Science Foundation Chapter 6
Carbon Sequestration and Greenhouse Gases in the Baylands

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INTRODUCTION

Increasingly, but slowly, we as a global society are moving towards linking our land use management, including ecosystem restoration, with activities that reduce our climate change footprint and increase our capacity to respond resiliently at the local level to the challenges of global environmental change. Thoughtful management of San Francisco Estuary's wetlands and Baylands can contribute to local ecosystem resilience as well as play a part in global climate regulation. We also have the opportunity to transfer knowledge gained here, where planning activities have greater support and capacity to be more forward-looking, to other parts of the country and the world.

Incorporating climate change mitigation benefits of wetlands management into planning frameworks is a new focal activity that has developed since the first Goals Report, yet quantification to support these activities in San Francisco Bay has yet to be developed. The management of upland forests for climate change benefits is now well established. Activities on agricultural lands, as found on our Baylands, are being examined to develop best practices to reduce greenhouse gas (GHG) emissions and improve soil carbon sequestration. Over the last 10 years these concepts gradually have moved down slope from drylands to include soil carbon conservation in wetlands, particularly peatlands, and most recently coastal wetlands. The term "blue carbon" has permeated the policy vernacular, reflecting the potential for improved management of coastal wetlands to contribute to climate change mitigation activities, while supporting environmental quality and climate change adaptation.

The management of tidal wetlands is more complicated than forests management, and their areal extent is substantially less; however, the high ecosystem value of coastal wetlands, the greater climate change impact per unit area and the fact that coastal wetlands are being lost a rate greater than any other environment globally and nationally places a particularly high importance on conserving these ecosystems and on finding ways of maintaining them as sea levels rise. Focusing on climate mitigation, activities that reduce or reverse emissions of greenhouse gases, we must factor in impacts of wetlands management on carbon sequestration, methane emissions and to a lesser extent nitrous oxide emissions. In this chapter, we identify key considerations for improving understanding of these processes, considering the extent of historic and current carbon dynamics, as well as possibilities for future processes.

Key questions we address in this chapter are:

- How have carbon and GHG dynamics within the Bay changed over time, and how might they change into the future?
- What role do the Bay's wetlands and Baylands play in greenhouse gas cycling?
- What additional research and data would improve our understanding of carbon and GHG dynamics, and how can this insight improve management of tidal wetlands in the future?
- What are the links between carbon dynamics and other management concerns, e.g., subsidence reversal, sediment management, etc.)?

BACKGROUND

Evolution of a Restoration Policy

The Bay Area has developed very effective policy and implementation for the conservation and restoration of coastal wetland ecosystems. Early tidal wetland restoration projects were small with isolated impacts, but through progressive experimentation and sharing of experience and knowledge, we have developed projects with rich environmental benefits. Projects have become larger and more complex, incorporated into a coastal mosaic that includes urban areas and agricultural lands. For example, the recent restoration of the Hamilton Wetland on the former Army Airfield beneficially used over six million cubic yards of dredge material from regional maintenance dredging projects and deepening of the Port of Oakland. We also have become one of the leading regions nationally and globally to incorporate sea-level rise projections into coastal wetlands restoration planning. This planning recognizes the need for strategic realignment of coastal infrastructure and sediment management, the importance of restoring early but also the value of projects of different successional ages, and an awareness that some vegetated wetlands will convert to mudflat and open water with sea-level rise, while others in sheltered locations with sufficient sediment inputs will flourish.

As conditions evolve, we are entering a new learning phase, recognizing the potential climate change mitigation (atmospheric GHG reduction) benefits of Baylands and wetlands management by accounting for the changes in carbon storage and fluxes of greenhouse gases. This new phase is at an early stage of development, guided by general principles but limited in detailed quantification. We know that drained wetlands can rapidly release carbon stores that accumulated over hundreds or thousands of years. Drained organic-rich soils may continue to release CO_2 over long periods, and prolonged emissions are evident in drained areas of the Sacramento-San Joaquin Delta, which has been emitting CO_2 continuously for over a century, perhaps as much as 1 billion tons (equivalent to 25% of the carbon in California's standing stock of forests). Conversely, emissions from more mineral-rich soils typically decline or halt over time, and wetland restoration can reinitiate the slow process of carbon sequestration once vegetation is reestablished. In addition to emissions of CO_2, some wetland soils can release nitrous oxide, N_2O (a greenhouse gas 310 more potent than CO_2) and methane, CH_4 (a greenhouse 34 times more potent than CO_2). Given these substantial greenhouse effects, both N_2O and CH_4 must be incorporated into any evaluations of overall carbon dynamics and GHG emissions. Nitrous oxide emissions are greatest in wetlands with high nitrate concentrations (e.g., those affected by high fertilizer loads); methane emissions occur in wetlands with

standing water, as well as in drainage ditches and duck ponds, and are more likely to occur at salinities below 18 ppt (~1/2 the salinity of seawater).

A number of actions are ongoing to link overall wetlands management to climate change mitigation responses, both locally and more broadly. California has established a state-level cap and trade system in order to reduce emissions. Early offset projects eligible for credits for reducing GHG emissions within the state have focused on technological solutions. However, in November 2013 the California Air Resources Board (ARB) took the major step of issuing the first eligible carbon credit for compliance offset credit for improved forest management for a coastal region near Willets. Though further behind forestry projects, management of organic soils on drained coastal wetlands and the restoration of these wetlands are being eyed as potential future offset projects. More broadly, carbon market institutions are also exploring the potential to expand their range of activities to recognize wetland management, and in particular coastal wetland management. In 2011 the Verified Carbon Standard (VCS) recognized Wetland Restoration and Conservation activities eligible as potential carbon projects, and recently first global methodology for Greenhouse Gas Accounting Methods for Tidal Wetland and Sea Grass Restoration was submitted for approval (Emmer et al. 2013). In addition, federal agencies have established an interagency team to support blue carbon efforts, and the United States annually reports the official national GHG Inventory, meeting commitments under the UN Framework Convention on Climate Change. Finally, last year the Intergovernmental P on Climate Change (IPCC) provided guidance on incorporating the human impacts to wetlands within accounting for national GHG emissions and reductions (IPCC, 2013).

Whether to support national and state climate change goals, e.g., under a carbon finance framework, or to encourage less formal good practice, there is a need for refined quantification of GHG emissions and removals due to wetlands management at the regional scale. Moreover, wetland climate change mitigation activities should be integrated with regional climate change adaptation strategies to avoid future conflicts in planning outcomes (Crooks et al. in prep).

Scientific Background

Carbon cycling through plant growth, decomposition, sequestration, and GHG emissions directly affects the sustainability of tidal wetlands. Carbon dynamics have direct links to management concerns for Bay wetlands and are affected by, as well as a feedback to climate change. Tidal wetlands are highly sensitive to sea-level rise but also maintain their balance and remove CO_2 from the atmosphere as they build soil organic matter. In this sense, carbon sequestration within tidal wetlands integrates across both adaptation and mitigation for climate change. Within the Baylands, carbon sequestration is of particular management interest because of the possibility of reversing the loss of elevation due to subsidence, as has been demonstrated on many restoration projects and on a managed subsidence reversal project at Twitchell Island in the Delta, while also providing many other ecosystem services, such as shoreline and flood protection, water quality improvement, and the development of habitat, food web support, and biodiversity.

Organic accumulation within tidal wetlands is strongly correlated with vertical accretion (Turner et al. 2000, Drexler 2011, Callaway et al. 2012, Morris et al. 2012, Schile et al. 2014) and contributes significantly to the capacity of a tidal wetland to build elevation in the face of sea-level rise. Significant stocks of carbon gradually have accumulated within Baylands soils over time, and overall rates of carbon sequestration within tidal wetlands around the world are higher than in many other terrestrial and aquatic ecosystems, particularly when factoring in the continuous burial with sea-level rise (Pendelton et al. 2012). In salt marsh and brackish wetland soils, organic matter commonly contributes up to 20% and 60 %, respectively, by weight in San Estuary (Callaway et al. 2012), with a much greater percentage by volume, given its relatively

low bulk density compared to soil mineral matter (Nyman et al. 1990). As in many tidal wetland systems, the soil carbon content (as a percentage of total soil mass) is generally greatest in low salinity tidal wetlands (freshwater wetlands > brackish wetlands > salt marshes). However, because of the inverse relationship between organic matter content and soil bulk density, the carbon density of wetland soils (g C/cm^3 of soil) is much less variable, with most wetland soils ranging from 0.02 to 0.04 g C/ cm^3 (Gosselink et al. 1984; Chmura, unpublished analysis).

