Modeling the Economic Value of Carbon Sequestration by Wetlands in the Delaware Estuary: Historic Estimates and Future Projections

Edward W. Carr[a]*
Yosef Shirazi[a]
George Parsons[a]
Porter Hoagland[b]
Christopher K. Sommerfield[c]

* Corresponding Author. Email: ecarr@udel.edu Phone: (207) 518-8662
[a] School of Marine Science and Policy, University of Delaware, Newark, DE, 19716. USA
[b] Marine Policy Center, Woods Hole Oceanographic Institution, 266 Woods Hole Rd., MS# 41, Woods Hole, MA 02543. USA
[c] School of Marine Science and Policy, University of Delaware, Lewes, DE, 19958. USA

Abstract

Coastal wetlands sequester large amounts of carbon in their soils, effectively removing carbon dioxide from the atmosphere and acting as a carbon sink. In this paper, we estimate the economic value of carbon sequestered by wetlands in the Delaware Estuary. We estimate the value of the current stock of wetlands, the value of the historic loss of wetlands, and under a range of different scenarios the expected future loss. We use historical topographic maps and Land Cover inventories of the Delaware Estuary to measure the acreage of tidal wetlands in nine distinct time periods from 1778 to 2011. Using these data, we estimate an annual rate of wetland loss of 1.03km^2. Coupling observed land cover change with exogenous factors including sea-level rise, population pressure, and channel dredging, we estimate changes in tidal wetland area under a range of future scenarios for our expected future economic loss estimates.

Keywords

carbon sequestration; blue carbon; tidal wetlands; ecosystem services; social cost of carbon

1. Introduction

The Delaware Estuary, situated in the Mid-Atlantic region of the United States is a working estuary, like many around the world. As of the 2010 census, the counties bordering the Delaware Estuary are home to 6 million people (U.S. Census Bureau, 2010). The Delaware Bay estuary has been important to regional maritime commerce since the 1800s, is heavily developed in its upper reaches and less developed in its lower reaches, and provides essential habitat to a number of important commercial and recreational species, including horseshoe crabs and the Red Knot (Myers et al., 2010). The Delaware estuary represents an important example for other working estuaries, where natural ecosystems and human activity collide.

The Delaware estuary is home to over 700 km^2 of coastal wetlands, and like many other wetlands in the United States and around the world, wetland area is declining (Dahl, 1990; Tiner et al., 2011; Tiner Jr, 1985). Human activities including diking (Weishar et al., 2005), conversion residential and farming land, pollution, and channel dredging have caused Delaware wetland area to decline by 54% since pre-colonial conditions. The Delaware Estuary is not unique in rapidly losing wetlands, and there is a pressing need for better understanding of the value of lost wetlands to improve management decisions.

Coastal wetlands play an important role in regulating atmospheric carbon dioxide, but are under threat from anthropogenic modifications. This work develops and outlines a replicable model that can be applied to value changing coastal marsh ecosystems, the results of which provide important insights for policy makers to improve coastal management decisions. We use the Delaware Estuary as an example for how to apply the model.

1.1 Ecosystem services

Coastal wetlands rank among the most highly productive ecosystems on earth, providing critical habitat for numerous aquatic, terrestrial, and avian species (Clark et al., 1993; Roman et al., 2000). Refuge and nursery habitat are just two of the ecosystem services provided by coastal wetlands. Ecosystem services are defined as the natural processes and components of ecosystems that provide goods and services that satisfy human needs, either directly or indirectly (De Groot, 1992; Fenichel et al., 2016; Guerry et al., 2015; Tiner, 2003). Other ecosystem services provided by wetlands include water filtration, sediment retention, storm surge buffering, and carbon sequestration (Barbier et al., 2011; Morgan et al., 2009; Pinsky et al., 2013). Carbon stored and sequestered in coastal and marine ecosystems is often termed “Blue Carbon” (McLeod et al., 2011; Nellemann et al., 2009) and is the primary focus of this paper.

Coastal wetlands play an important role in climate change mitigation. Taken together, all blue carbon sinks, including mangroves, seagrasses, and intertidal marshes, sequester carbon at a rate in excess of an estimated 100 $TgC \cdot yr^{-1}$, roughly equivalent to carbon sequestration from all terrestrial forests (Hopkinson et al., 2012).

Carbon dioxide fixed in coastal wetland plants via photosynthesis can be stored for thousands of years in anoxic soils as slowly decaying peat (Mitsch and Gosselink, 2015). Wetlands also capture and bury carbon rich detritus, further increasing the rate of carbon storage in soils. Wetlands emit methane (CH_4), a potent greenhouse gas, as a by-product of organic matter decomposition. However, sulphate reducing bacteria in coastal soils hinder the flux of methane, and thus coastal wetland systems emit much lower quantities of CH_4 than their freshwater counterparts (Bartlett and Harriss, 1993; Bridgham et al., 2006; Poffenbarger et al., 2011). Additionally, fluxes of the greenhouse gas N_2O, high in terrestrial ecosystems, are also low in coastal wetlands (Smith et al., 1983).

The estimated value of carbon sequestration in coastal wetlands varies widely. Barbier et al. (2011) estimate a value of $3,420 $km^{-2}·yr^{-1}$, while Costanza et al. (2014) calculate a mean value of $7,550 $km^{-2}·yr^{-1}$ (2015$) from a range of published and unpublished studies. Values from the United Kingdom range from $6,070 to $20,780 $km^{-2}.yr^{-1}$ (Beaumont et al., 2014). All values are converted to 2015$ using year of publication exchange rates, where necessary, and the consumer price index.

