Restoring tides to reduce methane emissions in impounded wetlands: A new and potent Blue Carbon climate change intervention

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Coastal wetlands are sites of rapid carbon (C) sequestration and contain large soil C stocks. Thus, there is increasing interest in those ecosystems as sites for anthropogenic greenhouse gas emission offset projects (sometimes referred to as “Blue Carbon”), through preservation of existing C stocks or creation of new wetlands to increase future sequestration. Here we show that in the globally-widespread occurrence of diked, impounded, drained and tidally-restricted salt marshes, substantial methane (CH\textsubscript{4}) and CO\textsubscript{2} emission reductions can be achieved through restoration of disconnected saline tidal flows. Modeled climatic forcing indicates that tidal restoration to reduce emissions has a much greater impact per unit area than wetland creation or conservation to enhance sequestration. Given that GHG emissions in tidally-restricted, degraded wetlands are caused by human activity, they are anthropogenic emissions, and reducing them will have an effect on climate that is equivalent to reduced emission of an equal quantity of fossil fuel GHG. Thus, as a landuse-based climate change intervention, reducing CH\textsubscript{4} emissions is an entirely distinct concept from biological C sequestration projects to enhance C storage in forest or wetland biomass or soil, and will not suffer from the non-permanence risk that stored C will be returned to the atmosphere.

Methane emissions from wetlands, predominantly from freshwater systems such as peatlands and tidal fresh and low salinity wetlands, comprise about 1/3 of global CH\textsubscript{4} emissions from all sources\textsuperscript{1}. Soil microbial respiration and low rates of methane oxidation in anaerobic, water-saturated soils result in substantial CH\textsubscript{4} emissions in those freshwater settings. Depending on the timescale of analysis, climate warming due to CH\textsubscript{4} emissions can partially or entirely offset climatic cooling due to C sequestration in freshwater wetland soil\textsuperscript{2}. In contrast, in saline wetlands, including salt marshes, saline mangroves, and seagrass beds, CH\textsubscript{4} emissions are typically minor because abundant sulfate ion in seawater limits microbial CH\textsubscript{4} production and emission\textsuperscript{3}. Thus, with high rates of net C storage and minor CH\textsubscript{4} emission, saline wetlands generally have a strong cooling effect on climate\textsuperscript{3–6}.

Commonly, however, tidal exchange of saline water between the coastal ocean and emergent, tidal wetlands, including salt marshes and mangroves (herein referred to as “tidal wetlands”), has been blocked or restricted by human activity, and those alterations can dramatically freshen and degrade the ecosystem\textsuperscript{7–9}. Both drainage and impoundment of tidal wetlands have been practiced for a wide range of purposes during the past several centuries of human civilization\textsuperscript{7–9}. Blockage or restriction of tidal flows, through installation of dikes or tide gates, is a common method to protect coastal infrastructure; to drain tidal wetlands for farming, mosquito control, and development; or to raise or manage water tables and reduce salinity for aquaculture, mosquito control, rice production, and wildfowl management. Inadvertent tidal restrictions also occur due to road, railroad and other infrastructure development, with affected wetlands landward of transportation corridors often becoming freshened and flooded due to retention of freshwater drainage from the watershed\textsuperscript{9}. As a result of those many causes for complete or partial tidal restriction in tidal wetlands, inhabited and developed coastal landscapes typically contain a patchwork of unaltered tidal wetlands interspersed with drained, impounded, and partially restricted.

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wetlands, ranging in size from <0.1 km² to multiple km² (Fig. 1). In many cases, if the alteration occurred decades or centuries ago, the parcels may no longer be recognized as former salt marsh by local residents or by the National Wetlands Inventory, but rather as fresh or brackish ponds or wetlands where impounded, or, where drained, former wetland may appear as woodland, shrubland, developed land, or agricultural land.

Restriction or blockage of tidal flows dramatically alters wetlands as habitat for flora and fauna, and further causes major changes in microbial and chemical processes in wetland soil and water. Here we emphasize that those changes can also alter the direction and magnitude of greenhouse gas (GHG) exchange between the wetland and the atmosphere. The fate of soil C in wetlands, and emissions of CH₄ and CO₂, largely depend on water salinity and water table elevation relative to the soil surface, with temperature and plant productivity also having important influence. Under flooded or impounded conditions, soil becomes anaerobic, effectively to the soil surface. Under those conditions, if blockage of tidal flows has caused water salinity to decrease from >18 psu (~50% seawater) to <18, then a substantial increase in CH₄ emission is expected to occur. Salinity is a critical driver because seawater contains abundant sulfate ion, and where sulfate is abundant, sulfate reduction tends to outcompete methanogenesis as a metabolic pathway for microbial decomposition of organic matter. Additionally, sulfate can serve as the terminal electron acceptor in the oxidation of methane to carbon dioxide, further reducing CH₄ emissions under saline conditions. Though measurements are uncommon, CH₄ emissions in inundated, low salinity portions of tidally-restricted wetlands are elevated, and similar to the average of published rates for tidal wetlands at salinity <18 psu. Drainage of both tidal and non-tidal wetlands, on the other hand, drains water from soil pore-spaces and exposes soil organic matter to air and associated oxygen, thereby promoting aerobic microbial respiration of C stocks that have accumulated over centuries to millennia, with CO₂ as the primary product. Drainage-associated soil CO₂ emissions are intense per unit area, and drained tidal wetlands and terrestrial peatlands are significant drivers of change. Drainage and restoration of terrestrial (non-tidal, freshwater) peatlands are not the focus of the present study, except as an informative comparison to GHG consequences of tidal restriction and restoration of tidal wetlands, since rewetting (re-flooding through restoration of natural hydrology) of terrestrial peatlands, to protect soil C stocks from ongoing respiration and loss as CO₂ emissions, is considered to have great potential as a landuse-based climate intervention.

