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MANAGEMENT APPLICATIONS

The Science and Policy of the Verified Carbon Standard Methodology for Tidal Wetland and Seagrass Restoration

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Abstract
The restoration of tidal wetland and seagrass systems has the potential for significant greenhouse gas benefits, but project-level accounting procedures have not been available at an international scale. In this paper, we describe the Verified Carbon Standard Methodology for Tidal Wetland and Seagrass Restoration, which provides greenhouse gas accounting procedures for marsh, mangrove, tidal forested wetland, and seagrasses systems across a diversity of geomorphic conditions and restoration techniques. We discuss and critique the essential science and policy elements of the methodology and underlying knowledge gaps. We developed a method for estimating mineral-protected (recalcitrant) allochthonous carbon in tidal wetland systems using field-collected soils data and literature-derived default values of the recalcitrant carbon that accompanies mineral deposition. We provided default values for methane emissions from polyhaline soils but did not provide default values for freshwater, oligohaline, and mesohaline soils due to high variability of emissions in these systems. Additional topics covered are soil carbon sequestration default values, soil carbon fate following erosion, avoided losses in organic and mineral soils, nitrous oxide emissions, soil profile sampling methods, sample size, prescribed fire, additionality, and leakage. Knowledge gaps that limit the application of the methodology include the estimation of CH_4 emissions from fresh and brackish tidal wetlands, lack of validation of our approach for the estimation of recalcitrant allochthonous carbon, understanding of carbon oxidation rates following drainage of mineral tidal wetland soils, estimation of the effects of prescribed fire on soil carbon stocks, and the analysis of additionality for projects outside of the USA.

Keywords Greenhouse gas accounting · Carbon sequestration · Methane and nitrous oxide emissions · Allochthonous carbon · Blue carbon · Coastal restoration

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Introduction

The restoration and creation of tidal wetlands have great potential to attract carbon financing due to the high soil carbon sequestration rates and large pools of carbon contained within these systems (Bridgham et al. 2006; Duarte et al. 2013). High rates of photosynthetic carbon fixation, low rates of soil organic matter decomposition under frequently saturated conditions, and increasing soil volume with accretion combine to generate large carbon sequestration rates. Tidal wetland ecosystems sequester soil carbon more efficiently than terrestrial ecosystems—soil carbon burial rates in coastal systems range from 18 to 1713 g m^{-2} $year^{-1}$, compared with a range of 0.7–13.1 g m^{-2} $year^{-1}$ for established terrestrial forests (Mcleod et al. 2011). Restoration practices can serve to establish or improve wetland conditions, thereby reestablishing high carbon sequestration rates and preventing the oxidation of existing carbon stocks (Pendleton et al. 2012). Accounting of greenhouse gas fluxes in coastal wetland systems is of increasing interest as a means to estimate greenhouse gas mitigation benefits. Restoration projects can convert such benefits into carbon credits under existing standards—such as the Verified Carbon Standard (VCS)—and gain access to markets that attach a monetary value to such credits, thus creating a funding mechanism for tidal wetland restoration.

Tidal wetlands are generally net sources of the greenhouse gases CH_4 and N_2O, which in some systems may substantially offset or exceed the carbon benefits of restoration. Methane emissions in tidal wetlands are partially dependent on salinity conditions due to the presence of sulfate in sea water, which allows sulfate-reducing bacteria to outcompete methanogens for available carbon substrates (Bridgham et al. 2013). Sulfate levels tend to be sufficient to consistently minimize CH_4 emissions in polyhaline systems (> 18 ppt); emissions are highly variable in lower salinity systems (Poffenbarger et al. 2011). Nitrous oxide emissions are generally low in tidal wetlands except in cases with high external nitrogen inputs because highly anaerobic conditions lead to a low ratio of N_2O/N_2 emissions (Firestone et al. 1980).

The VCS is a not-for-profit organization that develops and manages greenhouse gas emission accounting standards to allow vetted projects to receive carbon credits that are tradable in voluntary markets. Since its launch in 2007, the VCS has become the largest standard in the agriculture, forestry, and other land use (AFOLU) sector of carbon offsets and has initiated projects and methodologies for forest conservation, improved forest management, and agricultural land management (Verified Carbon Standard 2013; Hamrick and Goldstein 2015). The Wetlands Restoration and Conservation (WRC) category is the most recent project category in the VCS (Verified Carbon Standard 2013); it offers comprehensive guidance for how to account for greenhouse gas removals and emission reductions across tidal wetland ecosystems, guidance on eligible project categories, greenhouse gas sources and carbon pools, baseline determination, leakage calculation, and greenhouse gas emission reductions and removals calculation. To date, two methodologies have been approved under the VCS WRC category—the Methodology for Coastal Wetland Creation and the Methodology for Tidal Wetland and Seagrass Restoration.

In this paper, we discuss and explain essential science and policy components of the VCS Methodology for Tidal Wetland and Seagrass Restoration (Emmer et al. 2015a, b), including limitations and weaknesses of the methodology resulting from scientific and policy research gaps. Our guiding principle in developing this methodology was to design monitoring and verification requirements that are rigorous, comprehensive, and scientifically credible while being feasible to implement.

Overview of the VCS Methodology for Tidal Wetland and Seagrass Restoration

The VCS Methodology for Tidal Wetland and Seagrass Restoration provides greenhouse gas accounting procedures for both restoration and creation of marshes, mangroves, seagrasses, and forested tidal wetlands (Emmer et al. 2015a, b). The methodology considers emissions of CO_2 (including carbon stock changes), CH_4, and N_2O. The methodology fulfills the requirements for the VCS Wetland Restoration and Conservation and Afforestation, Reforestation and Revegetation project categories (Verified Carbon Standard 2013). The methodology covers the variety of restoration practices that may be used to restore degraded tidal wetland systems (Perillo et al. 2009). Restoration activities must have a net greenhouse gas benefit and fall under some combination of the following practices: creating, restoring, and/or managing hydrological conditions, altering sediment supply, changing salinity characteristics, improving water quality, (re-)introducing native plant communities, and improving management practices.