1.2 Wetland modification

The boundaries of coastal wetlands change both vertically and horizontally. Marshes can accrete vertically in sediment rich estuaries, but fail to keep pace with sea level rise in sediment poor estuaries or when belowground biomass productivity is insufficient to increase soil volume (Lentz et al., 2016; Weston, 2014). Tides and wave action influence the seaward boundary of marshes (Mariotti and Fagherazzi, 2010), whereas surface hydrology and plant interactions determine the landward boundary with uplands. Human modifications, such as draining and filling, diking and impoundment for water control, and land reclamation, all affect sediment dynamics and alter accretion and erosion rates (Kennish, 2001). Furthermore, wetlands and their surrounding areas are increasingly vulnerable to the effects of climate change, including sea level rise and temperature effects, causing ecosystem services to decline (Osland et al., 2016).

1.3 Carbon sequestration rates

Sequestration rates in North American salt marshes vary widely, ranging from 18 $gC·m^{-2}·yr^{-1}$ to 1,713 $gC·m^{-2}.yr^{-1}$ with a mean of 214.6 $gC·m^{-2}·yr^{-1}$ over 85 sites (Chmura et al., 2003). Mean values from the United Kingdom are typically lower, around 120-150 $gC·m^{-2}·yr^{-1}$ (Beaumont et al., 2014). Carbon sequestration rates vary widely, even at nearby sites, and among different methods of measurement. Thus, there is inherent uncertainty in estimating sequestration rates accurately from a collection of point estimates.

In addition to removing carbon from the atmosphere, wetlands also perform the important function of storing carbon in their soils for periods of hundreds to thousands of years if undisturbed (Artigas et al., 2015; O'Reilly et al., 2014). However, the fate of carbon sequestered in wetlands is difficult to trace once wetlands become degraded or eroded (Bauer et al., 2013; Crooks et al., 2011; Deverel and Leighton, 2010). There are three possible pathways by which marsh carbon can move after disturbance; microbial remineralization and release as CO_2, consumption by detritivores, and physical transport to other locations, such as other parts of the marsh, deep water, or other estuarine habitats (Macreadie et al., 2013). Pendleton et al (2012) speculate between 25%-100% of carbon in surface sediments (<1m deep) and wetland biomass may be reemitted to the atmosphere, mainly as CO_2. Landscape-scale studies show losses of soil carbon of up to 96% (Sigua et al., 2009) with similar results in the agricultural literature (Kirkels et al., 2014). Conversely, Lane et al. (2016) found that less than 10% of available carbon was actually mineralized, post-disturbance, while Macreadie et al. (2013) found disturbed marshes showed ~30% lower soil organic carbon, though the fate of the lost carbon is unclear. In short, the results are highly variable.

1.4 The social cost of carbon

The social cost of carbon (SCC) refers to the estimated economic damages (in present value terms) of the release of an additional metric ton of CO_2 (or CO_2 equivalent) into the atmosphere. Analogously, SCC represents the avoided damages from removing a metric ton of CO_2 (or CO_2 equivalent) from the atmosphere. SCC values are estimated using integrated economic damage assessment frameworks, or Integrated Assessment Models (IAMs), that employ coupled climatic and economic models. The latter involve the prediction of the potential damages associated with given atmospheric levels of CO_2 to estimate future costs (Hope, 2011; Nordhaus, 2014; Tol, 2009; Waldhoff et al., 2014). Uncertainty regarding the damage function's parameters, including the discount rate, and the resulting estimates of future economic losses affect SCC values greatly, which range from \$6 to upwards of \$125/tCO_2 (van den Bergh and Botzen, 2014).

2. Material and methods

We begin with a discussion of our study site, the Delaware Estuary, and then outline our model for valuing changes in carbon sequestration under uncertain land cover scenarios.

2.1 Study Site: The Delaware Estuary

2.1.1 Morphology and plant distribution in the Delaware Estuary

The Delaware Estuary is a funnel-shaped estuary that extends from the mouth of the Delaware Bay at Cape Henlopen, Delaware to the "head of tide" at Trenton, New Jersey. The Delaware Estuary can be divided into the Upper Estuary (oligohaline), the Lower Estuary (mesohaline), and the Delaware Bay (polyhaline) (DRBC, 2004). The Upper and Lower Estuaries around Philadelphia, PA and Wilmington, DE are highly developed, while farming and wetland dominate land cover in the Delaware Bay.

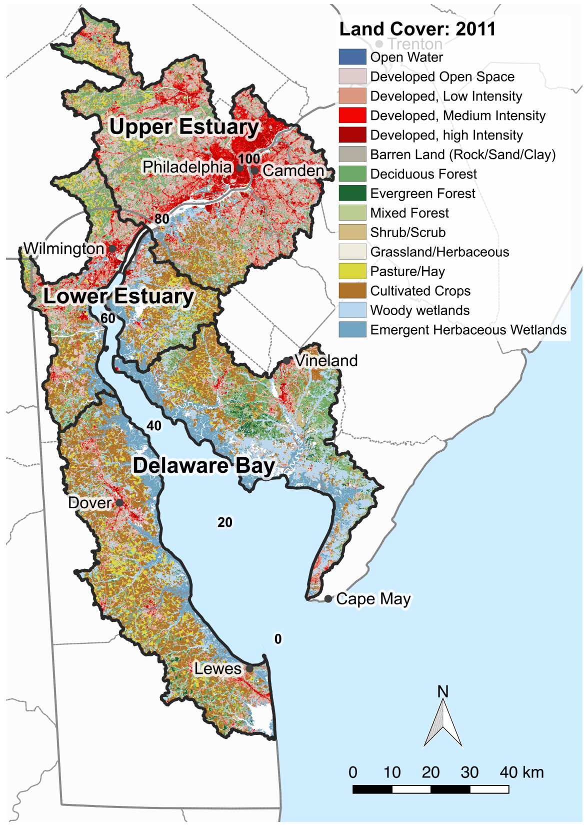

Figure 1: Land cover in the Delaware River Estuary, including the Upper Estuary, Lower Estuary, and Delaware Bay regions as defined by the Delaware River Basin Commission (DRBC, 2004). Numbers show DRBC river mile markers (DRBC, 2016). Land cover source: Homer et al. (2015)