In order for tidal wetlands to remain vegetated, they must accumulate enough mineral and organic sediment to keep pace with on-going rates of sea-level rise. As tidal wetlands accrete sediment, they accumulate carbon within that sediment. Slightly greater rates of sea-level rise will lead to slightly greater rates of carbon sequestration, given adequate suspended sediment concentrations (Morris et al. 2012). However, if wetlands cannot keep pace with sea-level rise and eventually lose elevation, plants become stressed, die, and conversion to unvegetated mudflats occurs, drastically reducing the ability to sequester site-produced carbon (Nyman et al. 1993).

Two critical processes related to carbon sequestration are photosynthesis (conversion of CO_2 to plant biomass) and decomposition (conversion of soil carbon back to CO_2), with the potential for additional carbon inputs and outputs as either particulate or dissolved organic carbon (POC and DOC). Emission of other GHGs (primarily CH_4 and N_2O) also can be important within tidal wetlands. Drainage of coastal wetlands results in the rapid release of CO_2 to the atmosphere through decomposition of carbon stocks that accumulated over hundreds to thousands of years (Deverel and Leighton, 2010; Crooks et al. 2011, Lovelock et al. 2011, Pendleton et al. 2012), and emissions from drained organic soils may continue for decades until the stock is depleted or management activities change (Drexler et al. 2009, Deverel and Leighton 2010). Reduction of these emissions through management changes and restoration of wetlands is a contributory action to mitigate climate change. There is further potential for reduced vulnerability to climate change impacts, as well-functioning tidal wetlands are effective at reducing storm surge that is likely to increase with on-going climate change.

In addition to CO_2 that may be sequestered within tidal wetland soils or be emitted from degraded wetland soils, tidal wetlands can emit CH_4 as a result of sequential oxidation-reduction processes under for anaerobic conditions. As indicated above, methane is an important greenhouse gases, with a much greater greenhouse effect per molecule than CO_2. Methane is produced under highly reduced conditions in many wetlands, but due to the high concentration of sulfate (SO_4^{2-}) in waters with a salinity greater than half that of sea water, activity of methanotrophic bacteria is limited (Poffenbarger et al. 2011). Consequently, CH_4 production is likely to be suppressed in most areas of the Estuary west of Suisun Bay, bar those nedar sources of freshwater or stormwater. Nitrous oxide, by contrast, is primarily produced as a byproduct from human pollution, particularly nitrogen leaching from agricultural fields but also sources in stormwater as well as atmospheric deposition from industry. Estuary and coastal waters are significant sources for nitrous oxide as they recycle nitrogen sources from human activity. Wetlands with permanently wet soils are a minor component of this cycle compared to agricultural soils and open water (EPA, 2010). These emissions are recognized in the development of the US national inventory which applies a default value for N_2O emissions based upon upstream nitrogen application to farm fields (IPCC, 2006)

In the broad geographic scope of the San Francisco Estuary, research on wetland GHG biogeochemistry has been advanced primarily in the Delta, where the majority of former wetland acreage now exists as drained subsided organic soils. Studies have assessed soil carbon stocks (Drexler et al. 2009); rates of CO_2 emissions with land drainage (Rojstaczer and Deverel 1995); soil carbon accumulation under managed soil building, known as subsidence reversal (Miller et al. 2008); and baseline fluxes of CO_2, CH_4 and N_2O on drained wetland under a variety of land uses (Teh et al. 2011). However, Delta wetlands are outside of the

scope of the BEHGU project. Within San Francisco Bay, research has focused primarily on wetland response to sea-level rise and restoration (Orr et al. 2003, Stralberg et al. 2011), including some recognition of carbon in wetland building or Holocene response to sea-level rise (Goman 2001), but much less so on CO_2 or other GHG fluxes at annual scales. Data are available for gross soil carbon sequestration rates within natural salt and brackish tidal wetlands (Drexler et al. 2009, Drexler 2011, Callaway et al. 2012), as well as from managed freshwater wetlands (Miller et al. 2008, Miller and Fujii 2010). However, little data are available for sequestration in tidal freshwater wetlands, restored wetlands, or managed wetlands, such as the duck clubs in Suisun Bay. Basic information on soil carbon stocks is limited to analyses of wetland carbon sequestration or to soil survey data collected from drained wetland soils during the 1960s. Beyond these datasets, GHG fluxes in San Francisco Estuary's tidal and drained wetlands are poorly studied. Yet it is this information that is needed to understand the impact of land use, including wetlands restoration, on landscape level GHG fluxes and climate change mitigation. Once basic data on emissions associated with various land uses on the Baylands and for wetland management we will be in a position to more effectively connect climate change mitigation and adaptation for our Baylands system.

APPROACH AND FINDINGS

Our approach in this section is to draw together available information to improve our understanding of historic, current, and future carbon and GHG dynamics within San Francisco Estuary tidal wetlands (including drained soils in former wetland areas). As with the rest of the BEHGU, a range of future management actions were evaluated, focusing on wetland restoration of Baylands. In our analysis of carbon dynamics and GHG emissions we considered:

1) historic emissions of CO_2 by estimating the loss of carbon from historic tidal wetland soils due to the diking and drainage of wetlands;

2) on-going carbon dynamics via sequestration and emissions of CO_2 and other greenhouse gases, CH_4 and N_2O (although specific estimates of current emissions rates were not possible); and

3) impacts of restoration on future carbon sequestration and GHG emissions.

Our analyses across all three areas (historic, current, and future carbon dynamics) were restricted by available data, which were quite limited in many cases. No additional field-based data collection was undertaken, and given the shortcoming of available data, we also discuss information gaps as part of our evaluation of management issues. Details on methods are included in Appendix 6.1.

Historic Carbon Emissions from Drained Baylands

Based on the existing soil (Table 6.1 and Figure 6. 1), the overall loss of soil volume due to drainage and diking (see current land cover in Figure 6. 2), and carbon density in these soils (Figure 6. 3), we estimate that approximately 1.2 million metric tons of carbon (4.6 million tons of CO_2) were released across the region due to tidal wetland drainage over many decades (Table 6.2). It is likely that the majority of these emissions took place in the months and years immediately following drainage because a significant component of the soil carbon that is stored under anaerobic wetland conditions is subject to rapid decompositions when wetland soils are converted to aerobic conditions. However, in areas where soil organic content is very high (e.g., freshwater and low salinity peat soils) emissions may continue for extended periods because of the large pool of organic carbon (Deverel and Leighton, 2010).

We estimated the loss in elevation and soil volume for diked and drained wetlands by assuming historic elevations for tidal wetlands at MHHW (as is the case for most tidal wetlands in the Estuary presently) versus existing elevations in diked areas. The calculated volume losses associated with drainage of wetland area within each segment of the bay, accompanied by estimates of emissions, are provided in Table 6.2.

The South Bay salt pond complex and Central Bay had lower average rates of elevation loss, compared to the North Bay and Suisun Bay; however, because of large areas affected by diking in the South Bay soil loss (13 million m^3) was comparable to North Bay (20 million m^3) and Suisun Bay (13 million m^3; Table 6. 2). Total CO_2 emissions were calculated based on the lost soil volume, along with an average soil carbon density of 0.0233 g C/cm^3 for salt marshes and 0.0334 g C/cm^3 for brackish marshes (based on data from Gosselink et al. 1984, and Callaway et al. 2012), with the emissions patterns across the sub-regions closely mirroring that of overall soil volume loss (Table 7. 2). While these broad estimates of emissions, they give a sense of the scale of total CO_2 flux. By way of comparison, this cumulative emission equates roughly to less than 2 years of ongoing CO_2 flux from soils in the Sacramento San Joaquin Delta (reflecting the high organic content of those soils; Deverel and Leighton 2010).