Greenhouse gas emissions are estimated for both a baseline scenario and a project scenario; accounting is then done by subtraction. The baseline scenario is a prediction of conditions that would most likely have occurred in the absence of the project, often referred to as the "business-as-usual" scenario. The project scenario is the conditions that actually occur with the project and is verified over time. Emissions may be either estimated or set to a conservative value. For the baseline scenario, carbon credits may be generated by estimating positive greenhouse gas emissions (e.g., increased CH_4 emissions); therefore, it is conservative to set the values of such emissions to zero. Similarly in the project scenario, credits may be generated by estimating negative greenhouse gas emissions (e.g., soil carbon sequestration); therefore, these values may be conservatively set to zero. Spatial stratification of the project area may be done to improve the accuracy and precision of greenhouse gas estimates; accounting is done separately for each stratum in the project area.

Accounting methods for each greenhouse gas include default values, emission factors, published values, models, proxies, and field-collected data (Tables 1 and 2). Default values were derived from literature sources specifically for use in the methodology; the methodology includes instructions for their applicability (note that in the methodology these are called "default factors"). Emission factors are values derived from the literature and published by organizations such as the IPCC (Intergovernmental Panel on Climate Change). They may only be used under specified conditions, and their use must be justified as appropriate to ecosystem type and conditions and the geographic region of the project area. IPCC tier 1 emission factors function as the lowest level of complexity and detail; their use is allowed in some cases but must be justified as appropriate for project conditions.

Table 1 List of abbreviations

Abbreviation	Term
AFOLU	Agriculture, forestry, and other land use
CO_2eq	CO_2 equivalents derived using global warming potential
GHG	Greenhouse gas
GWP	Global warming potential
IPCC	Intergovernmental Panel on Climate Change
NEP	National Estuary Program
NOAA	National Oceanic and Atmospheric Administration
RAE	Restore Americas Estuaries
VCS	Verified Carbon Standard
WRC	Wetlands Restoration and Conservation

Units of CO_2eq (CO_2 equivalents) expressed in this paper were derived using global warming potential (GWP) values for CH_4 and N_2O of 34 and 298, respectively, as used in the 2013 IPCC Fifth Assessment Report (Myhre et al. 2013).

Scientific Components of the Methodology

Soil Carbon Sequestration Default Value

The establishment of scientifically credible default values for greenhouse gas fluxes is an essential means to increase the feasibility of carbon project implementation. We derived a default value for soil carbon sequestration in marsh and mangrove systems of 1.46 t C ha^{-1} $year^{-1}$, which is the median

Table 2 Summary of key science components of the Verified Carbon Standard Methodology for Tidal Wetland and Seagrass Restoration

Component	Accounting method options	Default value (if applicable)	Rationale
Soil carbon sequestration	1) Proxies 2) Field-collected data 3) Published values 4) Default values or emission factors 5) Models	1.46 t C ha^{-1} $year^{-1}$ (Marshes and mangroves only)	Default value is median soil carbon sequestration value for marshes and mangroves from Chmura et al. (2003). Use of emission factors must be justified and is not permitted for marsh or mangrove systems.
Mineral-protected allochthonous carbon	1) Proxies 2) Field-collected data 3) Published values 4) Default values 5) Models	1.5% C mineral-protected carbon in allochthonous deposited sediment	Default value is the mean refractory organic carbon derived from data presented in Mayer (1994). This value may be combined with field data on percent soil carbon or organic matter to derive an estimate of mineral-protected allochthonous carbon.
Soil carbon fate following erosion	Conservatively set to 0% oxidation (return to atmosphere) in baseline scenario and 100% oxidation in project scenario		Projects may justify a greater percent oxidation in the baseline scenario or a lower percent oxidation in the project scenario. An improved approach is available in a complementary methodology in which default values are derived based on the hydrologic and geomorphic setting of the tidal wetland system through its influence on integrated oxygen exposure time (Needelman et al. 2018).
Avoided losses in organic and mineral soils	1) Proxies 2) Field-collected data 3) Published values 4) Default values 5) Models		Organic soils: Oxidation rates estimated based on changes in soil volume and bulk density. Mineral soils: Field-collected data option using historical or chronosequence data. Drainage must have occurred < 20 years (Mann 1986; Davidson and Ackerman 1993).
Methane emissions	1) Proxies 2) Field-collected data 3) Published values 4) Default values or emission factors 5) Models	Salinity >18 ppt: 0.011 t CH_4 ha^{-1} $year^{-1}$ Salinity > 20 ppt: 0.0056 t CH_4 ha^{-1} $year^{-1}$	Default values are mean emission values from data presented by Poffenbarger et al. (2011). Use of emission factors must be justified and is not permitted for tidal wetland systems.
Nitrous oxide emissions	1) Proxies 2) Field-collected data 3 Published values 4) Default values or emission factors 5) Models	Values range from 0.000157 t N2O ha^{-1} $year^{-1}$ to 0.000864 t N2O ha^{-1} $year^{-1}$ based on salinity and land use.	Default values are estimates presented by Smith et al. (1983). Use of emission factors must be justified.
Prescribed fire	1) Field-collected data on aboveground biomass	N_2O and CH_4 emission factors for vegetation burning	If using default value for soil carbon sequestration, must demonstrate that prescribed fire does not decrease carbon sequestration rates.

Source: Emmer et al. (2015a)

value from the review published by Chmura et al. (2003). We chose the median as a more conservative value than the mean of this lognormally distributed data set (mean = 2.1 t C ha^{-1} year^{-1}). Our value falls within the range of the emission factors used in the IPCC Wetlands Supplement (IPCC 2014) of 1.62 t C ha^{-1} year^{-1} for mangroves and 0.91 t C ha^{-1} year^{-1} for marshes, which were derived using a geometric mean. Unlike the IPCC, we used the same default value for marsh and mangrove systems because Chmura et al. (2003) did not find a statistically significant difference in soil carbon sequestration rates between these systems. We did not derive a default value for soil carbon sequestration in seagrass systems due to data sparsity. However, we did allow for projects to justify the use of external emission factors, including the IPCC emission factor for seagrasses of 0.43 t C ha^{-1} year^{-1} (IPCC 2014), which is below the range of seagrass carbon burial rates given by Mcleod et al. (2011)—0.45–1.9 t C ha^{-1} year^{-1}—based on their survey of multiple species at globally distributed sites. Soil carbon sequestration rates are generally determined using data from natural wetlands; there is insufficient research to determine whether median soil carbon sequestration rates consistently differ in restored systems (Howard et al. 2014).