To examine the intensity of GHG emissions due to tidal restriction in salt marshes, as well as the potential for emissions reductions through tidal restoration, we modeled resulting changes over time in the cumulative mass of atmospheric CO₂ and CH₄ emitted or sequestered, and their radiative forcing (RF, a measure of the change in energy in the global atmosphere). The concepts around changes in soil C stocks and in GHG emissions with impoundment or drainage of salt marsh, and with restoration of tidal flows, are diagramed in Fig. 2. Here, "tidal restoration" describes removal or opening of dikes, tide gates, or under-sized culverts to renew tidal water exchange between the salt marsh and coastal ocean, and thus restore natural, or nearer to natural, water level and salinity. We first compared the intensity, per unit area, of cooling (net negative RF) in natural Blue Carbon ecosystems (seagrass beds, salt marshes and mangroves) to intensity of warming due to enhanced CH₄ and CO₂ emissions in impounded and drained salt marsh, respectively (Fig. 3). As further context, we also modeled: climatic cooling (negative RF) from net C storage in U.S. forests; warming (positive RF) from tailpipe emissions of a typical U.S. automobile; and warming (positive RF) from enhanced CO₂ emissions in drained, terrestrial peatlands. We utilized RF calculations in this study to enable comparison of the climatic impact of CO₂ and CH₄ emissions or emissions reductions that are sustained over a period of decades to centuries. Use of standard global warming potentials (GWP) to compare the impact of different GHG is not appropriate in the context of sustained emissions that are characteristic of ecosystems and their management. Rather, standard GWP calculations are intended as a policy tool based on analysis of the atmospheric perturbation lifetime of a discrete pulse of GHG emission. Standard GWP calculations underestimate the climatic impact of sustained CH₄ emissions or emissions reductions relative to sustained changes in CO₂ emissions.

In the present study, emission factors (EF), or typical CH₄ and CO₂ emission rates for each ecosystem type or condition, were derived from the literature and from IPCC tier 1 EF (Table 1). To represent the range and variability of CH₄ emissions in low salinity tidal wetlands, and therefore expected range of emissions and EF in freshwater, impounded wetlands, we modeled the time course of RF using three different CH₄ EF based on the median and average of published emission rates at salinity <18, as well as the IPCC EF for low salinity tidal wetlands. The wide range in literature estimates for CH₄ emissions in low salinity tidal wetlands is likely due to variations across sites in other drivers of emissions, including water level, plant productivity and temperature. Therefore, it is important to examine a range of EF for these low salinity wetlands, to represent the diversity of emissions. For CO₂ EF, we used IPCC Tier 1 EF from drained and rewetted terrestrial and tidal wetlands, respectively (Fig. 3). As further context, we also modeled: climatic cooling (negative RF) from net C storage in U.S. forests; warming (positive RF) from tailpipe emissions of a typical U.S. automobile; and warming (positive RF) from enhanced CO₂ emissions in drained, terrestrial peatlands. We utilized RF calculations in this study to enable comparison of the climatic impact of CO₂ and CH₄ emissions or emissions reductions that are sustained over a period of decades to centuries. Use of standard global warming potentials (GWP) to compare the impact of different GHG is not appropriate in the context of sustained emissions that are characteristic of ecosystems and their management. Rather, standard GWP calculations are intended as a policy tool based on analysis of the atmospheric perturbation lifetime of a discrete pulse of GHG emission. Standard GWP calculations underestimate the climatic impact of sustained CH₄ emissions or emissions reductions relative to sustained changes in CO₂ emissions.