Ongoing Soil Carbon Dynamics in the Baylands

Due to the lack of data for the depth of organic soils within tidal Baylands, it is impossible to provide quantitative estimates of carbon stocks in existing tidal wetlands within the Estuary. Data gaps for emissions rates of CO_2 and other GHGs from both natural and diked Baylands, also make quantification of current Bay-wide emissions impossible. Carbon sequestration rates within existing tidal wetlands averaged ~80 g $C/m^2/yr$ over the last century based on ^{210}Pb-dated sediment cores from six wetlands across the Estuary including four salt marshes (Whale's Tail in the South Bay; China Camp, Petaluma River Marsh, and Coon Island in the North Bay) and two brackish marshes in Suisun Bay (Rush Ranch and Browns Island; Callaway et al. 2012). There was little variation in sequestration rates within individual wetlands (based on evenly spaced low, mid, and high stations across each of the six wetlands). In addition, variation across sites was also small with only one location (Browns Island) having slightly higher average sequestration rates (107 g $C/m^2/yr$). Based on an average value of 80 g $C/m^2/yr$ and the overall area of tidal wetlands within the Bay (18,200 ha or 45,000 acres), existing wetlands are accumulating an annual total of 14,560 `` of carbon across North Bay, South Bay, and Suisun Bay. Although sequestration data are available for mature wetlands within the Estuary, no data exist on rates within recently restored wetlands. Given high rates of sediment accretion in recently restored areas, sequestration rates in these restored wetlands could be higher than natural tidal wetlands over the short term.

In order to provide estimates on existing CO_2 emissions, new soil data are needed, including more comprehensive spatial data on % organic matter, bulk density, soil temperature, depth to water table, and salinity. These data are needed for each soil series, and based on specific land use (i.e., drained vs. natural conditions of the soils). Such datasets would provide the necessary inputs for calculations to estimate rates of subsidence and carbon emissions from soils subsequent to drainage for agriculture, and would form the basis of future simplified, low cost indicators of carbon emissions associated with land use.

In addition to carbon dioxide emissions, methane likely is being released from drainage ditches and areas of standing low salinity and brackish waters on drained Baylands, and nitrous oxide on sites with cattle or subject to nitrogen fertilizer application. Estimates of these emissions could be calculated with some data collection on land use and application of default values provided by the IPCC (2006, 2014). Ideally, the development of regionally specific emissions factors derived from direct measurements would provide more refined analysis of greenhouse gas fluxes within the landscape and climate of the San Francisco Estuary.

Table 1. Acreage of each of the nine dominant soil types classified by current landcover type, region, and segment.

		Suisun			North Bay					Central Bay				South Bay							
Soil Type	**landcover type**	**A**	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**I**	**J**	**K**	**L**	**M**	**N**	**O**	**P**	**Q**	**R**	**S**	**T**
Joice	**Channel**	46.9	59.9	3.7	12.7																
	Farmed Bayland		18.4																		
	Grazed Bayland	380.2	6.1																		
	Nontidal Wetland	7316.1	4658.3	1224.3	85.8																
	Salt Pond																				
	Tidal Flat	32.0	1.6	4.1	12.9				5.4												
	Tidal Marsh	1502.6	1866.5	3949.9	353.2				17.5												
	Urban	5.0	30.6	273.7	5.5				13.9												
	other	13.5	4.8	13.4	0.4				2.8												
	TOTAL	9296.2	6646.2	5469.1	470.5				39.6												
Novato	**Channel**						12.7	4.6						3.6	3.7	69.0	1.2				
	Farmed Bayland							6.7													
	Grazed Bayland						9.9														
	Nontidal Wetland						387.9	63.0						132.4	100.5	1565.2	631.8				
	Salt Pond							0.7						9.0	2904.1	4002.5	3321.2				
	Tidal Flat						3.1	42.6			11.9			13.1	50.0	280.7	60.4				
	Tidal Marsh						845.8	1149.1			12.0			148.2	1011.8	1215.7	501.7				
	Urban						5.1	45.8			25.4			98.3	272.5	95.0	95.0				
	other						6.2	6.4						0.5	8.2	13.9					
	TOTAL						1270.6	1318.9			49.3			405.1	4350.8	7242.0	4611.3				
Omni	**Channel**			2.0																	
	Farmed Bayland																	211.6		188.3	
	Grazed Bayland			27.7																	
	Nontidal Wetland			855.5					1.7									83.7	234.5	70.5	0.5
	Salt Pond																	0.6		4.3	
	Tidal Flat											0.2									
	Tidal Marsh			434.0					1.3										7.5	15.2	7.5
	Urban			653.4					12.6			13.1						7.8	28.5	213.7	127.9
	other			33.4					0.8			8.0								0.4	0.8
	TOTAL			2006.0					16.3			21.2						303.7	270.5	492.5	136.7
Reyes	**Channel**	24.2	46.4	1.1	61.4	39.9	48.6		2.9						15.1		5.5	4.3	1.9		
	Farmed Bayland		334.8		233.6	12456.4	4612.6	3600.4										108.4		112.9	
	Grazed Bayland	506.6	28.7		659.2	616.1	2022.8	139.5													
	Nontidal Wetland	5310.7	4220.9	71.0	9439.7	3133.0	787.6	1590.3	254.0				11.6		1531.5		271.5	115.8	117.7	1177.8	867.0
	Salt Pond			2.9	1078.6	139.2	347.9	29.5	1.1								1933.8	5234.2	3425.2	4790.9	
	Tidal Flat	7.0	0.9	23.6	175.1	27.4	20.3	2.7	25.5			1.6	1.8		45.6		196.9	35.2	23.6	36.6	71.0
	Tidal Marsh	477.2	599.1	800.4	4953.2	2731.1	3608.2	204.5	499.3			40.3	30.4		1077.2		431.5	1312.4	639.3	584.5	788.4
	Urban	24.3	159.4	98.8	533.2	234.4	494.0	198.8	494.3			0.3	71.6		146.8		293.0	55.2	54.4	54.7	300.5
	other	1.3	7.9		2.1	8.0	40.1	9.3	15.5			0.4	6.2						8.2	8.6	8.5
	TOTAL	6351.3	5394.4	997.9	17136.1	19385.5	11982.1	5774.9	1298.3			42.5	121.7		2816.2		3132.2	6865.5	4270.3	6765.9	2035.5

Table 1. Acreage of each of the nine dominant soil types classified by current landcover type, region, and segment.

		Suisun			North Bay					Central Bay				South Bay							
Soil Type	**landcover type**	**A**	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**I**	**J**	**K**	**L**	**M**	**N**	**O**	**P**	**Q**	**R**	**S**	**T**
Suisun	**Channel**	2.8	11.7																		
	Farmed Bayland																				
	Grazed Bayland	38.1																			
	Nontidal Wetland	3422.4	1148.5																		
	Salt Pond																				
	Tidal Flat		0.1																		
	Tidal Marsh	317.6	86.4																		
	Urban	7.9	12.3																		
	other	5.2	4.4																		
	TOTAL	3793.9	1263.4																		
Tamba	**Channel**	65.1	135.0																		
	Farmed Bayland		236.0																		
	Grazed Bayland	1171.7																			
	Nontidal Wetland	6457.0	4252.3																		
	Salt Pond																				
	Tidal Flat	0.2	11.6																		
	Tidal Marsh	1161.5	769.7																		
	Urban	36.3	122.8																		
	other	4.3																			
	TOTAL	8896.1	5527.5																		
Urban	**Channel**		6.6		6.1		1.1	3.8		24.4	11.3	3.4	4.9	7.5	10.8						
	Farmed Bayland							55.8									7.5				
	Grazed Bayland						80.2	16.4													
	Nontidal Wetland		40.0	105.4	117.8		20.7	449.9	18.8	65.8	208.6	1174.8	36.6	67.1	72.8	2.3	44.8				7.0
	Salt Pond				5.9		4.6	29.4		38.4	7.6	88.7	64.8	116.5	15.8						
	Tidal Flat		1.2	2.6	5.3		0.2	3.4	6.3	40.8	48.1	40.0	42.6	9.5	1.9	2.3					0.2
	Tidal Marsh		40.3	47.4	58.7		16.1	31.9	2.2	307.4	47.8	48.9	87.7	82.3	22.5	14.2	1.8				2.8
	Urban		458.9	472.8	1489.6		287.7	1561.7	621.1	3497.6	8280.4	7989.9	3795.8	6464.8	886.4	59.3	112.7			13.9	210.2
	other		1.3	0.3	17.1			4.3	15.7	123.6	20.2	49.8	74.0	91.6	1.1	17.3					
	TOTAL		548.3	628.4	1700.4		410.6	2156.6	664.0	4097.9	8623.9	9395.5	4106.5	6839.2	1011.2	95.5	166.8			13.9	220.3
Valdez	**Channel**	6.5																			
	Farmed Bayland																				
	Grazed Bayland	126.2																			
	Nontidal Wetland	6755.6	124.1		906.0																
	Salt Pond																				
	Tidal Flat																				
	Tidal Marsh	71.9	89.6		389.4																
	Urban	18.9	3.8		60.1																
	other	3.0																			
	TOTAL	6982.1	217.6		1355.6																
	Grand Total	35319.7	19597.4	9101.4	20662.6	19385.5	13663.4	9250.4	2018.2	4097.9	8673.3	9459.2	4228.2	7244.3	8178.2	7337.5	7910.4	7169.2	4540.8	7272.3	2392.5