Mineral-Protected Allochthonous Carbon

Allochthonous carbon is carbon that was removed from the atmosphere outside of the project area and transported into the project area. Carbon accounting principles require that a project only receive credit for the sequestration of allochthonous carbon in the project scenario if it would have been returned to the atmosphere in the baseline scenario (Verified Carbon Standard 2013).

A large fraction of organic matter inputs into wetland soils are labile and will be oxidized under either the baseline or project scenario (Middelburg et al. 1997). However, we recognized that there is a small fraction of mineral-protected allochthonous carbon that is relatively stable on project timescales (100 years) because it is protected by association with fine-grained mineral sediments (Mayer 1994, 1999; Hedges and Keil 1995; Derenne and Largeau 2001; Schmidt et al. 2011; Blair and Aller 2012; Renjith et al. 2012; Lehmann and Kleber 2015). (Note: the methodology refers to this fraction as "recalcitrant allochthonous carbon"). In the project scenario, deducting allochthonous carbon from carbon sequestration estimates reduces the amount of carbon credits, while in the baseline scenario, such a deduction would increase carbon credits. Therefore, projects have the option to estimate allochthonous carbon in both the baseline and project scenarios, but it is conservative to set the value to zero in the baseline scenario.

We developed a new method to estimate mineral-protected allochthonous carbon from the amount of mineral material accumulated in the soil and the percentage of carbon that accompanied the mineral matter at the time of deposition. This focus on mineral-associated carbon reflects the importance of physical protection in organic matter recalcitrance. First, the percentage of mineral mass in the wetland soil is estimated by subtraction from the percentage of total soil organic matter, which is derived from field-collected data. The percentage mineral-protected organic matter is then estimated by multiplying the percent mineral mass by an estimate of the percent mineral-protected organic matter that accompanied the mineral mass upon deposition (see below). The percentage of mineral-protected allochthonous organic matter is then converted to percent carbon using literature-derived relationships for mangrove, marsh, and seagrass systems (Allen et al. 1974; Craft et al. 1991; Fourqurean et al. 2012 as summarized in Howard et al. 2014). The deduction that must be taken for allochthonous carbon is expressed as a percentage of the total carbon sequestration rate. The resulting relationship between soil carbon content and the allochthonous carbon deduction is a power function (Fig. 1).

The percent mineral-protected organic matter that accompanied mineral deposition can be estimated from literature-derived or field-collected data or by using a default value. Mineral-protected organic matter is protected against decomposition by a variety of mechanisms including surface adsorption and occlusion in microaggregates (Blair and Aller 2012). We did not try to distinguish between these fractions but instead allowed projects to use a default value of 1.5% recalcitrant organic carbon, which is the mean refractory organic carbon derived from data presented in Mayer (1994). Note that the version 1.0 of the methodology contains a default value based on an error in a source document; this will be updated to the 1.5% carbon value.

We did not require the estimation of allochthonous carbon for organogenic wetland systems (systems that meet the

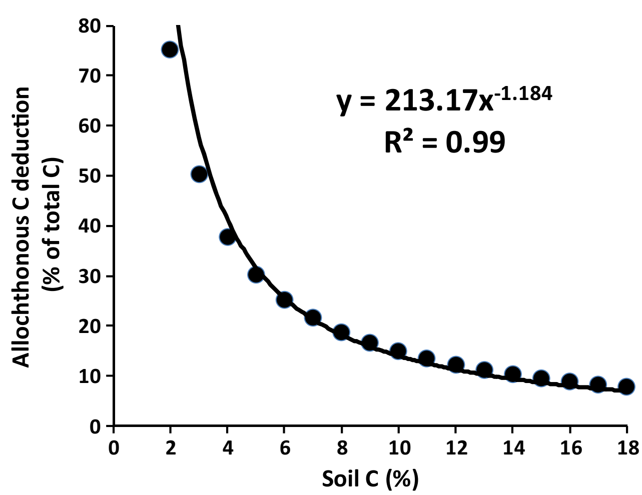

Fig. 1 Generalized estimate of the carbon credit deduction to account for allochthonous carbon for a marsh project with surface mineral soils

definition of an organic soil) because of research based on carbon isotope ($\delta^{13}C$) depletion rates that demonstrates that minerogenic tidal wetlands are strongly influenced by allochthonous material while organogenic tidal wetlands are dominated by autochthonous carbon (Haines 1976; Middelburg et al. 1997; Bouillon et al. 2003; Kennedy et al. 2010).

Soil Carbon Fate Following Erosion

Coastal wetland soil carbon pools are vulnerable to enhanced rates of oxidation to CO_2 when disturbed and transported through erosion. In the methodology, we used the conservative assumptions that 0% of eroded soil carbon is returned to the atmosphere in the baseline scenario while 100% is returned in the project scenario. A project may provide site-specific scientific justification for higher or lower emissions rates. One justification option is to apply the approach we have proposed in a complementary wetland conservation and restoration methodology currently under validation by the VCS (Needelman et al. 2018). We derived default values for carbon oxidation based on the hydrologic and geomorphic setting of the tidal wetland system through its influence on integrated oxygen exposure time (Blair and Aller 2012). In this approach, the rates of soil carbon oxidation are dependent primarily on exposure to aerobic conditions. Such exposure is dependent on transport and therefore may be limited in systems with limited hydrologic connectivity between the site and a river-estuary system. Carbon present in transported sediment may be exposed to aerobic conditions under continual resuspension through wave energy within an aerobic water column. Sediment can also have aerobic exposure if deposited into subaquatic environments with low organic carbon content and course-grained sediments with aerobic conditions in the upper portion of the soil profile. Exposure to aerobic conditions is limited in generally low-oxygen environments or when sediments are buried rapidly.