Delaware Bay wetlands are dominated by salt marsh and characterized by extensive stands of smooth cordgrass (*Spartina alterniflora*) in the low marsh and salt hay (*Spartina patens*) and spike grass (*Distichlis spicata*) in the high marsh (Philipp, 2005). The common reed (*Phragmites australis*), an invasive species, is present in localized stands throughout the estuary and bay (Philipp and Field, 2005).

2.1.2 State of wetlands in the Delaware Estuary

Tidal marshes surrounding the Delaware Estuary and Bay have been decreasing in areal extent for centuries under natural and human influences (Kraft et al., 1992; Philipp, 2005). Wetland areas are currently transitioning to open water, including interior ponds, which can accelerate with increasing rates of relative sea level rise (Kearney et al., 2002). Human factors such as diking and impoundment (Weishar et al., 2005), conversion to cropland, pollution, channel dredging, and invasive species, have caused Delaware wetland area to decline by 54% since pre-colonial conditions (Dahl, 1990; Tiner et al., 2011), largely transitioning to open water or unconsolidated intertidal shoreline (Tiner et al., 2011). Kearney *et al.* (2002) estimate that approximately 56% of the Delaware Estuary marshes are moderately to severely degraded.

2.1.3 Consequences of shipping channel deepening in the Delaware Estuary

The Delaware River is an important waterway, serving the Ports of Philadelphia and Wilmington, as well as several local berths. The natural depth of the Delaware River is 17 to 24ft, with a navigable channel 175 to 600ft wide (USACE, 2016). The river's main channel was deepened to 35ft in the early 20th century and further deepened to 40ft by 2010. In 2010, work began to accommodate "New Panamax" vessels, which require a depth of 45ft, by 2017.

The deepening of shipping channels by dredging increases estuarine erosion (Johnston, 1981; Kennish, 2002; Pye and Neal, 1994; Sasser et al., 1986) and may further affect sediment flows within an estuary, leading to changes in sediment dynamics (Cox et al., 2003; Turner, 1997).

2.1.4 Valuation of carbon storage in the Delaware Estuary

Flight *et al*. (2012) estimated the value of the pool of *in situ* carbon currently stored in wetlands. Using the carbon sequestration model from InVEST (Natural Capital Project, 2017) and modeled land use change projections from 2007-2022, Flight *et al*. (2012) estimated an annualized value of $1.59 million in lost carbon storage associated with the loss of 3,132, or 12.67 km^2, wetland acres in Delaware. Of the five key ecosystem services valued by Flight *et al*. for Delaware wetlands, carbon storage comprised two-thirds of the losses.

2.2 Area calculations

We focused only on wetlands in watersheds (HUC10) that were hydrologically connected to the Delaware Estuary (USGS, 2015a). Our earliest map showing wetland areas in the Delaware Estuary dates to 1778 (Faden, 1778), and we used USGS topographic maps showing wetland delineation for years between around 1900 to 1965 (USGS, 2015b). Full coverage of the Delaware Bay was not available for all years prior to 1918 so we combined wetland areas from the years 1899 to 1918 into one product, which we refer to as the 1918-year. After 1965 we used the land-use, land-cover (LULC) digital raster products from the National Land Cover Dataset (NLCD) for the years 1970-80, 1992, 2001, 2006, and 2011 (Fry et al., 2011; Homer et al., 2007, 2015; Price et al., 2006; Vogelmann et al., 2001).

NLCD datasets are classified according to the Anderson (1976) classification scheme. Tiner (2011) estimates that over 96% of estuarine wetlands in Delaware are "emergent," and thus we focused on emergent herbaceous wetlands. Further GIS analysis confirmed that, with some minor exceptions, all areas of "Emergent Herbaceous Wetland" were at elevations less than 1.5m, and thus we assume they are intertidal marsh; henceforth described as tidal wetlands.

In the final analysis we have data points for the years: 1778, c1918, 1946, 1959, 1966, c1975, 1992, 2001, 2006, and 2011.

2.3 Regressions for estimating historic and future Area Loss

We estimated a series of linear regressions using the time series data from above. We consider shoreline (within 1km of current shoreline) and inland wetlands separately. Our first regression considers trends over the inland and shoreline wetlands together:

$$W(t) = \beta_o + \beta_t yr(t) + \beta_d rd(t) + \varepsilon(t) \tag{1}$$

where t = year; W(t) = total wetlands in square kilometers in year t; yr(t) = year index where yr(1778) = 1; yr(1779) = 2, etc.; rd(t) = 1 if NLCD raster data, 0 if not, and $\varepsilon(t)$ = an error term. β terms denote regression coefficients for corresponding variable estimated by ordinary least squares regression methods.

The $rd(t)$ variable tests for possible systematic differences between NLCD raster and hand-delineated wetland identification, which may involve measurement error.

Finally, we consider the loss of shoreline wetlands (SW) and inland wetlands (IW), hypothesizing that different mechanisms may have accounted for the losses. Our models here are:

$$SW(t) = \beta_o + \beta_{sl} sea(t) + \beta_{dt} depth(t) + \varepsilon(t) \tag{2}$$

$$IW(t) = \beta_o + \beta_p pop(t) + \varepsilon(t) \tag{3}$$

where *pop(t)* = population/10,000 in contiguous counties in year t, *sea(t)* = sea level at Philadelphia in year *t (meters), depth(t)* = dredge depth in year t (feet), and $\varepsilon(t) =$ an error term.

