across 12,600 stratified random points, split into loss, gain and stable for each period (Supplementary Tables 3–6) and an accuracy assessment of salt marsh extent, including 6,845 stratified random points split between salt marsh and other land cover categories (Supplementary Table 7). We assessed recovery across 2,000 points split between loss and recovery (Supplementary Table 8). Each point was representative of the 30-m Landsat pixel in which it fell. We determined the status of a pixel with Google Earth Pro using historic imagery to determine change and ancillary imagery in the USA. Google Earth Pro imagery sources included Maxar, Airbus, the United States Geological Survey and NASA. The spatial resolution of these images varies but very-high-resolution (<3 m) imagery was used for most of our land cover verification. In limited instances, for example, Alaska, 30-m imagery was also used. Accuracy assessments included Alaska, Australia, Canada, China, Mexico, the UK and the USA to derive uncertainty estimates for each region. All locations had >1,800 points split between epochs, except Alaska with 1,344 and the USA with 2,236. We calculated confidence intervals using the accuracy assessment results. The overall accuracy was 93%, 91%, 91% and 90% in 2000–2004, 2005–2009, 2010–2014 and 2015–2019, respectively (Supplementary Tables 3–6).

The anomaly analysis relies on a threshold to determine changed and unchanged pixels. All quality pixels are compared with the reference average, and the average across the epoch must exceed the 0.2 threshold to be considered changed, a threshold used in mangrove change analyses. We further assessed the 0.2 NDVI threshold using an expanded accuracy assessment. We randomly selected salt marsh pixels with a >0.15 magnitude change and determined whether we observed a change in the pixels using Google Earth Pro and available ancillary imagery. We found that, for loss thresholds, 0.2 and 0.19 had similar accuracy, with 0.2 having slightly higher accuracy. The 0.2 threshold was also slightly higher accuracy for gain areas and, therefore, we used a 0.2 threshold to best capture a pixel-wide change in vegetative extent (Supplementary Table 9). We further assessed our marsh loss within the USA relative to the Global Land Cover and Land Use 2019 dataset (GLAD 2019). Here we compared our loss pixels to the Land Cover Land Use (LCLU) class of that location in the GLAD 2019 data. We found that approximately 12% of our loss locations were sparse or non-emergent wetlands, and 73% were open water, showing a strong agreement between our change and the GLAD data.

Mapping year
The mapping year, the year in which imagery was acquired, varied globally, and to verify the bias introduced by mapping year, we used beta regressions to assess the mapping year in the USA. The beta regressions compared percent change metrics (loss and gain) and mapping year (Supplementary Tables 10 and 11). North America has the largest concentration of mapped salt marsh of any continent. In the USA, the NWI is an irregularly revised mapping effort to track all USA wetlands. The NWI uses the Cowardin classification, consistent methods and varying aerial data sources. These statistical analyses were limited to the USA owing to the availability of ancillary data and the metadata of the NWI. However, the analysis demonstrates that mapping date had a limited effect on the gains in 2000–2004 and the losses in 2015–2019. These change rates were used for our quantification of yearly emissions from salt marsh change.

The average mapping year varied regionally. For example, the USA had an average mapping year of 2007, compared with Alaska with an average mapping year of 1986. Mapping year also varied globally, with areas in Spain and Russia mapped in the 1980s. The mapping date most affected the 2000–2004 and 2015–2019 periods for loss and gain anomalies, respectively (Supplementary Tables 10 and 11).
Global salt marsh extent maps are lacking for many regions. However, a complete map of salt marsh would not change the result of this study markedly. For example, the salt marsh extent of India has recently been mapped, finding 290.49 km², approximately half a percent of global salt marsh. Increased map accuracy and a uniform baseline mapping data are probably more beneficial to this analysis.