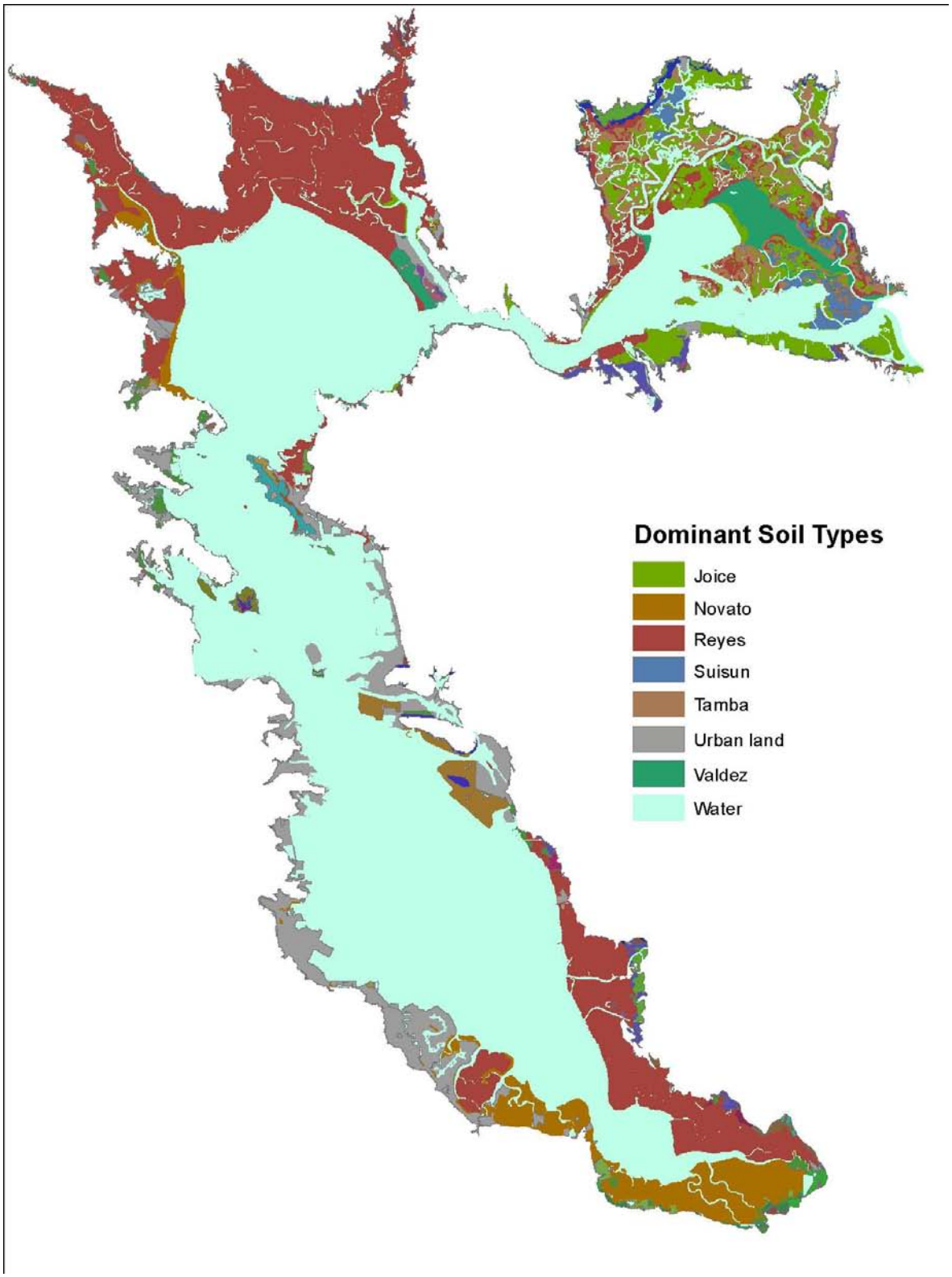

Figure 6.1. Distribution of the nine most dominant soil types in the San Francisco Estuary.