Sea level records at Philadelphia dating back to 1900, show a nearly linear rate of sea level rise of 2.94 mm/year (NOAA, 2016). Dredge depth follows the schedule outlined in Section 2.1.3. Population data come from the National Historical Geographic Information System (NHGIS) decadal census reports (Minnesota Population Center, 2011).

2.4 Carbon sequestration and the social cost of carbon

Sequestration rates are uncertain, therefore we employ a Monte Carlo simulation approach, using the sequestration rates in Table 1, to quantify changes in carbon sequestration. We do not account for CH_4 emissions from tidal freshwater wetlands in the Upper Estuary as freshwater wetlands account for just 5.9% of the wetland area.

Our 2011 base SCC values, in 2015 dollars are $12.77, $37.15, and $59.21 per ton CO_2 using discount rates of 5%, 3%, and 2.5% respectively (EPA, 2015), which approximates the current literature (Nordhaus, 2014; Tol, 2009; van den Bergh and Botzen, 2014). EPA also recommend a 95th percentile estimate of $104.48 (3%

discount rate) as an upper bound on the damages associated with atmospheric CO_2. We incorporate an annual SCC growth rate of 2.2% (Anthoff et al., 2011).

Finally, we characterize sequestration uncertainty in a Monte Carlo simulation and estimate the sensitivity of our results to discounting by using the three SCC values above. Sequestration rates are given in MgC·km2·yr-1, (1 Mg = 1000 kg), but SCC values refer to a Mg of CO_2, which is the standard way of reporting these values (to convert MgC to $MgCO_2$ use the ration of molar masses ($Mr[CO_2]/Mr[C] = 44/12$).

2.5 Scenarios and sequestration values

We estimate the value of sequestration loss for the current area of wetland, for the historic total area, and for future expected area changes. For historic losses, we first estimate the area lost since 1778. We then report the net present value of those wetlands today as if that stock of wetlands were present and sequestering carbon going forward from today indefinitely. We do not estimate historic sequestration values. For projected future wetland loss, we report the present value of expected area losses, i.e., the value of the declining flow of sequestration services in the future from lost wetlands. The general flow of the model described in this section is presented in Figure 2.

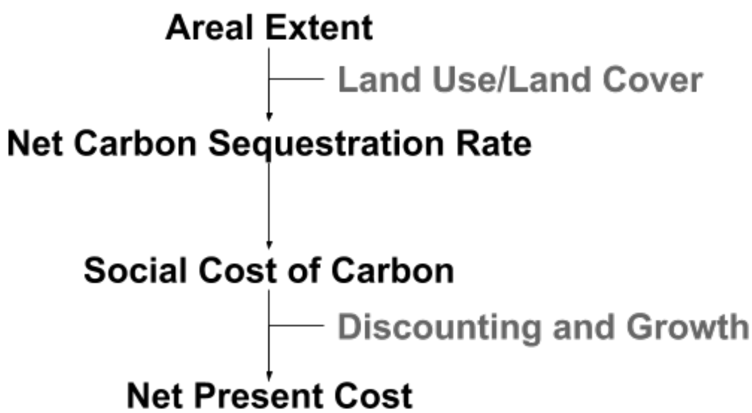

Figure 2: Flow diagram detailing the general structure of the model described in section 2.5

Our unit of value is dollars (2015$) per square kilometer, and we assume that all areas of tidal marsh sequester carbon equally. Suppose a square kilometer of wetlands sequesters α_{SR} Mg of carbon per year. Given a social cost of carbon, SCC_t, the *annual value* of a square kilometer of wetlands that is lost completely is:

$$v_t' = \alpha_{SR}\, SCC_t \tag{4}$$

v_t' includes the full discounted present value of the square kilometer of wetlands in year t, because *SCC* comprises the current and all future *avoided* damages of each ton of sequestered carbon.

The value in equation (4) assumes the lost wetland area is converted into a land cover type with no sequestration function. This is often not the case. Wetlands are converted into forest, open water, farmland, or some other type of use, wherein carbon sequestration would continue, but at a different rate. As discussed previously, for now we assume no remineralization of carbon to the atmosphere following this land cover change. The change, usually a loss, in the *net* annual carbon sequestration value given by:

$$v_t = (\alpha_{SR} - \gamma_{SR})\, SCC_t \qquad (5)$$

where γ_{SR} is the sequestration rate for the non-wetland land cover to which the wetland is converted. i.e. γ_{SR} is a weighted carbon sequestration rate from the total change in land use recorded in the Delaware Bay. Because future land cover changes are uncertain, we use a Monte Carlo analysis to capture the uncertainty. First, we calculate the sequestration rates in areas that converted from tidal wetland in 1975, to some other land cover type in 2011. We estimate sequestration rates in converted lands using a distribution for each land cover type shown in Table 1, Open Water (λ_{OW}), Developed Open (λ_{DO}) etc., to include all values in Table 1. For each land cover type we build a distribution, λ_i, sampling from a triangular distribution or using a point estimate (as shown in Table 1). The distribution γ_{SR} was then estimated from the total set of sequestration rates in converted lands, λ_i, weighted by the area, x_i, of each land cover type, as shown in equation 6.

$$\gamma_{SR} = \sum_{i=1}^{I} x_i\, \lambda_i \qquad (6)$$

The resulting value, γ_{SR}, is approximately a normal distribution with mean 207 $MgC \cdot km^{-2} \cdot yr^{-1}$ and standard deviation 57 $MgC \cdot km^{-2} \cdot yr^{-1}$. Thus, we assume that future wetlands will be lost through conversion to the same mix of land cover types observed from 1975 to 2011. In our case α_{SR} is defined as the rate of sequestration in tidal wetlands, as shown in Table 1. Then, α_{SR} and γ_{SR} are varied in our Monte Carlo simulation to account for variation in sequestration rates and given a range of net values for v_t in equation (5).