Drivers of salt marsh change
Panel regression models were used to compare watersheds over time with LSLC, change within 100 m of the salt marsh, urbanization within the watershed and hurricane landfall and category or highest category in instances of several landfalls. The LSLC measure identifies periods of sea-level change that can be driven by factors such as ocean currents, climate cycles and storm events. The LSLC was derived from National Oceanic Atmospheric Administration (NOAA) tide stations for each of the four 5-year periods. The ‘nroaa’ package was used to download tide station data directly into R (ref. 71). All stations were filtered to watersheds with salt marsh in the USA and by our study period, 2000–2019. All available data for each Center for Operational Oceanographic Products and Services (CO-OPS) station were split into the four epochs of our study, 2000–2004, 2005–2009, 2010–2014 and 2015–2019. Each period had a trend calculated by decomposing monthly mean sea level using the decompose function in base R, which uses a moving average to isolate trend, seasonality and error. The more complex seasonal-trend decomposition with Loess was not used owing to the recommendation that the season window is composed of at least seven time steps 28 and our use of monthly tidal data. Linear regression was then fit to the resulting trend estimating change per year in millimetres. Watersheds with several tide stations were averaged, resulting in a single local sea-level trend for each watershed. The study included 72 watersheds within the USA, of which 45 had tide station records and which we used in the panel analysis.

Hurricane track and intensity data (HURDAT2) were acquired from National Weather Service and processed using the R package ‘tidyverse’ 29,30. We processed HURDAT2 data for both the Atlantic and Pacific oceans, but no Pacific typhoons affected watersheds within the USA. We imported the processed HURDAT2 data as a delimited text layer into QGIS 3.12.263, creating a buffer surrounding each point based on the hurricane diameter 31. If the hurricane diameter was missing, we used the average hurricane diameter for the corresponding category of storm. We used these data to determine which watersheds were affected in each epoch and the highest category of hurricane impact.

We determined watershed urbanization using a global map of impervious surface increases from 1984 to 2018 derived from the Landsat archive 32. We calculated the amount of urbanization for each watershed using the zonal histogram tool in QGIS 3.12.2. Annual impervious surface estimates were aggregated to the study periods quantifying total artificial impervious surface added in each period.

Carbon monitoring
We estimated the impact of salt marsh change on ecosystem carbon, including aboveground carbon, carbon burial and soil organic carbon. Salt marsh loss was considered the complete loss of aboveground biomass, carbon burial and SOCS. These losses were calculated for both 30 cm and 100 cm to cover a range of loss estimates. Tidal wetland carbon monitoring has previously been conducted with regional values and ecosystem extents 4. Aboveground biomass was estimated from plots across the USA by Byrd et al. (705.9 ± 720 g m⁻² (standard deviation)) with a carbon conversion of 0.441 (refs. 33,34). Belowground biomass was assumed to be included within soil core measurements of SOCS and was not computed separately. We estimated carbon burial using the latest values from the literature of 168.7 ± 7 g C m⁻² year⁻¹ (ref. 35), which is lower than older estimates of carbon burial (218 ± 24 g C m⁻² year⁻¹) (ref. 36). Carbon burial rates were used to calculate carbon increases from gains in salt marsh extent. SOCS was extracted from the SoilGrids250m dataset 41. These estimates were averaged by change type (loss or gain), epoch and watershed. Consistent with previous work, we assume a complete loss of SOCS, which has been estimated to take place over years but still underestimates total carbon lost 38. These spatial estimates of SOCS loss were compared with globally derived estimates of SOCS from the Coastal Carbon Research Coordination Network (CCRCN) and the literature 38,41. Our in situ estimate derived from the CCRCN was 270.4 ± 2.8 Mg ha⁻¹ (ref. 38), which was slightly lower than values from the literature (317.2 ± 19.1 Mg ha⁻¹) 41. The Coastal Carbon Data Clearinghouse values were exclusively within North America and Europe (Supplementary Table 13). The use of 1 m to calculate SOCS ignores another source of notable uncertainty, which is soil depth, for example, the mean depth of cores representing deposit depth in emergent vegetation was 194.5 cm (Supplementary Table 13). In comparison, the SoilGrids dataset accounts for spatial variation but underestimates the carbon lost.

The upper bound of tidal marsh extent was estimated using the recent mapped extent of 90,800 km² (ref. 39). Salt marsh carbon estimates and our change rates were applied to this extent of tidal marsh. Previous blue carbon budgets have used an upper bound of tidal marsh, which included mangroves and marshes 41,42,43,44.