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**Results**

**Radiative Forcing per Unit Area.** Results indicate that climatic warming, or positive RF, due to tidal restriction (both impoundment and drainage) is large on a per unit area basis relative to the magnitude of cooling from C sequestration in forests and in unaltered coastal wetlands (Fig. 3). For instance, over a period of 20 years following initial alteration, CH₄ emissions from an impounded and freshened salt marsh can result in a cumulative RF that is a factor of ~3 to 20 greater than the magnitude of climatic cooling due to net C stock increase in continental U.S. forests and to soil C sequestration in unaltered salt marsh, and a factor of 1.5 to 10 greater than
Figure 1. Satellite photographs of tidally-restricted wetlands on the U.S. Atlantic coast, demonstrating a range of histories, causes and sizes: (a) The Herring River basin within the National Park Service Cape Cod National Seashore, Massachusetts. The solid black outline indicates 4 km² of former tidal salt marsh. The estuary and marsh were diked in the first decade of the 20th century, in an unsuccessful attempt to control mosquitoes. Portions of the basin are now drained marsh, and are characterized by acid–sulfate soil, loss of elevation, and colonization by native and non-native grass, shrub and trees. Other portions are impounded, freshened and colonized by native and non-native fresh and brackish wetland grasses. The wetland is proposed for tidal restoration, and pre-restoration carbon cycle and ecological data collections are underway. (b) An impoundment on ~0.05 km² of former salt marsh on the south shore of Cape Cod, Massachusetts, caused by a 19th century railroad that was subsequently converted to a bicycle path. Similar incidental impoundments are common and widespread along coastal transportation corridors. (c) The Prime Hook National Wildlife Refuge in the state of Delaware. The Refuge is a 41 km² complex of managed wetland and open water. In response to repeated storm damage, plans were in development to upgrade the artificial berm that impounded 16 km² of freshwater impoundment (https://www.fws.gov/refuge/prime_hook/). Prior to completion of the upgrades, Hurricane Sandy breached the artificial berm in 2012 and restored saline tidal flows to the impoundment. Subsequently, deliberate salt marsh restorations were undertaken. The inadvertent breach likely resulted in significant GHG emission reductions. Maps created by georeferencing each image from Google Earth (image source credits: (a) Google, Landsat/Copernicus, copyright DigitalGlobe; (b) Google, Landsat/Copernicus; (c) Google, USGS), base map and scale bars were added using ArcGIS for desktop version 10.3.1 (http://www.esri.com), and images arranged using Adobe Photoshop CC 2017 vers. 18.0.1 (http://www.adobe.com/products/photoshop.htm).
sequestration in natural mangrove. Over those 20 years, CH$_4$ emissions from one hectare of impounded wetland are equivalent to RF from 20 years of continuous tailpipe emissions of 1.7 to 6.3 automobiles (Fig. 3).

To quantify the potential for emissions reductions through wetland management, we calculated the time-course, over decades to centuries, of cooling (negative RF) predicted to occur due to CH$_4$ and CO$_2$ emissions reductions resulting from tidal restoration in tidally-restricted salt marsh, based on the cumulative difference between the contrasting management action and no action scenarios (Fig. 4). We compared climatic cooling due to tidal restoration to cooling from other wetland carbon management options, including creation of new salt marsh or sea grass beds as biological C sequestration projects, and to rewetting of terrestrial peatland to cease the high rate of CO$_2$ emissions from drained peatland soils. In general, the tidal restorations in salt marshes, to restore
natural salinity and water levels, will be dramatically more effective at cooling climate than other wetland-based climate change interventions (Fig. 4). The time course of cumulative RF resulting from reduction in the rate of CH$_4$ production in previously flooded and freshened soils (Fig. 4a,d), and reduction in CO$_2$ production in previously drained soils (Fig. 4b,e) indicate sizeable, rapid and sustained climatic cooling. Despite the high rate of C sequestration in salt marshes and mangroves, building and planting of new marsh, mangrove or seagrass bed, provides relatively modest climatic cooling (Fig. 4c,f). Avoided CO$_2$ emissions from rewetting of drained, terrestrial peatlands (Fig. 4c,f) can be substantial, but are partially offset by resumption of CH$_4$ emissions in soil rewetted with fresh water20 (Table 1).

Geography of Tidally-Restricted Wetlands and Scale of Emissions. There is substantial potential for GHG reductions based on the intensity of emissions, and on the areal extent of tidally-restricted wetlands in developed landscapes. Tidally-restricted wetlands are widespread throughout the inhabited world (e.g.7,8,13–15,18–31). Infrastructure and activities that alter wetland and coastal hydrology have been closely associated with intensity of human development and economic activity in terrestrial and marine portions of the coastal landscape31,32, and therefore are expected to increase in the future, particularly in portions of the world that are undergoing rapid development23,31. Quantitative geographical data on tidally-restricted wetlands are limited, and indeed the global area of tidal wetlands as a whole is not well-constrained 15. Thus, it is not possible to estimate global scale GHG emissions due to restrictions. As an indication of scale, we used mapped and areal data along portions of the U.S. Atlantic coast on managed impoundments and on incidental impoundments associated with transportation infrastructure. The analysis resulted in an estimate that those tidal restrictions have caused a reduction in salinity, and enhancement of CH$_4$ emissions, in approximately 27% of tidal wetlands on the U.S. Atlantic coast (Table 2). The impoundment and freshening result in enhanced CH$_4$ emissions in the range of 28,000 to 145,000 tonnes (t) CH$_4$ y$^{-1}$. It is worth emphasizing once again that these are sustained emissions, and thus RF due to those rates of continuous emission over a 20-year period are equivalent to 20 years of continuous emissions from 0.6 to 3.1 million automobiles (see Methods). Note that this is an estimate of CH$_4$ emissions alone, and does not consider changes in C sequestration rate in soil, nor loss of existing soil C stocks and enhanced soil CO$_2$ emissions, due to changes in water level and salinity. Further, the area and emissions calculations consider only the effect of freshening and impoundment of tidal wetlands, and do not include area or emissions of tidal wetlands that have been drained, unintentionally nor intentionally, for purposes such as agriculture or land development.