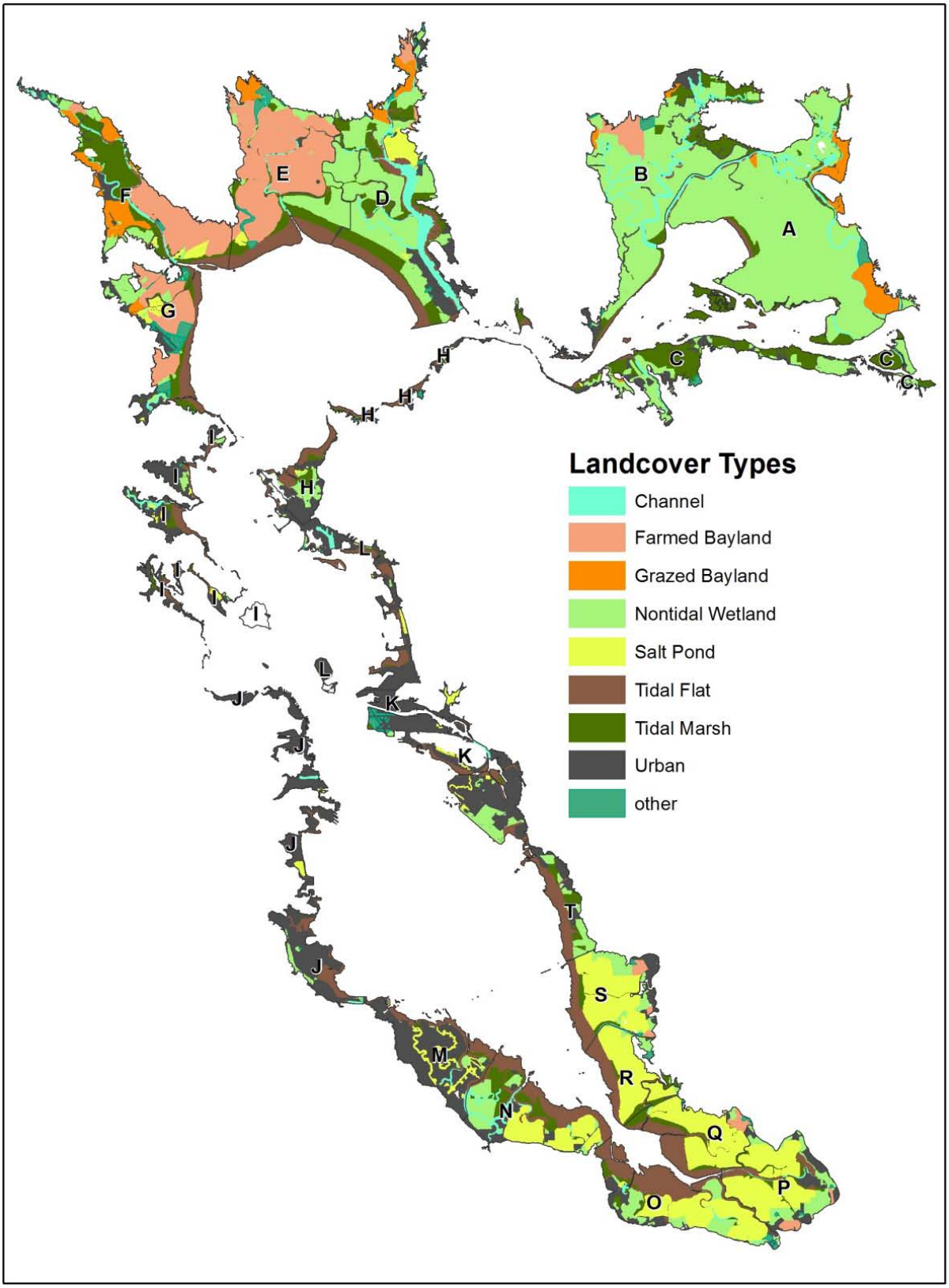

Figure 6.2. Dominant landcover types surrounding the San Francisco Estuary

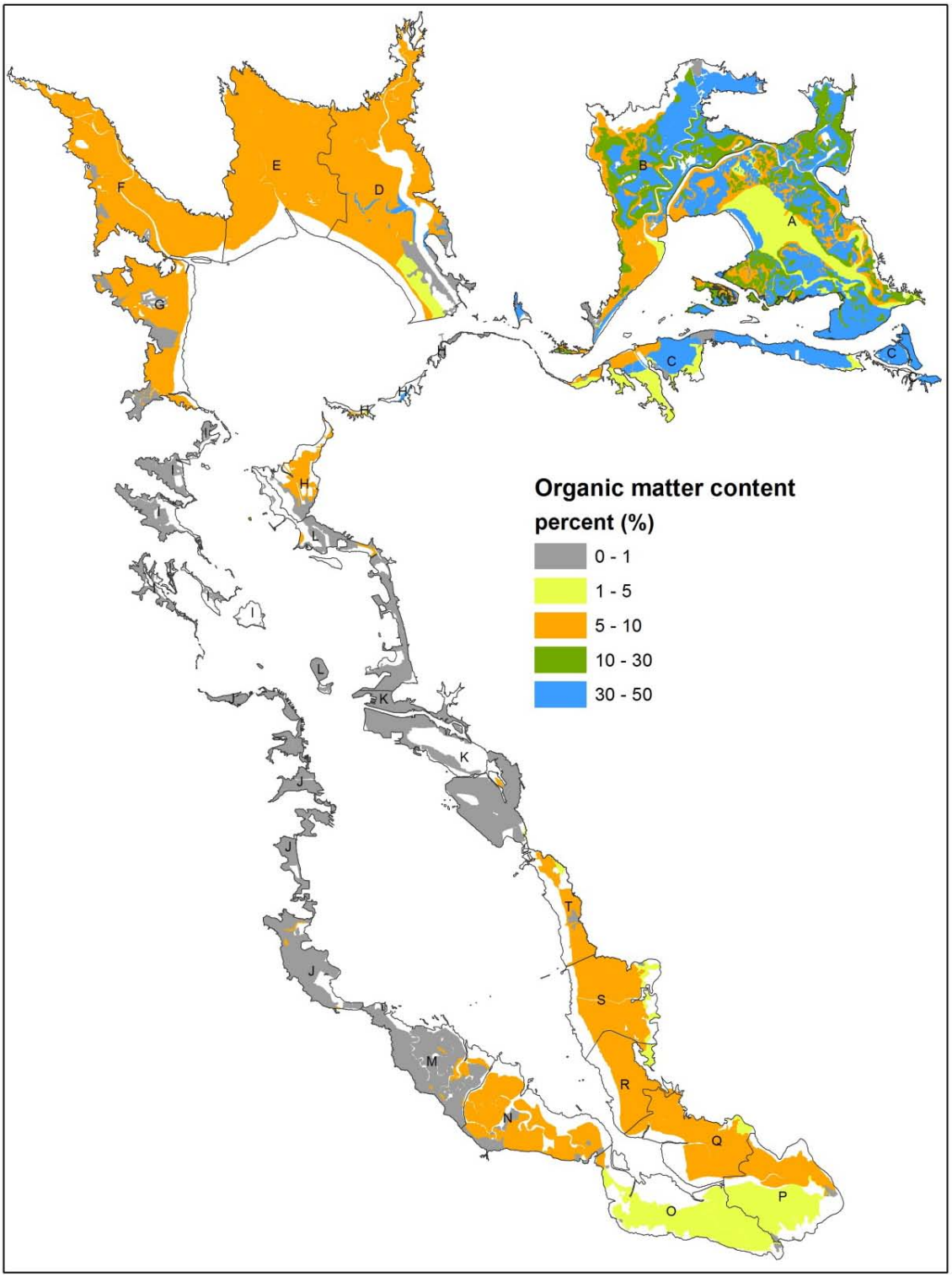

Figure 6.3. Average percent soil organic matter content.

Table 6.2. Estimated volume of soil and carbon lost when former tidal wetlands were converted to a nontidal landcover. Non-tidal wetlands are managed or muted wetlands or duck ponds, non-wetlands are areas that have been converted to farming, grazing or urban development, and salt ponds are or previously were non-tidal ponds managed for salt production.

		Volume (m^3) Lost by Converted Landcover Type					Carbon Lost	
Region	**Segment**	**nontidal wetland**	**non-wetland**	**salt pond**	**Total m^3**	**Total km^3**	**metric tons C**	**CO_2 equivalent**
Suisun	A	6,983,130	930,606		7,913,736	0.00791	264,319	968,582
	B	4,522,525	471,990		4,994,515	0.00499	166,817	611,291
	C	250,760	17,475		268,235	0.00027	8,959	32,830
North Bay	D	3,639,040	532,446	211,690	4,383,176	0.00438	102,128	374,243
	E	857,763	6,326,361	12,269	7,196,392	0.0072	167,676	614,439
	F	354,514	5,306,763	230,328	5,891,606	0.00589	137,274	503,035
	G	442,995	2,024,116	186,104	2,653,215	0.00265	61,820	226,536
	H	59,551	8,334		67,884	0.00007	1,582	5,796
Central Bay	I	17,148	118,258	31,323	166,729	0.00017	3,885	14,236
	J	32,960	154,474		187,434	0.00019	4,367	16,003
	K	15,648	504,034	18,721	538,403	0.00054	12,545	45,970
	L	71	7,165		7,236	0.00001	169	618
South Bay	M	117,221	2,053,593	415,976	2,586,791	0.00259	60,272	220,864
	N	814,486	129,703	14,806	958,995	0.00096	22,345	81,881
	O	731,175	333,374	3,096,949	4,161,498	0.00416	96,963	355,315
	P	673,283	1,306,415	1,660,603	3,640,300	0.00364	84,819	310,815
	Q	69,782	340,441	483,584	893,807	0.00089	20,826	76,315
	R	260,924	5,567	8,806	275,297	0.00028	6,414	23,505
	S	302,255	52,720	466,794	821,768	0.00082	19,147	70,164
	T	97,295	15,512		112,807	0.00011	2,628	9,632
Total		**20,242,526**	**20,639,348**	**6,837,952**	**47,719,827**	**0.04772**	**1,244,954**	**4,562,069**

Despite the lack of data for quantitative estimates, our analysis strongly suggests that, due to the extensive acreage and high organic matter content of its soils, Suisun Marsh is very likely an important source of on-going CO_2 emissions. Much of San Francisco Bay is likely to be a minor ongoing source as of CO_2 given the low organic content and mineral of the soils, and the substantial length of time since these soils were drained. In addition to being a likely source of CO_2 emissions the diked areas of Suisun Bay are also likely to be a source of CH_4 emissions, emanating from beneath standing water in ditches and duck ponds. In March 2014 an Eddy flux tower was placed on Rush Ranch as a first means to begin to quantify GHG fluxes in that region. (Lisa Windham-Meyers, personal communication).