Previous budgets estimated partial losses of SOCS 1. We offer a total loss that assumes an eventual complete loss of carbon. Carbon gain is calculated by combining carbon burial and aboveground carbon. The CO₂ emission estimates of soil carbon change used yearly change rate for loss and gain from the epochs 2015–2019 and 2000–2004, respectively. These epochs were the last affected by mapping year. We propagated error throughout the analysis with 100,000 Monte Carlo simulations for all confidence intervals. Uncertainty reported in parentheses is 90% confidence intervals or standard error after a ±.

Data availability
The data that support the findings of this study are openly available at https://doi.org/10.3334/ORNLDAAC/2122. Loss and gain maps are available at https://mangrovescience.earthengine.app/view/saltmarshchange and https://mangrovescience.earthengine.app/view/saltmarshsoc.

Code availability
Examples of the code used to process the data are available at https://github.com/canbpan/Global_saltmarsh.
60. O’Donnell, J. P. & Schalles, J. F. Examination of abiotic drivers and their influence on 
Sparrina alterniflora biomass over a twenty-eight year period using Landsat 5 TM satellite 
imagery of the Central Georgia Coast. Remote Sens. 8, 477 (2016).
61. Taillie, P. J. et al. Widespread mangrove damage resulting from the 2017 Atlantic mega 
hurricane season. Environ. Res. Lett. 15, 064010 (2020).
62. Lagomarsino, D. et al. Measuring mangrove carbon loss and gain in deltas. Environ. Res. 
Lett. 14, 025002 (2019).
63. Zhang, C., Durban, S. D. & Lagomarsino, D. Modeling risk of mangroves to tropical 
cyclones: a case study of Hurricane Irma. Estuar. Coast. Shelf Sci. 224, 108–116 (2019).
64. Mondal, P., Dutta, T., Qadir, A. & Sharma, S. Radial and optical remote sensing for near 
real-time assessments of cyclone impacts on coastal ecosystems. Remote. Sens. Ecol. 
Conserv. 8, 506–520 (2022).
65. Croddy, S. M. et al. Sea-level rise and the emergence of a keystone grazer alter the 
geomorphic evolution and ecology of southeast US salt marshes. Proc. Natl Acad. Sci. 
117, 17891–17902 (2020).
66. Courtemanche, R. P. Jr, Hester, M. W. & Mendelssohn, I. A. Recovery of a Louisiana barrier 
island marsh plant community following extensive hurricane-induced overwash. J. Coast. 
Res. 15, 872–883 (1999).
67. Ewanchuk, P. J. & Bertness, M. D. Recovery of a northern New England salt marsh plant 
community from winter icing. Oecologia 136, 616–626 (2003).
68. Flynn, K. M., McKee, K. L. & Mendelssohn, I. A. Recovery of freshwater marsh vegetation 
after a saltwater intrusion event. Oecologia 103, 63–72 (1995).
69. Olofsson, P., Foody, G. M., Stehman, S. V. & Woodcock, C. E. Making better use of 
accuracy data in land change studies: estimating accuracy and area and quantifying 
uncertainty using stratified estimation. Remote Sens. Environ. 129, 122–131 (2013).
70. Hansen, M. C. et al. Global land use extent and dispersion within natural land cover using 
Landsat data. Environ. Res. Lett. 17, 034050 (2022).
71. Cowardin, L. M., Carter, V., Golet, F. C. & LaRoe, E. T. Classification of Wetlands and 
Estuarine Habitats of the United States (U.S. Department of the Interior, 1979).
72. Viswanathan, C. et al. Salt marsh vegetation in India: species composition, distribution, 
zonation pattern and conservation implications. Estuar. Coast. Shelf Sci. 242, 106792 (2020).
73. Edmund, H., Chamberlain, S., & Ram, K. Package ‘moor’. (2014).
74. Cleveland, R. B., Cleveland, W. S., McRae, J. E. & Terpenning, I. STL: a seasonal-trend 
decomposition procedure based on loess. J. Off. Stat. 6, 3–73 (1990).
75. Landsea, C., Franklin, J. & Beven, J. The Revised Atlantic Hurricane Database (HURDAT2). 
https://www.nhc.noaa.gov/data/hurdat/hurdat2-1851-2018-052520.txt (NOAA/NHC, 2015).
76. Wickham, H. et al. Welcome to the Tidyverse. J. Open Source Softw. 4, 1686 (2019).
77. QGIS Development Team. QGIS Geographic Information System 3.12.2 (Open Source 
Geospatial Foundation Project, 2020).
78. Gong, P. Annual maps of global artificial impervious area (GAIA) between 1985 and 2018. 
Remote Sens. Environ. 236, 111510 (2020).
79. Thomas, N. et al. High-resolution mapping of biomass and distribution of marsh and 
forested wetlands in southeastern coastal Louisiana. Int. J. Appl. Earth Obs. Geoinf. 80, 
257–267 (2019).
80. Byrd, K. B. et al. Corrigendum to “A remote sensing-based model of tidal marsh 
average ground carbon stocks for the conterminous United States” [ISPRS J. Photogram. 
Rem. Sens. 139 (2018) 255–271]. ISPRS J. Photogramm. Remote Sens. 166, 63–67 (2020).
81. Wang, F. et al. Global blue carbon accumulation in tidal wetlands increases with climate 
change. Natl Sci. Rev. 8, nwsa296 (2023).
82. Hengl, T. et al. SoilGrids250m: global gridded soil information based on machine 
learning. PLoS ONE 12, e0169748 (2017).
83. Coastal Carbon Research Coordination Network (CCRCN). Coastal Carbon Atlas. https:// 
ccrcn.shinyapps.io/CoastalCarbonAtlas (2019).
84. Alongi, D. Carbon balance in salt marsh and mangrove ecosystems: a global synthesis. 
J. Mar. Sci. Eng. 8, 767 (2020).
85. Duarte, C. M., Middelburg, J. J. & Caraco, N. Major role of marine vegetation on the 
oceanic carbon cycle. Biogeosciences 2, 1–8 (2005).
86. Woodwell, G. M., Rich, P. H. & Hall, C. in Brookhaven Symposium in Biology, Vol. 24. 
221–240 (Brookhaven National Laboratory, 1973).