Avoided Losses in Organic and Mineral Soils

Avoided losses refers to projects that curtail or avoid greenhouse gas emissions that would have occurred under the baseline (without restoration) conditions. The avoidance of CO_2 emissions through soil organic matter oxidation due to drainage or excavation in the baseline scenario can be the largest greenhouse gas benefit of some restoration projects (Needelman et al. 2018). The potential for avoided losses is greatest for organic soils due to their large carbon stocks and rapid oxidation rates upon drainage. For organic soils, soil organic matter oxidation rates are estimated based on decreases in soil volume (estimated from soil depth) either with or without a corresponding increase in soil bulk density (Armentano and Menges 1986; Ballhorn et al. 2009; Drexler et al. 2009). Mineral soils may also have substantial carbon losses under drained conditions; however, the estimation of organic matter oxidation in mineral soils is challenging because it is weakly correlated with soil volume changes and therefore must be estimated using other methods. This rate may be estimated using historical data collected from the project area or chronosequence data collected at similar sites in addition to the other standard quantification methods of published values, modeling, proxies, and field-collected data. Published data are not available on the dynamics of organic matter oxidation in mineral tidal wetland soils following drainage or excavation; therefore, we relied on the extensive data available on the cultivation of high organic matter terrestrial grassland soils, which is often accompanied by land drainage. Based on these data, we required projects to account for the general tendency for organic carbon concentrations in mineral soils to exhibit a non-linear rate of decrease, including the tendency to approach steady-state equilibrium (Jenny 1941; David et al. 2009). We also restricted avoided losses projects on minerals soils to those that had been drained for < 20 years (Mann 1986; Davidson and Ackerman 1993).

Methane Emissions

Methane emissions have been found to be consistently very low in tidal marshes with salinities greater than 18 ppt (Poffenbarger et al. 2011). These emissions are generally considered “negligible”; however, when analyzing the > 18 ppt salinity data used in the Poffenbarger et al. (2011) study, we found that the mean emissions were 0.011 Mg CH_4 ha^{-1} year^{-1} (0.374 Mg CO_2eq ha^{-1} year^{-1}), which is 7% of the median soil carbon sequestration of 5.35 Mg CO_2eq ha^{-1} year^{-1} observed in marshes and mangroves in the review by Chmura et al. (2003). To be considered negligible (*de minimis*) by the VCS, all the non-accounted emissions must be less than 5% of the total greenhouse gas benefit—therefore, CH_4 emissions from polyhaline systems are not negligible. We allowed projects to use this mean CH_4 emission rate as a default value for systems with a salinity average or low point > 18 ppt (polyhaline systems); they have the option to use other accounting methods if they want to claim lower CH_4 emission rates. We used 18 ppt as a threshold because it matched a break in the trend of the methane-salinity relationship and because it is already an established threshold between polyhaline (> 18 ppt) and mesohaline (5–18 ppt) systems in wetland classification (Cowardin et al. 1979). However, there is one outlying point among the > 18 ppt salinity data in the Poffenbarger et al. (2011) study, which has a salinity of 18.1 ppt and a CH_4 emission rate of 0.057 Mg CH_4 ha^{-1} year^{-1}; therefore, we included a second default value for sites with a salinity average or low point > 20 of 0.0056 Mg CH_4 ha^{-1} year^{-1} (0.19 Mg CO_2eq ha^{-1} year^{-1}). We did not include CH_4 emission default values for tidal wetlands with salinities below

18 ppt due to the high standard deviations observed in these data (0.76, 2.21, and 0.11 Mg CH_4 ha^{-1} $year^{-1}$ for fresh, oligohaline, and mesohaline systems, respectively). This lack of a default value represents a significant limitation of the application of the methodology in freshwater and brackish systems; more expensive CH_4 quantification methods such as field-data collection will be required until validated models and proxies are developed (Bridgham et al. 2013). We also did not allow CH_4 emissions to be estimated from salinity values using the equation published by Poffenbarger et al. (2011) due to the overall weak fit of the data to this curve ($r^2 = 0.52$).

Our default values differ from the tier 1 emission factors supplied in the IPCC 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands (IPCC 2014). We did not use the IPCC emission factor of zero for polyhaline systems (salinity > 18 ppt) because it does not follow the *de minimis* guidelines of the VCS. We did not use the IPCC emission factor of 0.19 Mg CH_4 ha^{-1} $year^{-1}$ (6.6 Mg CO_2eq ha^{-1} $year^{-1}$) for freshwater and brackish waters (salinity < 18 ppt) because of the high variability that has been observed in CH_4 emissions from these systems (Poffenbarger et al. 2011).

VCS guidelines specify the use of GWP to convert units of CH_4 emissions to CO_2 equivalents using a 100-year time scale. Neubauer and Megonigal (2015) reported that applying the *sustained global warming potential*, which accounts for the persistent release of greenhouse gases over time, would be more appropriate for the estimation of the effect of wetland restoration on radiative forcing. Sustained global warming potential are substantially greater than GWP values (by up to ~40%), such that use of GWP values by the VCS and other greenhouse gas accounting systems may systematically underrepresent the radiative forcing of CH_4 and N_2O emissions.

The data in the Poffenbarger et al. (2011) paper were derived from a diversity of tidal marsh systems; similar integrative analyses have not been performed for mangroves or seagrass systems, although site-specific analyses have found a similar strong relationship between salinity and CH_4 emissions in mangrove systems (Purvaja and Ramesh 2001). We allowed projects to use the default CH_4 emission values derived from polyhaline marsh systems in the Poffenbarger et al. (2011) study for polyhaline mangrove and seagrass systems because the biogeochemistry of CH_4 emissions is thought to be controlled by high sulfate availability rather than vegetation type (Kristensen et al. 2008). In polyhaline wetlands, all of the conditions are present for CH_4 production including organic matter availability and highly reduced anaerobic soils. High levels of sulfate allow sulfate-reducing bacteria to outcompete methanogens for electron-donating carbon substrates (Megonigal et al. 2004). Thus, we used salinity as a proxy for sulfate. Vegetation can influence CH_4 emissions through mechanisms such as substrate availability and plant-influenced CH_4 transport mechanisms. There are few data available on CH_4 emissions from seagrass systems. The use of the default value derived from marsh systems was deemed conservative for polyhaline seagrass systems because these systems are likely to emit less CH_4 than marshes due to lower organic inputs and the CH_4 oxidation potential of the overlying water column (Oremland 1975; Deborde et al. 2010). The methodology could be improved through the addition of separate default values for mangrove and seagrass polyhaline systems if the emission rates prove to differ substantially between systems at this salinity.