Future Soil Carbon Dynamics in the Baylands

Based on projected restoration plans across the bay, a total of 0.28 to 0.30 million metric tons of carbon could be sequestered in restored tidal wetlands across the San Francisco Estuary (Table 6.3). This represents cumulative carbon sequestration across projected restoration projects described in Chapter XX and assumes that carbon would accumulate throughout the rooting depth of tidal wetland plants and across the entire vegetated wetland, with carbon densities reaching values equal to those found in natural tidal wetlands (as above). While this would take multiple decades, it indicates the magnitude of increased carbon sequestration with projected restoration that is possible.

Table 6.3 Estimated amount of carbon sequestered through current or planned restoration projects using two methods of calculation: GIS differencing and elevation range for vegetation establishment (0.7 to 2.0 m NAVD88).

				GIS method		Vegetation range method	
Segment	**Surface Area (m^2)**	**former wetland area (m^2)**	**% area of segment restored**	**Volume soil affected (m^3)**	**metric tons C**	**Volume soil affected (m^3)**	**metric tons C**
A	582739	8251215	0.07	800044	26721	757561	25303
D	1894290	5880508	0.32	1703556	39693	2462577	57378
E	883579	7111593	0.12	807378	18812	1148653	26764
F	563334	4422832	0.13	780685	18190	732334	17063
G	793293	1857552	0.43	1411562	32889	1031281	24029
N	861535	1066977	0.81	979949	22833	1119996	26096
O	1242885	4238253	0.29	2388472	55663	1615750	37656
P	1231506	3358368	0.37	2173095	50633	1600957	37302
S	1442008	2162356	0.67	519928	12114	1874610	43678
T	44137	173472	0.25	27160	633	57378	1337
Grand Total	**9539305**			**11591829**	**278182**	**12401097**	**296606**

There are likely to be additional GHG implications of restoring tidal wetlands beyond building carbon stocks that should be recognized. As within the Delta, reducing ongoing emissions may have greater benefits than the actual rebuilding of the carbon stock (Windham-Myers, unpublished data). Reductions in baseline carbon dioxide, methane and nitrous oxide emissions may well occur in Suisun with restoration of wetlands. Potentially there may also be reductions in GHG emissions from drained wetlands west of Carquinez Straits, though as mentioned above the existing soil carbon stock levels in these areas appear depleted.

At the current price of a carbon credit under the California market (approximately $12 per ton of CO_2[1]) carbon financing would not underwrite the cost of a wetlands restoration project. However, those funds might support existing staff to maintain a science program to provide the monitoring, reporting and verification for crediting. It has yet to be seen what the future price of carbon will be in coming year but given then need for GHG reductions the price is likely to rise.

In order to facilitate restoration and carbon sequestration, a range of novel restoration techniques should be evaluated, including the use of freshwater from waste water treatment plants. Subsidence reversal approaches have been tested in the Delta for over 10 years (Miller et al. 2009, Windham-Myers, unpublished data); however these methods have yet to be applied around the bay. In locations where natural water supplies would be too saline for reed growth freshwater could be derived from redirected waste water outflow. One such opportunity could be the Bel Marin Keys V property in Marin, a highly subsided potential restoration area immediately adjacent to the Novato Sanitary Districts waste water discharge pipeline. Here expansive areas of low drained Baylands could be converted to managed wetlands, which controlled water elevation management used to rebuild organic soils beneath reed beds. Such approaches could offer a solution to building elevations and filling large accommodation spaces that exist at Bel Marin Keys V and elsewhere. The combined approach of beneficial reuse of waste water, habitat restoration, and rebuilding shoreline elevations would be a useful climate change adaptation strategy in the urbanized setting of San Francisco Bay.

Similarly, restoration of duck pond areas is Suisun would provide multiple benefits, as restoration will likely promote carbon sequestration in recently restored tidal wetlands and simultaneously reduce net GHG emissions. Current management promotes standing water over organic soils, which may reduce current CO_2 emissions and protect soil carbon stocks compared to diked areas; however, this management could increase CH_4 emissions. Restoration of tidal wetlands in this area through dike breaching and subsidence reversal could potentially have a net benefit beyond projected increases in soil carbon sequestration through the reduction of methane emissions; however, further quantification of emission benefits under different land uses is needed.

Other land management options that might be tested include periodic lowering of water levels in the ponds to aerate soils and reduce methane production, as has been explored in rice production (Ma and Lu, 2011). However, halting subsidence through rice production does not address the issues of rising sea level and the combined resulting problems of increases level failure with increased hydrostatic pressure and the reduced restoration opportunity as land surface elevations that progressively fall below that at which vegetation will colonize and tidal wetlands build.

Again, further quantification would aid land use decision making. Subsidence reversal projects, whereby peat accumulation is promoted by impounded surface water, emergent vegetation and substantial accommodation space (e.g., 25 cm depth), may be successful in North Bay locations where climates favor high productivity and low decomposition rates. As per patterns observed in the Delta, CH_4 fluxes may be significant in early stages (Miller and Fujii 2010), and will likely decline with direct or airborne sulfate inputs. Opportunities to apply subsidence reversal organic soils building should examined for the Baylands. One of our greatest challenges is filling the open volume behind levees created by drainage and subsidence of levees. Subsidence reversal may fill some of that volume.

A further management option might be to manage drained Baylands in such a way as to increase carbon sequestration prior to breaching. Carbon accumulated on the drained Baylands would be stored beneath

[1] One ton of carbon equates to 3.66 tons of CO_2

the building wetland. An example of such an activity might include enrichment of soil carbon stocks by conversion to grassland or the application of compost or biochar materials.

An assessment of carbon sequestration on a restoring wetland in the Snohomish Estuary, Puget Sound, documented a breached brackish wetland colonized by soft-stemmed bulrush (*Schoenoplectus tabernaemontani*) to be accumulating carbon at a rate of 352 g $C/m^2/yr$ (Crooks et al. 2014). Though only a single data point it is comparable with rates of soil carbon accumulation of managed wetlands in the Sacramento San Joaquin Delta (Miller et al. 2009) and a comparable species to that which would colonize restored wetlands in Suisun Bay.

Data Gaps and Research Needs

As indicated above, major gaps remain in data and understanding of carbon dynamics within Bay wetlands. For example, rates of carbon sequestration in restored wetlands are not quantified, although managed wetlands in the Delta (Miller et al. 2009) and recently restored brackish wetlands in Puget Sound (Crooks et al. 2014) indicate the capacity of restored tidal wetlands to rapidly sequester carbon. We highlight research needs to quantify emissions and sequestration and to improve models of GHG and carbon dynamics:

Quantification of emissions and sequestration

1. **Measure GHG fluxes (CO_2, CH_4 and N_2O) in wetlands across the salinity gradient and under a range of conditions for nitrogen loading.** We have little data for CH_4 and N_2O emissions within the Baylands and need to improve estimates of emissions across the salinity gradient. Furthermore, emissions may be affected by nutrient loading, as N_2O emissions within tidal wetlands are a function of anthropogenic loading.

2. **Measure GHG fluxes and carbon dynamics for disturbed, managed, converted and restored wetlands.** These data would provide fundamental understanding of emissions and removals on drained Baylands or other modified wetlands (i.e., supporting the refinement of GHG emissions for land use activities including modified agricultural practices and wetland restoration). Restored tidal wetlands potentially sequester more carbon than undisturbed natural wetlands in equilibrium with sea-level rise as restored wetlands accrete at rates higher than sea-level rise and rebuild the soil carbon profile.

3. **Improve understanding of the fate of carbon and nitrogen released from eroding tidal wetlands.** If tidal wetlands lose elevation or are eroded, the long-term fate of carbon and nitrogen stored within wetland soils is uncertain. Some portion of the carbon and nitrogen released from the eroding wetlands could return back to the atmosphere, while some could be buried within adjacent subtidal sediments.