**Discussion**

**Key features of avoided wetland methane emissions.** The results of this study indicate that, despite a high rate of C sequestration in coastal wetlands, in many cases tidal restoration in salt marshes will have dramatically greater potential per unit area as a climate change intervention than the other examined ecosystem management actions. Coupled with the common, widespread occurrence of tidal restrictions, there is significant potential for GHG emissions reductions through tidal restoration in salt marshes. Though we did not address tidal restriction and restoration in mangroves, many of the processes and rates presented for salt marshes would be expected to be similar in mangroves. We further note that avoiding tidal restrictions with future coastal development would have similar and perhaps greater benefits in terms of avoided GHG emissions than the restoration scenarios examined in this study. Here we discuss several features of tidal restoration, and particularly avoided methane emissions, that highlight further advantages relative to enhanced CO$_2$ sequestration in other
Table 1. Methane and carbon dioxide emission rates utilized as inputs to model radiative forcing in a series of wetland settings and restoration scenarios. aPoffenbarger et al.18, Table 4.12. bHiraishi et al., Table 2.1, average of rates for boreal, temperate and tropical peatlands; Scenario in which major respiratory loss of soil C continues until restored. cIn all scenarios, assumed lag of 5 years, associated with ecosystem establishment, prior to initiation of new C sequestration in soil18. dHiraishi et al., Table 2.3, assumed equivalent to rate for drained inland peatlands. ePoffenbarger et al., Table 3.1, average of rates for boreal and temperate peatlands. fHiraishi et al., Table 4.13; Scenario in which soil has been drained for <30 years and major respiratory loss of soil C continues until restored. gAssumed net C sequestration equivalent to saline wetland. hScenario in which soil has been drained for several decades, and major respiratory loss of soil C has gone to completion. iHiraishi et al., Table 3.3, average rate for boreal and temperate peatlands. jHiraishi et al., Table 4.12. kHiraishi et al., Table 3.1, average of rates for forest. lHiraishi et al., Table 4.14. mKeller et al., Table 1.18, Table 4.13; Scenario in which soil has been drained for <30 years and major respiratory loss of soil C continues until restored. nHiraishi et al., Table 2.3, average of compiled data from wetlands with salinity <18. oHiraishi et al., Table 4.14; Scenario in which soil has been drained for several decades, and major respiratory loss of soil C has gone to completion. pHiraishi et al., Table 3.3, average rate for boreal and temperate peatlands. qHiraishi et al., Table 4.13; Scenario in which soil has been drained for <30 years and major respiratory loss of soil C continues until restored. rHiraishi et al., Table 2.1, average of rates for forest. sHiraishi et al., Table 4.13; Scenario in which soil has been drained for several decades, and major respiratory loss of soil C has gone to completion. tHiraishi et al., Table 3.3, average rate for boreal and temperate peatlands. uHiraishi et al., Table 4.13; Scenario in which soil has been drained for <30 years and major respiratory loss of soil C continues until restored. vHiraishi et al., Table 2.1, average of rates for forest. wHiraishi et al., Table 4.13; Scenario in which soil has been drained for <30 years and major respiratory loss of soil C continues until restored. xHiraishi et al., Table 2.1, average of rates for forest. yHiraishi et al., Table 4.13; Scenario in which soil has been drained for <30 years and major respiratory loss of soil C continues until restored. zHiraishi et al., Table 2.1, average of rates for forest.

landuse-based climate change interventions, due to key aspects that result in rapid, substantial, and sustained reduction in RF:

First, CH4 has a radiative efficiency, or instantaneous heat absorption capacity, that is a factor of ~73 greater than that of CO2, and a factor of ~94 greater with consideration of indirect effects of CH4 on atmospheric chemistry35. Second, reductions in both CH4 and CO2 emissions are soil microbial responses, and thus, though more data are needed, onset following restoration of natural water level and salinity is likely to be rapid46,57 relative to development of restored C storage capacity following full ecosystem recovery or development in biological sequestration projects58. In the present study, ecosystem recovery was represented as a lag of 5 years prior to onset of the full rate of C sequestration in salt marsh soil (Table 1, note k). Thus, climatic cooling due to avoided CH4 emissions is relatively large on short timescales. In contrast, in projects that promote biological C sequestration in wetland soil, or similarly in forest biomass, accrual of climate change mitigation benefits is a process of gradual accumulation on a timescale of centuries (see “New salt marsh” curve in Fig. 4c and f), due to both the gradual rate of ecosystem development following restoration, and to the modest radiative efficiency of CO2, coupled with a long atmospheric perturbation lifetime. As a result, at 15 years post restoration, for instance, negative RF (cooling) due to CH4 reduction (Fig. 4a) was a factor of 12 to 59 greater than that due to CO2 sequestration in new salt marsh (Fig. 4c).

Third, in the absence of restoration, enhanced CH4 emissions in an impounded and freshened wetland can be expected to continue indefinitely, due to a continuous supply of new organic matter substrate for methanogenesis from ongoing primary production55. Thus, although CH4 is considered to be a short-lived climate pollutant, with an atmospheric perturbation lifetime of only 12.4 years, restoration can avoid a long-term, sustained emission, and the cumulative mass of avoided CH4 and magnitude of climatic cooling will continue to increase indefinitely through time (Fig. 4d). In contrast, although avoidance of CO2 emissions by tidal restoration in drained salt marshes can have an RF in the first few decades that is as great as avoided CH4 in freshened marshes, the duration and ultimate cumulative magnitude of enhanced CO2 emissions in a drained wetland is limited by the mass of the C stock contained in soil above the level of drainage. Therefore, where wetlands have been drained for decades to centuries, the benefit of restoration may be relatively minor (Fig. 4b emission factor 1). Where pre-restoration CO2 respiration is rapid, re-wetting will be of substantial benefit (Fig. 4b emission factor 2), but the anticipated duration of that benefit will be uncertain (Fig. 4e). Based on IPCC Tier 1 marsh soil C stocks and respiration rates in drained marshes, the stock in the top meter of soil would be consumed within approximately 3 decades following drainage18.