If remineralization of carbon were to occur then we would need to add a term to equation 5 that includes the rate of remineralization, discounted by the time at which is occurs. As discussed, the extent of remineralization is largely unknown, so for now we assume that the carbon is stored permanently. Shortly, we will show the potential loss if there were full remineralization.

Now consider the *asset value* for a square kilometer of wetlands, or its annual flow of carbon sequestration services, in perpetuity. The assumption here is that if the wetland is not converted, it would have continued to sequester α_{SR} tons of carbon year-after-year, and, if converted, would have sequestered at a rate of γ_{SR} year-after-year. This yields an annual stream of losses of v_t indefinitely. The lost asset value, V_t, using a discount rate of r then is:

$$V_t = v_t / r \qquad (7a)$$

Note that equation (7a) adds a second level of compounding, beyond that implicit in SCC_t, because it is now accounting for the year-after-year sequestration that would have been carried out by the foregone wetlands. We use discount rates, *r*, that correspond to the discount rates used to estimate the SCC in all our calculations.

Anthoff et al. (2011) argue that SCC_t is increasing at an annual rate (g) of 2.2% due to the cumulative damaging effects of atmospheric carbon. Incorporating the growth rate of SCC_t, we now have the value, in perpetuity, of a square kilometer of converted wetlands given as

$$V_t = v_t / (r - g) \qquad \text{(if r > g)} \tag{7b}$$

Finally, in several of our scenarios, wetland loss is recurring, so new area is lost each year. In these cases, each square kilometer of loss generates a lost asset value of V_t shown in equation 7b. So, a wetland lost today gives a full loss of V_0, and wetland loss next year gives a loss of V_1 discounted by one year because it's lost "work" begins one year in the future. Our expression for the lost value in this case for one square kilometer lost annually is

$$SUM\text{-}V_t = \sum_{t=0}^{T} V_t (1 + r)^{-t} \tag{8a}$$

where T is the limit to the contraction of wetlands or simply the number of square kilometers of wetlands. In scenarios where we gain wetlands through restoration, we use the area (km2) of wetlands in 1778 as the upper limit of area. Where the actual rate of annual change in wetland area is some factor (k) of one square kilometer, the equation becomes

$$SUM\text{-}V_t = \sum_{t=0}^{T/k} k V_t (1 + r)^{-t} \tag{8b}$$

k is positive in restoration scenarios and negative in instances where wetland loss is occurring.

2.6 Issues in mapping

Tidal wetland area estimates from 1778 should be used cautiously when compared with pedigreed USGS topographic maps. Maps of the area at that time often focused on hazards to navigation, with limited attention paid to landward features. The tidal wetland area estimated from the Faden map in 1778 fits with the trend observed in more recent and reliable data sources, however, and so we include this data point as a window into what revolutionary-era tidal wetlands may have looked like in the Delaware Estuary.

USGS topographic maps have improved over the years. From 1900-1918, topographical features were mapped using field surveys, and it wasn't until the 1930's that the USGS began to employ aerial photography in their map production. Early maps potentially obscured the presence of small features such as pools and pans within the marsh that will have been mapped with updated technology. The trend in increasingly refined and resolved data on tidal wetlands, including satellite data in recent years, may lead to sources of bias in our results, as smaller features within the marsh are integrated into our area calculations in more recent years.

3. Results and discussion

3.1 Total area change

We estimate that approximately 76km^2 of coastal wetlands were lost between 1778 and 1918, or 0.54km^2 per year. Between 1918 and 2011, an additional 148km^2 of coastal tidal wetland area was lost in the Delaware Estuary, or 1.6km^2 per year (Figure 3).

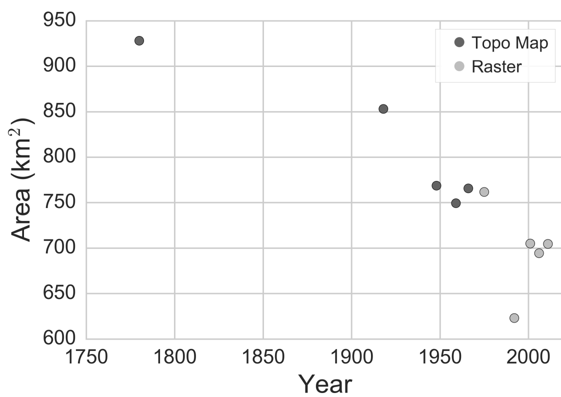

Figure 3: Area of tidal wetlands (in km^2) within HUC10 regions adjacent to the Delaware Estuary. Lightly shaded data points are extracted from NLCD raster datasets. Darker shaded points show hand-delineated data from historical topographic maps.

The relationship between wetland area and year is roughly linear, except for 1992 (Figure 3). The classification scheme for the 1992 NLCD data contained many misclassified cells, confounding attempts to compare results from that year with other years. See Fry et al. (2009) for a complete discussion. For our analyses, 1992 was dropped.

While wetland area increased since 1975 in the lower estuary, partially due to the deposition of dredge spoils along the shoreline, the net change overall in tidal wetland area was negative (Figure 4). Most wetland loss occurred in the Bay portion of the estuary, where extensive shoreline wetlands transitioned to open water, and around Philadelphia, where urban development replaced the wetlands (Figure 5).