Models, emissions factors and simplifications

We are not yet at the stage of developing process-based or empirical models that enable refined assessment of wetland GHG dynamics or evaluate implications of management activities within the Baylands. Such models would improve GHG management across the landscape and greatly reduce the cost of addressing management questions. Eventually these models could lead to the development of emissions factors, which would greatly simplify emissions and removal predictions. The following modeling advances are needed:

1. **Develop models that describe GHG emissions and reductions with landscape change (e.g., wetland migration, conversion, restoration).** Spatial models exist that describe natural wetland development with sea-level rise across the estuary (Stralberg et al., 2012) and recently were extended to

include dynamic organic matter accumulation rates (carbon sequestration) within natural wetlands (Schile et al., 2014). When data gaps described in the section above have been addressed, additional elements, such as carbon sequestration within restored wetlands, could be added to these landscape models.

2. **Apply process-based models to improve understanding of carbon and nitrogen cycling**. Process-based models (e.g., the DNDC model) require detailed parameterization and have not been applied to Bayland wetlands. Application of these models could improve understanding of GHG emissions and support landscape modeling and management decisions related to Baylands.

3. **Identify simplified monitoring approaches / indicators for a range of emissions-related processes.** Quantification of carbon sequestration by wetlands is an established practice; however quantification across other processes or other types of wetlands is not simple. Based upon empirical quantification and / or model development described above, simplified monitoring approaches could be developed to aid assessment of management activities on drained organic soils or other modified wetlands (e.g., relationships between water table depth, temperature and CO_2 and CH_4 emissions). These simplified approaches would greatly reduce the cost of validation monitoring and model calibration.

4. **Determine default factors of emissions and removals by activity.** Some emissions factors can be developed now based upon existing data (e.g., sequestration of carbon by natural wetlands), some with relatively simple data collection (e.g., carbon sequestration by restoring tidal wetlands) and some will require more detailed quantification of emissions associated with land use (e.g., CH_4 emissions associated with water management on managed wetlands). Once these factors are developed, accounting for GHG emissions and removals across the Baylands will have achieved the same level of understanding that exists for terrestrial lands.

Recommended Management Actions

In addition to addressing the data gaps above, we suggest that the following management actions be considered:

1) Restore wetlands sooner rather than later. This will maximize carbon sequestration by initiating it immediately and increase resilience of restored wetlands to sea-level rise by building wetland elevation.

2) Prioritize restoration of wetlands in areas of high sediment availability to promote wetland resilience to sea-level rise and maintain long-term carbon sequestration.

3) Explore subsidence reversal and soil building beneath managed wetlands. Suisun Bay is a priority area for these activities, but opportunities exist for test applications on Baylands seaward of the Carquinez Straits. One opportunity is Bel Marin Keys V, as discussed above. Such activities offer the opportunity to increase subsided elevations prior to breaching of dikes, as well as increase carbon sequestration.

4) Evaluate activities that can be undertaken on Baylands to reduce ongoing GHG emissions and improve carbon sequestration. These could involve: raising water tables to reduce soil carbon loss from dry soils, filling ditches to reduce CH_4 emissions, and reducing use of fertilizer or cattle densities to reduce soil CH_4 and N_2O emissions. Additional options include the developing approaches to use of compost to soils from recycled food waste. This may be further enhanced with integrated management of waste water disposal (as currently occurs on the St. Vincent Property). Enhancing soil carbon sequestration

on Baylands would have benefits of their own as well as establishing a pool of carbon that would be buried beneath a restoration project.

5) Finally, develop a more detailed plan for prioritizing activities to incorporate climate change mitigation into Baylands management. Developing such a plan would require additional effort beyond this early assessment provided within this chapter.

SUMMARY

- To understand the climate regulation implications of landscape management strategies, we must compare GHG emissions from the present day landscape (known as the baseline) and compare with GHG fluxes due to management activities.

- Substantial organic carbon was lost from estuarine wetlands when they were diked and drained (approximately 1.2 million metric tons of carbon).

- Organic-rich soils in diked brackish marshes across the San Francisco Estuary are likely to continue releasing CO_2, while more mineral-rich soils in diked salt marshes are probably depleted of oxidizable carbon. Data are not available to describe the emissions of other GHGs across the landscape.

- Tidal wetlands within the Estuary sequester greater carbon per unit area compared to many other ecosystems, with 14,560 metric tons of CO_2 sequestered annually across all tidal wetlands in the estuary.

- Similarly, data are lacking to estimate GHG emissions from tidal wetlands; however sulfate likely limits CH_4 emissions in high salinity wetlands. No data are available to evaluate N_2O emissions across the Baylands. Potentially, GHG reductions could be attained by reducing the extent of impounded freshwater behind barriers in Baylands, as well as recognizing the benefits of reducing nitrogen loading to coastal waters from industry and agriculture.

- Restored wetlands are net sequesters of carbon and considering the salinity gradient across the estuary are likely net removers of greenhouse gases over all. It would be helpful to inform management decisions if the fluxes in GHGs across the landscape were quantified.

LITERATURE CITED

Anderson, F., B. Bergamaschi, L. Windham-Myers, R. Miller, and R. Fujii. 2012. Observations of the atmospheric carbon cycle in restored wetlands. FluxNet Newsletter **4**:17-22.

Callaway, J. C., E. L. Borgnis, R. E. Turner, and C. S. Milan. 2012. Carbon sequestration and sediment accretion in San Francisco Bay tidal wetlands. Estuaries & Coasts **35**:1163-1181.

Cloern, J. E. and A. D. Jassby. 2012. Drivers of change in estuarine-coastal ecosystems: Discoveries from four decades of study in San Francisco Bay. Reviews of Geophysics **50**: RG4001, doi:4010.1029/2012RG000397.

Crooks, S. 1999. Formation of overconsolidated horizons within estuarine alluvium: Implications for the interpretation of Holocene sea-level curves. Pages 197-215 *in* S. Marriot, J. Alexander, and R. Hey, editors.

Floodplains: Interdisciplinary Approaches: Geological Society Special Publication, 163. Geological Society, London.

Crooks, S. 2000. Sedimentological controls on the erosion and morphology of saltmarshes: Implication for flood defence and habitat recreation. Pages 207-222 *in* K. Pye and J. R. L. Allen, editors. Coastal and Estuarine Environments: Sedimentology, Geomorphology and Geoarchaeology: Geological Society Special Publication, 175. Geological Society, London.

Crooks, S., D. Herr, J. Vanderver, J. Tamelander, and D. Laffoley. 2011. Value of coastal wetlands and marine ecosystem restoration and management in regulating climate change: Policy opportunities, ecosystem status and trends. World Bank Environment Working Paper.

Crooks, S., Rybczyk,J., O'Connell, K.,. Devier, D.L., Poppe, K., and Emmett-Mattox, S. (2014) Coastal Blue Carbon Opportunity Assessment for Snohomish Estuary: The Climate Benefits of Estuary Restoration. Report by Environmental Science Associates, Western Washington University, EarthCorp and Restore America's Estuaries, January 2014.

Crooks, S., Emmer, I., von Unger, M., Brown, B., Murdiyarso, D. and Orr, M.K. (in prep). Guiding Principles for Delivering Coastal Wetland Carbon Projects. Report to the United Nations Environment Program and the Center for International Forestry Research.

Deverel, S. J. and D. A. Leighton. 2010. Historic, recent, and future subsidence, Sacramento-San Joaquin Delta, California, USA. San Francisco Estuary and Watershed Science 8 http://www.escholarship.org/uc/item/7xd4x0xw.

Drexler, J. Z. 2011. Peat formation processes through the millennia in tidal marshes of the Sacramento-San Joaquin Delta, California, USA. Estuaries and Coasts **34**:900-911. DOI 910.1007/s12237-12011-19393-12237.

Drexler, J. Z., C. S. de Fontaine, and T. A. Brown. 2009. Peat accretion histories during the past 6,000 years in marshes of the Sacramento-San Joaquin Delta, CA, USA. Estuaries and Coasts **32**:871-892.

Emmer,I., Needleman, B., Emmett-Mattox, S., Crooks, S., Megonigal., P.M., Myers, D., Oreska, M. and McGlathery, K. (2013). Methodology for Tidal Wetlands and Seagrass Restoration. Draft submitted for review to the Verified Carbon Standards, Dec 2013.