A fourth critical feature of wetland tidal restoration as a climate change intervention is that the CH4 emissions thus avoided will have what can be referred to as “inherent permanence”: following tidal restoration and onset of reduced methane emissions, the emissions thus avoided cannot be cancelled, even if emissions were to resume in the future due to a change in ecosystem status. For instance, if a tidal restriction were re-established 30

| Restoration Scenario | CH4 prior (g C m⁻² y⁻¹) | CH4 post (g C m⁻² y⁻¹) | CO2 prior (g C m⁻² y⁻¹) | CO2 post (g C m⁻² y⁻¹) |
|----------------------|--------------------------|-------------------------|--------------------------|--------------------------|
| 1. Tidal restoration to: Flooded & freshened salt marsh | 8.4⁴ | 0.46⁴ | -91¹ | -91¹ |
| 2. Tidal restoration to: Drained salt marsh | 41.6¹ | 0.46 | -91 | -91 |
| 3. Create salt marsh | 0 | 0.46 | 0 | -91 |
| 4. Create seagrass bed | 0 | 0 | 0 | -43³ |
| 5. Re-wet drained peatland | 6.4⁶ | 12.15⁸ | 517.5⁸ | -15.5⁷ |
| 6. Saline mangrove | 0.46² | -16² | -69.6⁴ | -91 |
| 7. Terrestrial forest | - | - | - | - |
years after restoration, emissions would resume, but the biosphere and atmosphere would maintain for extended time a deficit of heat and of GHG equivalent to 30 years of reduced emissions. Further, since GHG emissions from tidally-restricted wetlands are caused by human activity, they are anthropogenic emissions, and reducing anthropogenic wetland emissions will have an effect on climate that is equivalent to reduced emission of an equal quantity of fossil fuel GHG. Therefore, avoiding CH4 emissions as a landuse-based climate change intervention is a distinct concept from biological C sequestration projects to enhance C storage in forest or wetland biomass or soil. Biological C sequestration accumulates climate benefit at a relatively slow rate, and the C sink must persist for a century or more to have significant impact on climate (see Fig. 4f, new salt marsh example). During that time, there is a risk that the stored C will be rapidly returned to the atmosphere, through processes such as fire or ecosystem degradation and organic matter decomposition. As a result, the monetary and climate change mitigation value of biological C sequestration projects must be discounted in C markets and in trading programs to account for the “non-permanence risk” that stored C will be returned to the atmosphere36,37. To be clear, this discussion is not intended to suggest that temporary reductions in CH4 emissions will achieve effective management of climate change, but rather to highlight a distinction between avoided emissions and C sequestration.

Implications for GHG emission inventories and reduction programs. Tidal restoration to avoid CH4 emissions is a relatively new concept for Blue Carbon management, and warrants consideration within GHG markets and emission offset programs, national and local-level efforts to reduce emissions, and national GHG inventories based on IPCC guidance. Within international climate change treaty negotiations there has been increasing emphasis on co-benefits accruing from GHG mitigation actions that involve improved landscape management, including forests, soils and potentially coastal and marine ecosystems. In 2013, the International Panel on Climate Change released the 2013 Supplement to the 2006 IPCC Guidelines for National GHG Inventories: Wetlands18, to guide accounting of emissions and management of wetland soils. The Supplement includes guidance on CO2 emissions for drained wetlands, but does not consider GHG emissions from impoundments in general, nor freshened or impounded wetlands due to tidal restriction. The U.S. has recently completed the first national inventory worldwide to include coastal wetlands in a National GHG Inventory, and the UN Framework Convention on Climate Change has asked countries to provide feedback on experiences with application of the Supplement, to inform future expansion of the accounting guidance where science is available. The analysis presented here indicates that tidally restricted wetlands meet the primary criteria for inventoried ecosystems in that

Figure 4. Time-course of cumulative radiative forcing modeled as the difference between emissions under scenarios of no action and of restoration (Table 1). Negative values indicate cumulative climatic cooling (picowatt-years per m2 of earth surface) per square meter of wetland restored, over a 30-year period (a,b,c) and a 200-year period (d,e,f). Methane emission reductions due to tidal restoration in impounded (flooded and freshened) former salt marsh (panels a and d) result in large reduction in RF, particularly on a timescale of decades to a century. Carbon dioxide emission reductions due to rewetting of drained salt marsh (b,e) and terrestrial peatland (c,f) also have relatively large climate benefit, but depend on assumptions regarding the period of time that the wetland would have been maintained in a continuously drained condition, and the period of time that the finite carbon stock in the drained soil would continue to respire (uncertainty regarding timeframe of benefit indicated by dashed lines; see text).
they are managed landscapes, with emissions and sinks of substantial magnitude and rate of change. If other countries ultimately follow suit, then inclusion of those emissions in the U.S. Inventory will promote widespread recognition and management of the issue, and justify development of CH₄ EF for tidal restrictions and impoundments in IPCC guidance for GHG inventories. Note that since ecological restoration through tidal reconnection has been a widespread practice in some parts of the world, restorations that were conducted since the baseline year, typically 1990, may represent considerable, but as yet unrecognized, emissions reductions.