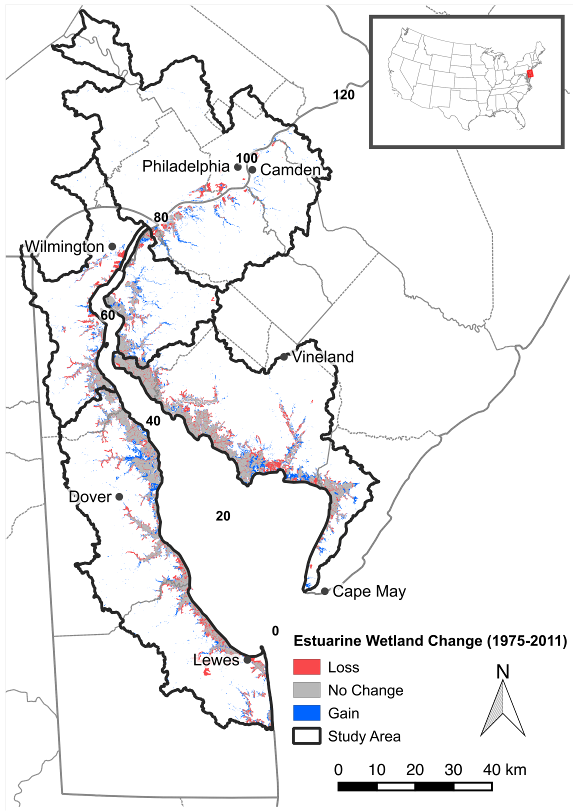

Figure 4: Spatially defined changes in wetland area determined by comparison of NLCD land cover data from 1975 and 2011.

The net loss of tidal wetlands in the Delaware Estuary from 1975-2011 was approximately 1.35km^2 per year, or 0.91 acres per day (Figure 5).

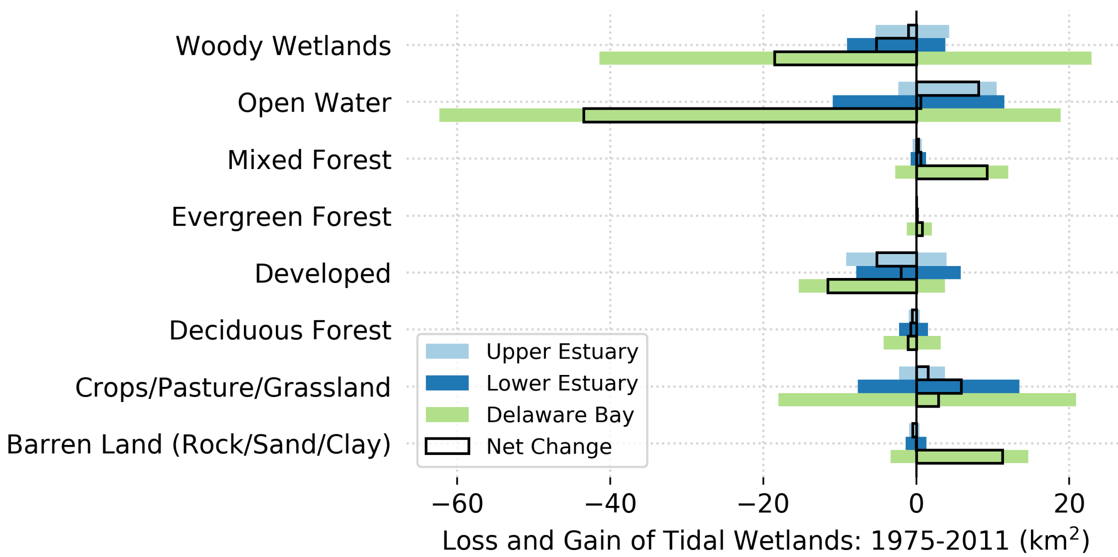

Figure 5: Land cover change between tidal wetlands and other land cover types 1975-2011. Negative values show where tidal wetlands were lost to other land cover types, positive values show where tidal wetlands were gained from other land cover types. Outlined bars show net loss or gain.

3.2 Rates of wetland area change

Table 2 presents our regression results. As noted above, these regressions give estimated loss rates from 1778 to 2011 or 1918 to 2011 depending on the case. First, consider the simple time-trend regressions, comprising models 1, 2, 3, and 5 in Table 2.

The coefficient on *yr(t)* for model 1 (1778-2011) is 1.03 km^2/year, which is consistent with the result in model 2, when controlling for the influence of raster data, *rd(t)*. These results agree with estimates of salt marsh losses locally in Delaware, and nationwide (Dahl, 1990; Tiner et al., 2011). The coefficient on *yr(t)* is significant ($p < 0.01$) in both model 1 and model 2, while the coefficient on raster data is not significant in all models, indicating that data source does not seem to bias results.

The tidal wetland loss rates are similar for both shoreline and inland samples, shown by models 3 and 5. Combining simple time trend results for inland and shoreline wetlands from models 3 and 5, gives an annual rate of wetland loss of 1.53$km^2 \cdot yr^{-1}$ between 1918 and 2011, indicating that the rate of tidal wetland loss may be increasing compared to the 1778-2011 regression.

Sea level rise, *sea(t)*, has a significant ($p < 0.05$), negative, and linear effect on wetland area. For every 1cm rise in sea level, wetland area declines by 1.69km^2. We found no statistical influence of dredge depth, *depth(t)*, on tidal marsh area. The coefficient on population is negative, revealing a decline in wetland area of 0.2km^2 for a population increase of 10,000 people ($p = 0.014$).