Anthropogenic nutrient and soluble organic matter loading may increase CH_4 emissions in mangrove systems (Kristensen et al. 2008); however, limited data are available in polyhaline systems. Purvaja and Ramesh (2001) did observe CH_4 emissions of approximately 1.8 Mg CH_4 ha^{-1} $year^{-1}$ in a polyhaline mangrove system receiving high loads of domestic sewage. If additional research is able to quantify and define this anthropogenic influence, it may be necessary to add a restriction to the VCS methodology to prevent the use of the default value for polyhaline systems subject to sufficient anthropogenic influence to substantially increase CH_4 emissions.

Ponded areas may act as CH_4 hotspots in tidal wetlands if they do not have sufficient tidal exchange to replenish sulfate; this may make them function like a lower salinity system with a concomitant increase in CH_4 emissions. For this reason, areas of ponds, ditches, or similar bodies of water within the project area that do not have surface tidal water connectivity were required to be treated as separate strata.

Nitrous Oxide Emissions

Nitrous oxide emissions from tidal wetlands tend to decrease with shallower water tables due to lower oxygen availability, which favors complete denitrification (reduction of NO_3^- and N_2O to N_2). For this reason, we only required projects to account for N_2O emissions if water tables are lowered, such as impoundment breaching projects or wetland creation projects in which the initial (baseline) condition is open water.

In addition to proxy-based, field-collected, published, and modeled data, the methodology includes default values for N_2O emissions. These values are the mean values measured over 3 years at three marsh sites (fresh, brackish, and salt) and adjacent open water areas in the Barataria Basin in the Gulf of Mexico (Smith et al. 1983), which indicated higher N_2O emissions from wetlands than adjacent open water. There are limited data on the difference in N_2O emission rates between wetland and open water systems, yet this comparison is critical in order to represent a change from baseline (open water) to project (wetland) conditions. The Smith et al. (1983) data were the only published source that compares tidal wetland and adjacent open water systems. Despite this lack of data, we allowed these values to be applied to other wetland systems because of our judgment that the underlying mechanisms are broadly conserved across systems. The underlying biogeochemical principle is that N_2O emissions are primarily driven by denitrification operating at the aerobic-anaerobic interface. Some oxygen limitation is needed to allow for denitrifying microbial communities to compete with aerobic organisms for available organic matter; however, if oxygen availability is too low, then denitrification will lead to complete reduction of NO_3^- to N_2 gas (Firestone and Davidson 1989). Wetlands are generally not favorable for N_2O emissions because of limited oxygen availability. The reason for lower emissions from open water is that N_2O generated in sediments is subject to oxidation as it diffuses through the relatively aerobic water column.

We did not allow these default values to be used for systems that receive direct inputs of nitrogen because nitrate loading greatly increases N_2O emissions from tidal wetland systems. For example, Adams et al. (2012) found rates of N_2O emissions as high as 1.93 g N_2O m^{-2} year^{-1} in marshes restored through dike breaching (managed realignment) in a hypereutrophic estuary receiving N inputs from farmland and a sewage treatment facility. This emission rate is substantially higher than our default value for salt marshes of 0.10 g N_2O m^{-2} year^{-1} and offset as much as 49% of the carbon sequestration occurring at these study sites. Natural sites in the same estuary exhibited lower N_2O emissions, with a maximum of 0.33 g N_2O m^{-2} year^{-1}, likely due to higher water tables at the restored sites.

We did not require seagrass projects to account for N_2O emissions. Although denitrifying bacteria are typically present in seagrass bed sediments, the release of N_2O from seagrass systems is generally lower than other coastal habitat types (baseline scenarios) (Purvaja et al. 2008). Also, seagrass communities are often nitrogen-limited and therefore conservative with respect to inorganic nitrogen release (Welsh et al. 2000).

Soil Profile Sampling Methods

Direct measurement of the rates of soil organic carbon content change over time is an important means to estimate soil carbon sequestration rates. In order to implement this approach, a consistent reference plane must be established in the soil profile, below which the material may be conservatively assumed to have zero change due to project activities. The change in soil organic carbon content above this plane is then used to determine the annual rate of carbon sequestration based on the age of the reference plane (for the baseline scenario) or the start of project activities (for the project scenario). The diversity of means to establish this reference plane are allowed, including marker horizons (most commonly using feldspar) (Cahoon and Turner 1989), a strongly contrasting soil layer (such as the boundary between organic and mineral soil materials), an installed reference plane (such as the shallow marker in a surface elevation table) (Cahoon et al. 2002), a layer identified biogeochemically (such as through radionuclide, heavy metal, or biological tracers) (DeLaune et al. 1978), and a layer in which the soil organic carbon concentration is indistinguishable from the soil organic carbon concentration estimated to be present in the baseline scenario (Greiner et al. 2013; Oreska et al. 2018). This last method is included specifically because it permits identification of net carbon accumulation in depth-calibrated seagrass sediment cores relative to an average background carbon concentration profile. The latter can be obtained at time = 0 or by coring bare control sites. Subtidal sediment resuspension commonly mixes sediment in the uppermost part of a seagrass bed, preventing the use of marker horizons and surface elevation tables. This bed profile comparison approach alleviates concerns about seagrass stock-change estimation raised by Johannessen and Macdonald (2016).

Soil samples collected above the reference plane must be analyzed for soil dry bulk density and total organic carbon or organic matter. There are relationships available in the literature to estimate bulk density from organic carbon or organic matter; however, we did not allow their use because there tends to be high variability in these relationships (e.g., r^2 of 0.68 in Anisfeld et al. 1999). We did allow for the use of literature-derived relationships to estimate total organic carbon from organic matter estimates (loss-on-ignition), due to the stronger relationship between these variables, with r^2 values ranging from 0.87 to 0.99 in the studies of Craft et al. (1991) and Howard et al. (Howard et al. 2014; summarizing data from Fourqurean et al. 2012).

Sample Size

Field sampling can represent a significant project cost and a corresponding barrier to project implementation. For this reason, we provided default values and other alternatives to direct sampling whenever scientifically defensible. However, projects may still choose to collect field samples to identify greenhouse gas benefits greater than those estimated through other accounting options. Also, sampling is necessary when a default value or other methods are not available—in particular, we did not provide a default value for CH_4 emissions from fresh and brackish systems and published data and validated models for these systems are generally not yet available.