Wetland C and GHG management can also play a role as an additional method for reducing regional, state and local GHG emissions, including through regulatory or voluntary carbon markets or cap & trade programs. Verified Carbon Standard (VCS) has developed a Methodology for crediting tidal wetland carbon offset projects, designed for both voluntary and regulatory C markets worldwide, and within that methodology there is accommodation for avoided CH₄ projects as a creditable activity. The implications of the results presented in the present study may support continued development of concepts related to crediting of climate change mitigation through avoided CH₄ emissions in wetlands, to reflect the distinctions between avoided CH₄ and C sequestration projects:

First, at present, in the VCS methodology, there must be reasonable expectation that a wetlands-based, emissions offset project will have a lifetime of at least 100 years. The mandated lifetime is in response to the concept that in projects that promote C sequestration in soil or biomass, the sequestered C must be kept out of the atmosphere for an extended time to have significant climate change mitigation value. However, in the context of avoided CH₄ emissions, the 100-year time frame is arbitrary, since such offset projects do not rely on storage of C in a soil or biomass reservoir, and they accrue considerable, long-term and quantifiable mitigation benefit on timeframes of decades. Given that the persistence of any project for 100 years is difficult to assure, a shortened minimum lifetime for avoided CH₄ projects could increase utilization of tidal wetlands as offset projects.

Second, the application of a 100-year CH₄ global warming potential of 21 to calculate carbon dioxide equivalents (CO₂e) of avoided CH₄ emissions underestimates the climate change mitigation value of the avoided CH₄ to a substantial degree. As an example, if we were to apply that GWP value in a CO₂e approach to scenarios 1 and 3 in Table 1 (tidal restoration in salt marsh to avoid CH₄ emissions vs. creation and planting of new salt marsh to enhance C sequestration), the range of CH₄ emission factors would result in a calculated GHG reduction benefit for avoided CH₄ (scenario 1) that is similar to the C sequestration benefit in scenario 3 (scenario 1 = 0.7 to 3.5 times the value of scenario 3; not shown). The ratio would remain constant for the 100-year lifetime of the projects. That result is in contrast to the more nuanced calculation of change in radiative forcing that we noted previously in the manuscript, in which reduction in RF (cooling) due to CH₄ reduction (Fig. 4a) was a factor of 12 to 59 greater than RF reduction due to CO₂ sequestration in new salt marsh (Fig. 4c), at 15 years post restoration. Even at 100 years post restoration, the benefit of avoided CH₄ emissions (scenario 1) remains a factor of 3 to 11 greater (Fig. 4d and f). As noted previously, a standard GWP approach is inappropriate for sustained emissions or

| Location                        | Tidal Wetland in Study Area (km²) | Wetland Area Affected (km²) | Fraction of wetland area affected (%) |
|---------------------------------|----------------------------------|----------------------------|---------------------------------------|
| Transportation-Related Restrictions: |                                    |                            |                                       |
| Southern Maine                  | 32                               | 9                          | 28                                    |
| New Hampshire                   | 26                               | 5                          | 20                                    |
| Massachusetts                   | 212                              | 58                         | 27                                    |
| North Shore                     | 113                              | 6                          | 5                                     |
| Cape Cod                        | 70                               | 20                         | 28                                    |
| Buzzards Bay                    | 33                               |                            |                                       |
| Rhode Island                    | 16                               | 11                         | 70                                    |
| Total Transportation            | 286                              | 84                         | 29                                    |
| Impounded for Waterfowl or Mosquito Management: |                                    |                            |                                       |
| N. Carolina                     | 643                              | 21                         | 3                                     |
| S. Carolina                     | 2,041                            | 285                        | 14                                    |
| Georgia                         | 1,590                            | 32                         | 2                                     |
| Florida (Atlantic)              | 778                              | 143                        | 18                                    |
| Total Impounded for Wildlife Management | 5,053                            | 482                        | 10                                    |
| Total impounded tidal wetlands U.S. Atlantic coast, Transportation + Wildlife | 9,710                            | 3,787                      | 39                                    |
| Total area impounded and freshened: | 9,710                            | 2,650                      | 27                                    |

Table 2. Geography of tidally restricted wetlands on the Atlantic coast of the United States. Available published data were compiled on surface areas of managed wetland impoundments and incidental, full or partial impoundment resulting from transportation infrastructure. Tidally restricted wetland areas were used as a sample to estimate restricted wetland area as a percentage of total tidal wetland area. Percentages were then extrapolated to the areas without data. Based on salinity data and vegetation as an indicator of salinity, it was estimated that 70% of the impounded wetlands were fresened as a result. (see Methods).
sinks that are typical of ecosystems\(^4\). A 100-year GWP that is calculated based on simulation of a single pulse of CH\(_4\) with an atmospheric lifetime of 12.4 years, integrates 12 years of heat retention across 100 years of impact, and ignores the fact that with sustained emissions a pool of CH\(_4\) would be maintained in the atmosphere indefinitely, rather than for just 12 years.