3.3 Carbon sequestration values

3.3.1 Present value of the current stock and historic loss of tidal wetlands

Using an SCC of $37.15 and a 3% discount rate, the annual carbon sequestration value, v_t', for a square kilometer of tidal wetlands is $42,000, with a mean asset value, V_t', of $5.20 million (Table 3), assuming a 2.2% annual increase in the SCC. For sensitivity purposes, we provide results for 2.5%, 5% and 3% upper 95% confidence interval discount rates; however, all values referred to hereafter in the text use our central 3% discount rate. For wetlands converted to another land cover type, the mean difference in annual sequestration value, v_t, is $13,000, which has an asset value, V_t, of $1.63 million. Meta-analysis by De Groot et al. (2012) found climate regulation values ranging from $810 to $21,300 per km^2. Our lower-bound estimates overlap with this range, but suggest that the value of carbon sequestration in coastal wetlands may be greater than previously thought. Applying our estimate over the present 704km^2 of tidal wetlands in the estuary, gives a sequestration value of $3.66 billion.

From 1778 to 2011, tidal wetland area decreased from 928 km^2 to 704.5 km^2, a decrease of 223.5 km^2. Since 1918, tidal wetland area declined by 148.5 km^2, and from 2001-2011, 0.43 km^2 of wetlands were converted to other land cover types.

Our central estimate of the future value of wetlands lost for these three increments is \$374, \$249, and \$0.72 million respectively (Table 4).

3.3.2 Present value of future wetland changes

3.3.2.1 Current rate of loss

As an upper bound estimate, if current rates of 1.03 $km^2.yr^{-1}$ of wetland loss persist, wetland area will diminish to zero by the year 2694 CE, and the central estimate for the value of sequestration, $SUM\text{-}V_t'$, lost over that time is \$220.6 million (Table 4).

The mean soil carbon density in Mid-Atlantic wetlands is 0.037 $gC.cm^{-3}$ (Chmura et al., 2003). As an upper bound estimate, assuming an average marsh depth of 1m and immediate 100% re-emission of the carbon stock in marsh soils, an annual loss of 1.03 km^2 of marsh may release 139,700 Mg of CO_2 into the atmosphere, with a social cost of \$5.19 million (3% discount rate). If not completely remineralized, or at a slower rate, this value would need to be discounted by the fraction of remineralization and the year(s) in which remineralization occurred. Regardless, factoring in carbon remineralization would always result in higher social costs for wetland losses.

3.3.2.2 Sea-level rise scenarios

Sea level has risen at a rate of 2.94 $mm.yr^{-1}$ at Philadelphia since 1900 (NOAA, 2016), which translates to 0.497 km^2 of tidal wetlands lost per year (model 3). Assuming conversion to land with carbon sequestration rate γ_{SR}, the lost sequestration value, $SUM\text{-}V_t$, due to observed SLR is \$106.9 million. If SLR causes erosion of wetlands to open water, the lost sequestration value rises to \$330.6 million.

Intergovernmental Panel on Climate Change Assessment Report 5 (Church et al., 2013) models project global mean sea levels to rise at a rate of between 2.0 $mm.yr^{-1}$ (RCP2.6 lower bound) and 15.7 $mm.yr^{-1}$ (RCP8.5 upper bound), which might result in tidal wetland conversions of between 0.338 $km^2.yr^{-1}$ (SLR = 2.0 $mm.yr^{-1}$) and 2.65 $km^2.yr^{-1}$ (SLR = 15.7 $mm.yr^{-1}$). These loss rates translate into between \$73.2 million and \$501 million in lost economic value, respectively. As before, If SLR causes conversion of wetlands to open water, then the lost sequestration value rises to between \$224.8 million and \$1.54 billion.

3.3.2.3 Dredging

The p-value on the dredging coefficient is 0.69. The insignificance of the p-value makes it difficult to draw any inferences from our dredging results.

3.3.2.4 Restoration scenarios

Since 1989, the United States currently has operated under a policy of no net loss of wetlands from human development. Considering wetland restoration at a rate of 0.25 $km^2.yr^{-1}$ (~62 acres per year), wetland area would reach revolutionary-period levels in 894 years, with a net present value, $SUM\text{-}V_t$, of between \$53.7 million (r = 2.5%). As shown in Table 4, at high discount rates, the additional benefits of restoring to revolutionary-era levels represent little gain in net present value over restoring to 1975 levels. Wetland restoration costs are highly variable. Salt marsh restoration estimates from San Francisco Bay put the costs at between $8000 and $18,700 per acre (Zentner et al., 2003). Thus, restoring 0.25 km^2 of salt marsh might cost between $0.5 and $1.16 million per year, with a net present value of between $17.17 million and $39.83 million if restored over 894 years. Therefore, the benefits of restoration, from carbon sequestration alone and not considering other ecosystem services, far outweigh the costs of restoration.

Restoration requires an investment, and thus the net present social benefits of restoration are greater the earlier that restoration occurs, as the benefits of restored ecosystem services accrue to society earlier.

4. Conclusions

The models derived here present an easily transferable approach to valuing changes in the flow of atmospheric carbon into wetland soils following disturbance. These models value the flow of carbon sequestration services provided by tidal wetlands, which could be incorporated into a complete accounting of the natural capital value of these wetlands.

Our findings support previous literature (Kearney et al., 2002; Tiner et al., 2011), and highlight conservation areas showing that losses in wetland area are largely due to land cover change from wetland to open water along the bay shore edges of wetlands and pool formation in the shoreline and inland reaches of the Delaware Bay.