We used the Clean Development Mechanism tool entitled “Calculation of the number of sample plots for measurements within Afforestation/Reforestation Clean Development Mechanism project activities” (v. 02.1.0) as the basis for determining sample size because it is accepted best practice in the field of greenhouse gas accounting. This tool allows for a targeted confidence interval of either 90% with a 20% allowable error or 95% with a 30% allowable error. The equations in this Clean Development Mechanism tool determine sample size requirements as a power function of the coefficient of variation of the quantity being estimated (e.g., emissions) (Fig. 2). The 95% confidence interval requires fewer samples overall than does the 90% confidence interval due to the greater allowable error.

This tool should not pose a barrier to projects interested in collecting field data on soil carbon stocks or sequestration rates because of the moderate levels of spatial variation associated with soil carbon. The largest coefficient of variation present in the soil carbon sequestration studies reviewed by Chmura et al. (2003) was 0.7, with most projects having a coefficient of variation less than 0.5, which would translate to a sample requirement of about 10 per stratum (Fig. 2). Methane emissions have greater spatial variability—in a study of two tidal freshwater wetland forests, CH_4 emissions had a mean coefficient of variation of 1.0 (Megonigal 1996; Megonigal and Schlesinger 2002), which would require about 40 samples per stratum (Fig. 2). This is a substantial burden to project implementation due to the high cost of CH_4 flux sampling and the need for frequent measurements to account for temporal variation.

Prescribed Fire

The methodology allows for prescribed fire in the project scenario because this is a common tidal wetland management activity. Combustion causes a direct release of CO_2; however, research has shown that the removal of a senesced plant canopy through burning can cause increased soil temperatures and light availability, leading to increased above and belowground plant production (Bickford et al. 2012, 2015). This increased plant production may offset the loss of carbon through combustion, and may potentially even cause a net overall soil carbon stock increase (Cahoon et al. 2010; Bickford et al. 2012). Due to the limited research in this area, projects implementing prescribed burns must demonstrate that the project does not decrease carbon sequestration rates if using the default value for soil carbon sequestration.

Methane and N_2O are also produced during combustion of plant materials. There were no data available to generate emission factors for these gases from the burning of coastal wetland species; therefore, projects were instructed to use emission factors that have been determined for grassland vegetation (e.g., IPCC 2006).

Policy Components of the Methodology

Additionality

To ensure that carbon credits are truly offsetting greenhouse gas emissions, carbon markets only provide funding for projects that create greenhouse gas benefits that are "additional" beyond what would have occurred in the absence of the funding (the business-as-usual scenario). For this reason, projects must meet "additionality" requirements established by carbon standards such as the VCS. The VCS has two approaches to demonstrate additionality: a *project-level* approach in which the project provides information to the VCS and a *standardized* approach in which it is demonstrated by the methodology developer that a project type is rare—is not business-as-usual—and therefore any project of that type is deemed additional. In this section, we describe and critique the method we used to establish the standardized approach for tidal wetland restoration projects within the USA and discuss

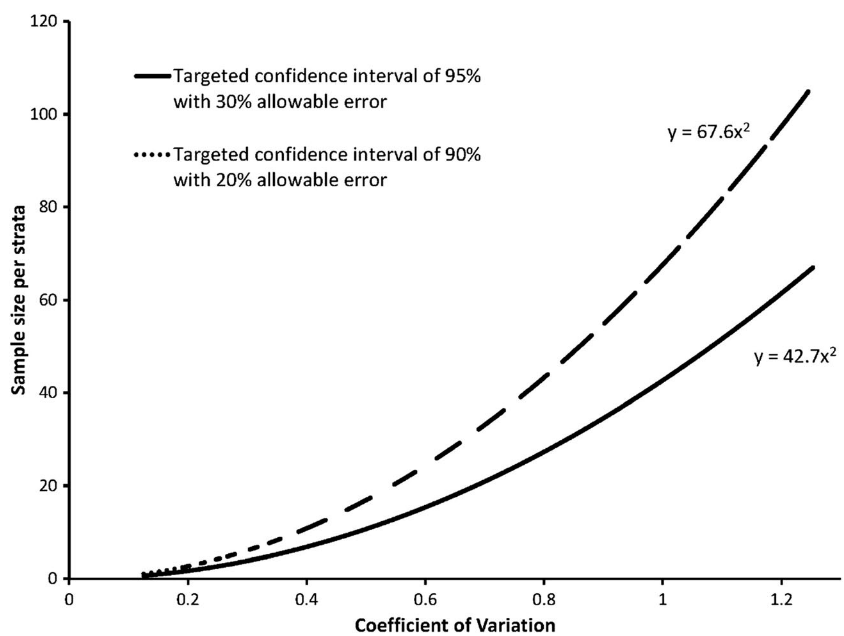

Fig. 2 Relationship between coefficient of variation and sample size requirement using the CDM tool "Calculation of the number of sample plots for measurements within A/R CDM project activities"

how a standardized approach could be developed for projects outside the USA.

To demonstrate that a project type is sufficiently rare to qualify for the standardized approach, the VCS requires that the *activity penetration* of this project type be less than 5%. For the tidal wetland restoration methodology, activity penetration is the percentage of tidal wetland restoration that has occurred relative to how much could occur. We estimated the activity penetration in the USA for tidal wetland restoration as 2.73% and seagrass restoration as 0.2% (Emmer et al. 2015a). Activity penetration was calculated separately for tidal wetland restoration and seagrass restoration due to a lack of a single data set containing information about both activities.

Activity penetration is calculated as follows:

$$APy = 100 \times OAy/MAPy$$

where *APy* is activity penetration of the project activity in year y (percentage), OAy is observed adoption of the project activity in year y (area), and *MAPy* is maximum adoption potential of the project activity in year y (area).