Acknowledgements This research was supported in part by the NASA Carbon Monitoring 
System programme (grant number 16-CMS16-0073). A.D.C. was supported by the NASA 
Postdoctoral Program Fellowship administered by Oak Ridge Associated Universities. We 
would also like to thank A. Stovall and C. Doughty for reading and offering edits on an early 
version of the paper. Maxar data were provided under the National Geospatial-Intelligence 
Agency’s NextView license agreement.

Author contributions A.D.C. and L.F. conceived and designed the experiments. D.L., L.F. and 
L.G. revised the manuscript. A.D.C. and L.F. wrote the first draft of the manuscript, with 
input from L.F. A.D.C., L.F. and D.L. analysed the data. A.D.C. and L.F. conceived and designed the experiments. D.L., L.F. and 

Competing interests The authors declare no competing interests.

Additional information Supplementary information The online version contains supplementary material available at 
https://doi.org/10.1038/s41598-022-05355-z.

Correspondence and requests for materials should be addressed to Anthony D. Campbell. 
Peer review information Nature thanks Patrick Megonigal, Stuart Phinn and the other, 
amonymous, reviewer(s) for their contribution to the peer review of this work. Peer reviewer 
reports are available.

Reprints and permissions information is available at http://www.nature.com/reprints.
Extended Data Table 1 | Salt marsh gains and losses in km² for each 5-year epoch by region

| Region                  | Loss          | Gain          |
|-------------------------|---------------|---------------|
|                         | 2000-2004     | 2005-2009     | 2010-2014     | 2015-2019     |
| Africa & Middle East    | 3.4±0.9       | 14.5±3.0      | 15.1±2.8      | 10.3±1.8      |
| Asia                    | 66.9±17.9     | 115.9±23.9    | 48.8±9.0      | 45.6±8.0      |
| Central & North America | 85.1±22.7     | 88.6±18.3     | 113.2±21.0    | 222.7±39.1    |
| Europe                  | 17.7±4.7      | 71.5±14.8     | 53.7±9.9      | 64.2±11.3     |
| Oceania                 | 0±0           | 26.1±5.4      | 79.7±14.8     | 68.5±12.0     |
| Russia                  | 0±0           | 11.4±2.3      | 12.0±2.2      | 5.6±1.0       |
| South America           | 13.2±3.5      | 9.8±2.0       | 8.2±1.5       | 8.1±1.4       |