There is ongoing debate regarding the value, in terms of addressing climate change, of reducing CH\(_4\) emissions vs CO\(_2\) emissions, and the debate is relevant to selection of a GWP value for CH\(_4\) and crediting timeframe, in C markets and offsets. Though there is not a simple answer, it is clear that aggressive reduction in emissions of CH\(_4\) and other short-lived climate pollutants (SLCP) can reduce the rate of increase in global temperature during the current century, and can reduce peak temperature if coupled with rapid elimination of CO\(_2\) emissions\(^{39}\). Further, it has been shown that, although CH\(_4\) is a short-lived climate pollutant, during the time that it resides in the atmosphere, a portion of the heat that it traps within the biosphere will be retained in oceans and contribute to sea level rise for several centuries\(^{39}\). However, if actions to reduce emissions of SLCP result in delayed reductions in CO\(_2\) emissions, then higher peak temperatures will result\(^{38}\). Therefore, society is likely to benefit most from simultaneous, but separate, efforts to reduce both CO\(_2\) and SLCP emissions, to manage both short-term and long-term climate change. Utilization of offsets and emissions trading programs therefore requires relative evaluation of pollutants with distinct interactions with climate. Solutions might be to limit maximum utilization of offsets in C markets, as is done in California's Cap and Trade Program\(^{40}\), or to develop separate reduction and trading programs for CO\(_2\) and SLCP. For instance, the state of Massachusetts has specific emissions reduction goals for natural gas transmission\(^{43}\), New Zealand has separate targets for methane reductions in their Intended Nationally Determined Contribution\(^{42}\), and many national and subnational governments have programs to reduce methane emissions from agricultural sources\(^{41}\). As described in the present study, degraded tidal wetlands can be significant sources of both CO\(_2\) and CH\(_4\), while intact wetlands are generally a strong CO\(_2\) sink. Thus, wetland management can contribute to management of both short and long-lived climate pollutants.

Finally, beyond C markets and national GHG inventories, awareness of anthropogenic emissions in tidally-restricted wetlands, and the potential for mitigation through ecosystem restoration, may be of significant value to land management agencies with wetlands, tidal impoundments, and coastal infrastructure under their purview, such as the U.S. Fish and Wildlife Service, the National Park Service, local and state government entities, and private land owners. In wetlands that are intentionally managed, in many cases opening tidal restrictions would be prohibited by ongoing important land uses, or where restoration would put developed land at risk of flooding. But, increasingly, decisions are being forced, by accelerating sea level rise and increasing storm intensity, regarding whether to spend resources to preserve and upgrade aging dikes and tide gates that maintain wetlands in drained or impounded condition (for example see Fig. 1c, Prime Hook NWR). Management decisions and objectives are based in part on anticipated changes in the value of ecosystem services to society, including coastal protection, quality and type of habitat, recreational utility, and others. Reduction in GHG emissions has significant value to society, and can be highlighted as a benefit, regardless of whether the benefit is monetized. Even where opening tidal restrictions is contraindicated, there still may be potential for management of water levels and salinity at targeted times of the year, specifically for the purpose of reducing GHG emissions and promoting C sequestration, while maintaining utility of the landscape for those other uses or protecting infrastructure from flooding.

In other circumstances, the decisions may be less complicated in terms of tradeoffs of services, such as where tidal restriction is due to inadvertent, transportation-related obstruction of flow, and thus there is no current beneficial use of the impaired wetland (for example, see Fig. 1b). In some cases, ecosystem restoration through opening of tidal restrictions can be consistent with coastal resilience planning, since exposure to tidal flow will tend to promote resumption of natural accretion of wetland elevation in response to sea level rise, resulting in enhanced protection of the infrastructure landward of the wetland.

There is significant potential for C and GHG management in coastal wetlands, but continued advances are needed on the scientific basis for policy frameworks and for quantification of emission factors in response to management actions. There has been particularly little study of changes in soil and biomass C stocks, and of GHG fluxes, in response to tidal restriction and tidal restoration in salt marshes and mangroves. Further, in all areas of the world the state of knowledge is poor regarding the geographic distribution of tidal wetlands, of salinity within those wetlands, of occurrence of tidally-restricted salt marshes and mangroves, and of the potential for tidal restoration. The findings presented here therefore suggest a research agenda to study GHG emissions and the fate of carbon stocks in natural, restricted and restored wetlands, across geographic settings, as well as mapping occurrence and characteristics of tidal wetlands and tidally-restricted wetlands.