For observed adoption (*OAy*) of tidal wetland restoration, we reviewed data reported by the 28 National Estuary Programs (NEPs) and their partners and determined the average annual acreage of tidal wetland restoration. The NEP data set was the most comprehensive, available national data set of restoration activities in US tidal wetlands. The NEPs have been reporting data since 2000; therefore, the year 2000 was used as the starting point for determining total area restored. Since 2000, national attention and resources focused on coastal restoration have increased significantly. The four most recent years of data, 2009 through 2012, were reviewed. Analysis involved reviewing every reported project from each of the 28 NEPs; only projects meeting the methodology's definition of tidal wetland restoration were counted. An annual average from these 4 years was calculated and applied to the remaining years since 2000 in order to arrive at the total observed adoption level for the NEPs. To further extrapolate from the NEP data to a national estimate for area restored, we compared the total land area of the 28 NEPs to the total land area of US coastal counties and scaled accordingly. Land area of coastal counties was obtained through US Census Bureau and National Oceanic and Atmospheric Administration (NOAA) data. We also assumed that the observed adoption of tidal wetland restoration in areas represented by an NEP is likely to be at least twice as great as restoration levels in areas not represented by an NEP because of the strong collaborative management approaches and consistent state and federal funding present in NEPs.

For seagrass beds, observed adoption (*OAy*) was estimated from the percentage of total restoration reported in the Restoration Atlas maintained by NOAA (National Oceanic and Atmospheric Administration 2015). As the lead federal agency mandated with coastal and marine fisheries habitat restoration and protection, including seagrass meadow habitat, NOAA's level of funding for seagrass meadow restoration was adopted in the methodology as representative of the overall level of seagrass restoration. The Restoration Atlas contains information on approximately 2700 habitat projects that have occurred since 2000, of which 120 (4%) were seagrass meadow projects. Seagrass restoration projects tend to be smaller than other types of restoration, and a conservative estimate for seagrass meadow restoration activities is 4% of the total acreage in the NOAA database.

The next step was to determine the maximum adoption potential for both tidal wetland and seagrass habitat restoration. *Maximum adoption potential* is an estimate of the highest possible level of a given project activity. For tidal wetlands, *MAPy* was estimated as the area of former tidal wetland that have converted to open water or other land uses. We used a conservative estimate of areas lost to open water nationwide as the area of wetlands lost to open water in coastal Louisiana. For areas converted to other land uses, we estimated that 33% of the 100-year coastal floodplain as reported by the Federal Emergency Management Agency (FEMA) could be reconverted to tidal wetland (FEMA 1991). Because the FEMA estimate was made in 1991, we added to the *MAPy* the coastal wetland losses reported in the Status and Trends of Wetlands in the Conterminous United States covering 1992 to 2013 (Dahl 2000, 2006, 2011). Activity penetration for tidal wetlands was then calculated to be 2.71%, which is below the 5% threshold for the VCS standardized approach. For seagrass meadows, the maximum adoption potential was estimated to be the areas reported lost between 1937 and 2006 (Waycott et al. 2009). Activity penetration for seagrass meadows was calculated to be 0.2%.

Through the use of the best available data and numerous assumptions, our estimates demonstrate that both tidal wetland and seagrass restoration in the USA are not business-as-usual. However, our analysis of additionality in the USA points strongly to the need for more consistent and representative data sets for tidal wetland and seagrass habitat losses, as well as restoration activities. This would reduce the number of assumptions needed to calculate restoration levels and result in a more accurate estimate.

In order to demonstrate additionality utilizing the activity method for tidal wetland and seagrass restoration *outside* of the USA, similar data sets would need to be identified and analyzed. Alternatively, in the absence of such data, if one could demonstrate that the USA is likely to have the highest level of restoration in the world, through an analysis of the factors that affect restoration activity levels such as policies and funding, as well as past degradation of tidal wetland areas, then it could be assumed that any other country or region will have a lower level of activity penetration than the USA. This may be sufficient to demonstrate to the VCS that the penetration level for tidal wetland and seagrass restoration worldwide satisfies the 5% threshold.

Leakage

Leakage occurs when a project leads to an increase in greenhouse gas emissions or decrease in greenhouse gas removals outside of the project area. Leakage may occur by shifting an activity from the project site to some other location, referred to as *activity shifting*; when a project reduces the local supply of a product increasing production elsewhere, referred to as *market leakage* (Aukland et al. 2003); or by causing an increase in emissions in an ecosystem outside of the project boundary that is hydrologically connected to the project area, referred to as *ecological leakage* (Verified Carbon Standard 2013). A greenhouse gas accounting methodology has two general ways to account for leakage: (a) tracking leakage emissions in the project scenario and (b) avoiding leakage altogether by setting strict applicability conditions including careful establishment of project boundaries. We only allowed for the second option in the methodology because tracking leakage is a significant burden on project developers that may sometimes render the project unfeasible (e.g., when leakage emissions overwhelm project emission reductions or monitoring costs are high).

In order to avoid activity shifting and market leakage, the methodology sets applicability conditions that either (a) demonstrate that prior to the start of the project the land is free of a land use that could be displaced outside the project area, (b) require that a land use that could be displaced outside the project area (e.g., timber harvesting) is not accounted for in the baseline scenario, or (c) require a pre-project land use that will continue at a similar level of service or production during the project crediting period (e.g., reed or hay harvesting, collection of fuelwood, subsistence harvesting). The focus is on avoiding situations where shifted activities cause the drainage or other forms of wetland degradation, which may result in substantial greenhouse gas emissions. However, there may be restoration projects with net greenhouse gas benefits that are associated with activity shifting leakage that is not necessarily caused by the project. A research need is to develop criteria and methods to allow projects to justify that such activity shifting would have also occurred in the baseline scenario.

Ecological leakage is avoided in the methodology by setting applicability conditions regarding project design ensuring that hydrological connectivity with adjacent areas does not lead to a significant increase in net greenhouse gas emissions outside the project area. The ecological leakage procedure was adapted from non-tidal systems where construction of a permeable dam can prevent changes in water levels outside the project boundary. However, in tidal systems, such dam construction is generally not feasible. This issue underscores the importance of establishing a project boundary wide enough to capture expected water level changes that are linked to project activities. The methodology contains the following criteria designed to avoid certain hydrologic alterations outside of the project areas: must maintain wetland conditions to prevent soil carbon oxidation resulting from the lowering of the water table; may not convert open water to non-seagrass wetlands, which prevents a lowering of the relative water table that may lead to increased N_2O emissions; may not convert non-wetland to wetland conditions, which prevents the raising of the water table that may cause increased CH_4 emissions; and may not convert vegetated to poorly vegetated conditions, which prevents decreased plant productivity that may result in lower rates of carbon sequestration.