EPA [Environmental Protection Agency] (2010) Methane and Nitrous Oxide emissions from Natural Sources. EPA 430-R-10-001. EPA, Washington, DC.

ESRI Inc. 2013. Environmental Science Research Institute.

Dugdale, R. C., F. P. Wilkerson, V. E. Hogue, and A. Marchi. 2007. The role of ammonium and nitrate in spring bloom development in San Francisco Bay. Estuarine Coastal and Shelf Science **73**:17-29.

Goman, M. 2001. Statistical analysis of modern seed assemblages from the San Francisco Bay: applications for the reconstruction of paleo-salinity and paleo-tidal inundation. Journal of Paleolimnology **26**:393-409.

Keller, J. K., A. A. Wolf, P. B. Weisenhorn, B. G. Drake, and J. P. Megonigal. 2009. Elevated CO_2 affects

porewater chemistry in a brackish marsh. Biogeochemistry **96**:101-117.

Knowles, N. and D. R. Cayan. 2002. Potential effects of global warming on the Sacramento/San Joaquin watershed and the San Francisco estuary. Geophysical Research Letters **29**:Article Number 1891.

Laanbroek, H. J. 2010. Methane emission from natural wetlands: interplay between emergent macrophytes and soil microbial processes. A mini-review. Annals of Botany **105**:141-153.

Lovelock, C. E., R. W. Ruess, and I. C. Feller. 2011. CO_2 efflux from cleared mangrove peat. Plos One **6**.

Ma, K., and Lu, Yahai (2011) Regulation of microbial methane production and oxidation by intermittent drainage in rice field soil. Microbiology Ecology. 75, 446-456.

Malamud-Roam, F. and B. L. Ingram. 2001. Carbon isotopic compositions of plants and sediments of tide marshes in the San Francisco Estuary. Journal of Coastal Research **17**:17-29.

McKee, L. J. and D. C. Gluchowski. 2011. Improved nutrient load estimates for wastewater, stormwater and atmospheric deposition to South San Francisco Bay (South of the Bay Bridge). [Available at http://bayareanutrients.aquaticscience.org/sites/default/files/u23/Report_Nutrient_load_to_South_Bay_2011-8-31_revised.pdf]. Bay Area Clean Water Agencies, San Francisco, CA.

Miller, R. L., M. Fram, R. Fujii, and G. Wheeler. 2008. Subsidence reversal in a re-established wetland in the Sacramento-San Joaquin Delta, California, USA. San Francisco Estuary and Watershed Science **6**: Available at http://escholarship.org/uc/item/5j76502x.

Miller, R. L. and R. Fujii. 2010. Plant community, primary productivity, and environmental conditions following wetland re-establishment in the Sacramento-San Joaquin Delta, California. Wetlands Ecology and Management **18**:1-16.

Morris, J. T., J. Edwards, S. Crooks, and E. Reyes. 2012. Assessment of carbon sequestration potential in coastal wetlands. Pages 517-531 *in* R. Lal, K. Lorenz, R. Huttl, B. U. Schneider, and J. von Braun, editors. Recarbonization of the biosphere: ecosystems and the global carbon cycle. Springer, New York.

Null, K. A., N. T. Dimova, F. L. Knee, B. K. Esser, P. W. Swarzenski, M. J. Singleton, M. Stacey, and A. Paytan. 2012. Submarine groundwater discharge-derived nutrient loads to San Francisco Bay: Implications to future ecosystem changes. Estuaries and Coasts **35**:1299-1315.

Nyman, J. A., R. D. DeLaune, and W. H. Patrick, Jr. 1990. Wetland soil formation in the rapidly subsiding Mississippi River Deltaic Plain: Mineral and organic matter relationships. Estuarine, Coastal and Shelf Science **31**:57-69.

Nyman, J. A., R. D. DeLaune, H. H. Roberts, and W. H. Patrick, Jr. 1993. Relationship between vegetation and soil formation in a rapidly submerging coastal marsh. Marine Ecology Progress Series **96**:269-278.

Orr, M., S. Crooks, and P. B. Williams. 2003. Will restored tidal marshes be sustainable? San Francisco Estuary and Watershed Science **1**:Article 5. Available at: http://repositories.cdlib.org/jmie/sfews/vol1/iss1/art5/.

Pendleton, L., D. C. Donato, B. C. Murray, S. Crooks, W. A. Jenkins, S. Sifleet, C. Craft, J. W. Fourqurean, J. B. Kauffman, N. Marba, P. Megonigal, E. Pidgeon, D. Herr, D. Gordon, and A. Balder. 2012. Estimating global "blue carbon" emissions from conversion and degradation of vegetated coastal ecosystems. PLoS ONE **7**:e43542. doi:43510.41371/journal/pone.004342.

Poffenbarger, H., B. Needelman, and J. Megonigal. 2011. Salinity influence on methane emissions from tidal marshes. Wetlands **31**:1-12.

Reddy, K. R. and R. D. DeLaune. 2008. Biogeochemistry of wetlands: Science and applications. CRC Press, Boca Raton, FL.

Rojstaczer, S. and S. J. Deverel. 1995. Land subsidence in drained histosols and highly organic mineral soils of California. Soils Science Society of America Journal **59**:1162-1167.

Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. 2013. Soil Survey Geographic (SSURGO) Database for California. Available online at http://soildatamart.nrcs.usda.gov. Natural Resources Conservation Service, USDA, Washington, DC.

Stralberg, D., M. Brennan, J. C. Callaway, J. K. Wood, L. M. Schile, D. Jongsomjit, M. Kelley, V. T. Parker, and S. Crooks. 2011. Evaluating tidal marsh sustainability in the face of sea-level rise: A hybrid modeling approach applied to San Francisco Bay. PLoS ONE **6**:e27388. doi:27310.21371/journal.pone.0027388.

Teh, Y., W. Silve, O. Sonnentag, M. Detto, M. Kelly, and D. Baldocchi. 2011. Large greenhouse gas emissions from a temperate peatland pasture. Ecosystems **14**:311-325.

Turner, R.E., E. M. Swenson, and C. S. Milan. 2000. Organic and inorganic contributions to vertical accretion in salt marsh sediments. Pages 583-595 *in* M. P. Weinstein and D. A. Kreeger, editors. Concepts and controversies in tidal marsh ecology. Kluwer Academic Publishers, Boston, MA.

Van Groenigen, K. J., C. W. Osenberg, and B. A. Hungate. 2011. Increased soil emissions of potent greenhouse gases under increased atmospheric CO_2. Nature **475**:284-286.

Table 7.J 1 Basic characteristics of soil series in the San Francisco Estuary (Soil Survey Staff, 2012).

Soil Series	% Organic Matter, average	Minimum Depth to Water Table in Summer (cm)	Soil Temperature (C)
Joice	45.0	76.0	17.2
Novato	5.5	46.0	14.0
Omni*	2.5	50.0	16.0
Reyes	6.0	91.0	15.5
Suisun	50.0	76.0	17.0
Tamba	22.5	91.4	17.2
Valdez*	1.3	91.4	16.7

*Minimum depth to water table only available for drained soil.

Table 7.J 2 The areal extent of particular soil series in agricultural Baylands.

Soil Series	Agriculture Baylands	Area in km^2
Joice	Farmed Bayland	0.08
	Grazed Bayland	1.56
Novato		
	Farmed Bayland	0.03
	Grazed Bayland	0.04
Reyes		
	Farmed Bayland	86.84
	Grazed Bayland	
Suisun		
	Farmed Bayland	0
	Grazed Bayland	0.15
Tamba		
	Farmed Bayland	0.96
	Grazed Bayland	4.74

Table 7.J 3 Area of Soil Series Coverage per Sub-region of the San Francisco Estuary (km^2). Soils series are not indicated within the Central Bay as Bayland areas in this sub-region are all classified as "urban".

Soil Series	Suisun	North Bay	South Bay	**Total**
Joice	1.64			1.64
Tamba	5.7			5.7
Suisun	0.15			0.15
Novato		0.07		0.07
Reyes	3.52	98.5	0.9	102.92
Total	11.01	98.57	0.9	**110.48**