Methods

Radiative Forcing—To calculate radiative forcing, we modeled the time-course of change in the atmospheric inventory of each GHG per square meter of wetland, based on emission rate and on rate of destruction or removal from the atmosphere. Methane was modeled as a single reservoir with atmospheric perturbation lifetime of 12.4 years, and first order removal due to oxidation to CO\(_2\). RF from management of methane emissions includes forcing from the resulting changes in both the pool of atmospheric CH\(_4\) and CO\(_2\) produced through CH\(_4\) oxidation. We calculated the mass of CH\(_4\) in the atmosphere, per square meter of wetland at time \(t\), according to Neubauer and Megenical\(^{18}\) eq. 1:

\[
M_{CH_4,C}(t) = F_{CH_4,-C}dt + \left[ M_{CH_4,-C}(t-1) \times e^{-dt/T_{CH_4}} \right] 
\]

where \(M_{CH_4,-C}\) is the mass of atmospheric CH\(_4\) (g C m\(^{-2}\)), \(F_{CH_4,-C}\) is the emission factor (g C m\(^{-2}\) y\(^{-1}\)), \(dt\) is the time step (0.2 y), and \(T_{CH_4}\) is the atmospheric perturbation lifetime of CH\(_4\).
Mass of atmospheric CO$_2$ was modeled based on a synthesis of models, utilizing four non-interacting reservoirs with distribution of emissions among the reservoirs and atmospheric perturbation lifetimes in the reservoirs as in Joos et al.$^{44}$, Table 5 and IRF model within that publication. The mass of CO$_2$ in the atmosphere was calculated as:

$$M_{CO_2} = \sum_{i=1}^{4} f_i (F_{CO_2} - C_{i-1}) + \left[ M_{CO_2} - C_{i-1} \right] \times e^{-df(T_{CO_2})},$$

where $f_i$ is the fraction of CO$_2$ emissions distributed to the $i$th reservoir, and other terms are as defined for the CH$_4$ model.

In both models, the instantaneous RF was calculated at each time step as the product of the mass of each gas in the atmosphere and its radiative efficiency$^{33}$ ($1.75 \times 10^{-15}$ W m$^{-2}$ (kg CO$_2$)$^{-1}$ and $1.28 \times 10^{-13}$ W m$^{-2}$ (kg CH$_4$)$^{-1}$) with CH$_4$ radiative efficiency adjusted by a factor of 1.65 to account for indirect effects$^{45}$. Cumulative RF is the sum of instantaneous RF over a given time period. N$_2$O emissions were not considered due to insufficient data.

Geography—Large scale, geographic data on tidal restrictions are rare, but a large number of examples and case studies of restriction and restoration occur within the literature. As an indication of scale, approximately 50% of pre-development tidal saltmarsh area in the U.S. has been destroyed by human activities, with impoundment and drainage as prominent causes$^{43}$, and a quarter of remaining coastal wetlands in the state of Massachusetts, for example, is landward of nearly 600 transportation-related tidal restrictions$^{46-48}$. To estimate the scale of tidally restricted wetlands area and the proportion of existing wetlands that are restricted, we compiled published information on the occurrence of tidal restrictions managed wetland impoundments and incidental, full or partial impoundments caused by transportation infrastructure (Table 2). In the northeast U.S., wetland and impoundment areas landward of primarily transportation-related restrictions comprise 29% of total tidal wetland. In the southeast U.S. 10% of total tidal wetland area is diked and impounded for waterfowl and mosquito management. The tidal wetlands in the regions and states represented in the analysis comprise 55% of the total tidal wetland on the U.S. Atlantic coast, and thus are expected to be approximations of the entire coast and may be similar to many developed coastal regions elsewhere. Based on the assumption that similar levels of transportation-related and diked restrictions occur throughout the U.S. east coast, we estimate that 39% of total tidal wetland area on the U.S. Atlantic coast, ~3,800 km$^2$, is above tidal restrictions or blockages. Montague et al.$^{23}$, reported that 87% of the southeast U.S. mosquito and waterfowl impoundments included in Table 2 had salinity less than 16 psu. Thus, they are expected to be significant methane sources. It is not known what proportions of restricted wetlands related to transportation infrastructure are in freshened, drained, or relatively unaltered condition. In Massachusetts 70% of tidally restricted wetland area is described as freshwater marsh or fresh shrub, with 52% of the vegetated area colonized by invasive Phragmites australis, while the remainder remains brackish to saline marsh$^{49,50}$. In general, where purposeful actions have not been taken to drain tidally-restricted wetlands, such as is the case with incidental restrictions caused by transportation infrastructure, flooding and freshening are the most likely result. Thus, we can approximate that 70% of restricted wetland area compiled here, or 2,650 km$^2$, is in a flooded/freshened condition. To calculate minimum and maximum emissions rates of CH$_4$ from those wetlands, we multiplied the area by the minimum and maximum annual emissions rates of CH$_4$ from impounded wetlands (Table 1, Scenario 1, EF1 and EF3 pre-restoration minus post-restoration). For context, the annual emission rates from 2,650 km$^2$ of freshened wetlands were compared to average annual tailpipe emissions from a U.S. automobile$^{43}$, at 4.690 kg CO$_2$ y$^{-1}$, with the comparison calculated as cumulative RF over a 20-y period of continuous emissions at those rates. The resulting anthropogenic wetland emission estimate was 28,000 to 145,000 tonnes CH$_4$ y$^{-1}$, with RF equivalent to 20 years of continuous emissions from 0.6 to 3.1 million automobiles (not shown).

Data availability. The data that support the analyses presented are included in tables and in cited references. Results of radiative forcing calculations are available upon request from the lead author.

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K.D.K. conducted the analyses and led writing of the manuscript. S.C., S.M.V. and J.T. helped to conceive and draft the manuscript.

Additional Information
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