Discussion and Conclusions

The VCS Methodology for Tidal Wetland and Seagrass Restoration was designed to allow the diversity of tidal wetland restoration projects to receive VCS-approved carbon credits. We attempted to develop a methodology that is feasible to implement and highly flexible, while maintaining scientific rigor. The science and policy of greenhouse gas emissions and carbon storage in tidal wetlands are evolving—we included several innovative approaches in our methodology, yet it remains limited by knowledge gaps (Table 3). We established default carbon sequestration values for marsh and mangrove systems but not for seagrass systems due to data sparsity. We did allow projects to justify the use of the IPCC emission factor for seagrasses; however, this value is derived from only six data points (IPCC 2014)—additional data are needed to increase the confidence of estimates in these systems. We developed a methodology for estimating mineral-protected allochthonous carbon in tidal wetland systems using field-collected data on soil carbon or organic matter and literature-derived default values of the recalcitrant carbon in aerobic marine sediments. The validation of this method and refinement of our understanding and ability to identify recalcitrant organic matter in tidal wetland and marine systems are important research areas that may lead to improvements of our methodology.

The avoided losses of carbon through soil organic matter oxidation have great potential for greenhouse gas benefits but are difficult to estimate for mineral soils. Additional research is needed in this area to complement the understanding of carbon losses following cultivation of high-organic matter terrestrial soils to provide methods to estimate these losses in minerogenic tidal wetlands.

We found that CH_4 emissions from polyhaline soils are not negligible according to VCS guidelines; therefore, we provided default values for these systems. We did not derive CH_4 emission default values for fresh and brackish tidal wetlands nor did we allow for the use of IPCC emission factors due to the high variability of emissions in these systems

Table 3 Summary of key science and policy research needs related to the Verified Carbon Standard Methodology for Tidal Wetland and Seagrass Restoration (Emmer et al. 2015a)

Component	Key science and policy research needs
Soil carbon sequestration	• Develop default values for seagrass systems. • Determine whether average rates of soil carbon sequestration differ between natural and restored systems.
Mineral-protected allochthonous carbon	• Validate new method to estimate mineral-protected allochthonous carbon.
Soil carbon fate following erosion	• Develop and validate scientific justifications for oxidation rates less than 100% in the baseline scenario.
Avoided losses in organic and mineral soils	• Improve understanding of the dynamics of organic matter oxidation in mineral tidal wetland soils following drainage or excavation.
Methane emissions	• Develop and validate cost-efficient methods to estimate methane emissions in fresh and brackish wetlands, such as modeling and proxies. • Determine how methane emissions differ between polyhaline marsh, mangrove, and seagrass systems. • Determine if anthropogenic nutrient and soluble organic matter loading increases methane emissions in polyhaline systems. • Develop methods to estimate methane emissions from potential methane hotspots such as ponds, ditches or similar bodies of water that do not have surface tidal water connectivity. • Collect data on methane emissions from seagrass systems.
Nitrous oxide emissions	• Research differences in nitrous oxide emission rates between wetland and open water systems. • Develop methods to estimate nitrous oxide emissions from systems receiving direct inputs of nitrogen. • Collect data on nitrous oxide emissions from seagrass systems.
Prescribed fire	• Improve understanding of conditions in which prescribed fire decreases soil carbon sequestration rates. • Develop emission factors for methane and nitrous oxide resulting from the combustion of aboveground wetland vegetation.
Additionality	• Determine if tidal wetland restoration projects outside of the USA qualify for the standardized approach to demonstrate additionality.
Leakage	• Develop criteria and methods to allow projects to identify activity shifting leakage outside of the project area that is not caused by the project.

(Poffenbarger et al. 2011; IPCC 2014). Field sampling costs are likely to be prohibitively high due to the high spatial and temporal variability of CH_4 fluxes; therefore, there is a pressing need for validated models or proxies to estimate CH_4 emissions from fresh and brackish systems. We did provide default values for CH_4 emissions from polyhaline tidal wetland systems; these values were derived from an analysis of tidal marsh data—additional data are needed to validate these values or develop different values for other tidal wetland systems.

We provided default values for N_2O emissions for marsh and mangrove systems in the absence of direct nutrient inputs, but these values were taken from a single study. Additional data are needed comparing N_2O emissions from open water (baseline scenarios) versus tidal wetland systems (project scenario) to validate or improve these default values.

We allowed for the use of literature-derived relationships to estimate soil carbon percentage from soil organic matter (loss on ignition) data. The functions we used are based on strong relationships but are based on spatially limited data sets—broader analyses may help strengthen these estimates and identify systems that deviate from established relationships.

Procedures are included in the methodology to estimate greenhouse gas emissions from prescribed fire, but projects using the default value for carbon sequestration are required to demonstrate that prescribed fire does not cause a net loss of soil carbon. Additional research is needed to broadly establish conditions in which plant stimulation caused by prescribed fire is sufficient to offset carbon losses. Tidal wetland-specific research is also needed on CH_4 and N_2O emissions occurring during combustion of aboveground plant materials.

We compared potential and actual tidal wetland restoration to demonstrate additionality for all tidal wetland restoration projects in the USA. A significant policy need is to conduct a similar analysis globally or by region; until this analysis is performed, projects outside of the USA will be constrained by the need to demonstrate project-specific additionality.

The methodology avoids leakage by setting applicability conditions and guidelines on establishing project boundaries. We did not provide procedures for the tracking of leakage because this is a significant burden on project developers, but this does limit the scope of our methodology. For example, projects with a displaceable land use are not allowed; revisions to the methodology could provide guidelines for how to demonstrate the lack of activity shifting of such land uses.

The methodology is a flexible system that allows for the accounting of greenhouse gas fluxes in a diversity of tidal wetland restoration projects. It is specifically designed for projects to generate saleable VCS carbon credits, but the accounting tools in this methodology have broader applicability and may be used to complement currently available systems of national to project-level greenhouse gas accounting for tidal wetland systems.

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