The range in our estimates reflects the high degree of uncertainty in sequestration rates, future land cover change, and most importantly, the effect of discount rate on the social cost of carbon. These remain as significant research needs. Perhaps the most complex research need is how to incorporate remineralization of stored carbon into our results. We provide an upper-bound estimate but research in this area remains highly uncertain.

The value of preserving wetlands far outstrips the value of allowing them to decline and then be restored. Although we value only carbon sequestration, the decision-maker should consider a range of other ecosystem services. Water filtration, storm surge protection, fish nursery habitat, and recreational bird watching, among others, have all been shown to be highly valuable ecosystem services from coastal wetlands.

When weighing the costs and benefits of a land use change project, if the benefits to society of a given project do not exceed the carbon sequestration benefits outlined above, then preserving coastal wetlands likely provides the greatest benefit to society. Of course, additional consideration of other ecosystem services in the general valuation framework remains of paramount importance. The framework and results presented here demonstrate the high value of carbon sequestration in coastal wetlands, and we hope that our approach provides a useful tool to quantify the social benefits of future coastal planning decisions.

5. Acknowledgements

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6. Tables

Table 1: Carbon sequestration rates, broken down by land cover type, used in our analysis

Land Cover	Carbon Sequestration Rate ($MgC \cdot km^{-2} \cdot yr^{-1}$)	Sources
Open Water	8.77	Herrmann et al. (2014)
Developed Open Grassland, Sedge Shrub/Scrub	40.8	Zhu and Reed (2014)
Forest (Evergreen, Deciduous, Mixed)	50, 140, 180	Goulden et al. (1996) Birdsey (1992) Brown et al. (2001) Nowak and Crane (2002)
Pasture/Hay Agricultural Crops	24, 40, 71	Lal et al. (1999) West and Post (2002)
Developed (Low, Medium, High) Barren Land	0	
Woody Wetlands	75, 190, 336	Euliss et al. (2006) Weston et al. (2014) Chmura et al. (2003)
Tidal Wetlands	67, 287, 562	Chmura et al. (2003) Drake et al. (2015) Hussein et al. (2004) Tucker (2016)

Note: Carbon sequestration rate was modeled using a random triangular distribution where three rates are shown; otherwise we use a point estimate. 1 Mg = 1000 kg

Table 2: Time-trend regression model results describing change in tidal wetland area. Standard errors are shown in parentheses.

	Whole Estuary[a]		Shore[b]		Inland[b]	
Variable	1	2	3	4	5	6
Year *yr(t)*	-1.03** (0.12)	-0.95** (0.15)	-0.77** (0.09)		-0.76** (0.16)	
Raster *rd(t)*		-16.33 (20.71)				
Sea Level *sea(t)*				-168.91* (64.28)		
Dredge Depth *depth(t)*				-0.79 (1.86)		
Population *pop(t)*						-0.20* (0.06)
Constant	947.69** (21.63)	941.90** (23.42)	261.48** (5.44)	240.91* (76.09)	574.38** (9.94)	625.81** (27.89)
Adjusted R^2	0.91	0.90	0.92	0.74	0.76	0.61

[a] Whole estuary regression includes data from 1778 to 2011

[b] Shore and inland regressions include data from 1918 to 2011

* Significant at p = 0.05 ** Significant at p = 0.01

Table 3: Estimates for SCC, v_t', v_t, V_t', and V_t. SCC is given in 2015 dollars. Values in parentheses show the standard deviation of our distributions, resulting from variation in carbon sequestration rate.

		Discount Rate (percent)			
		5	3	2.5	3 (Upper Bound)
SCC[a]	Social Cost of Carbon (2015$)	12.77	37.15	59.21	104.48
v_t'	Annual sequestration value per sq. km. (thousand 2015$)	14 (5)	42 (14)	66 (22)	117 (39)
v_t	Annual net sequestration value per sq. km. (thousand 2015 $)	5 (5)	13 (16)	21 (25)	38 (44)
V_t'	Perpetual sequestration value per sq. km. (million 2015$)	0.51 (0.169)	5.20 (1.721)	22.10 (7.313)	14.63 (4.84)
V_t	Perpetual net sequestration value per sq. km. (million 2015$)	0.16 (0.194)	1.63 (- 2.21 - 5.53)	7.12 (8.401)	4.71 (5.56)

[a] SCC values are given as the starting values in 2015, anticipated to grow at rate g = 0.022

Table 4: Mean value of lost wetland area over a range of discount rates in millions of 2015 USD. Values in parentheses show standard deviations

	Discount Rate (percent)			
	5	3	2.5	3 (Upper Bound)
Present Value of Historic Wetland Changes				
Since 1778	37 (43)	374 (442)	1,591 (1,878)	1,053 (1,242)
Since 1918	24 (29)	249 (294)	1,057 (1,247)	700 (825)
Last Decade:	0.071 (0.083)	0.72 (0.85)	3.06 (3.61)	2.03 (2.39)
Present Value of Future Wetland Changes				
Current rate of loss: 1.03 $km^2.yr^{-1}$	6.3 (0.9)	220.6 (16.4)	2,162.2 (112.2)	620.3 (46)
SLR Scenario #1: 2.94 $mm.yr^{-1}$	3.1 (0.4)	106.9 (7.9)	1,187.3 (54.6)	300.8 (22.2)
SLR Scenario IPCC AR5 Lower Bound: 2.0 $mm.yr^{-1}$	2.1 (0.3)	72.7 (5.4)	818.5 (37.2)	204.6 (15.1)
SLR Scenario IPCC AR5 Upper Bound: 15.7 $mm.yr^{-1}$	16.3 (2.2)	498.0 (41.8)	3,473.7 (258.7)	1,400.6 (117.5)
Restore to Revolutionary Levels: 0.25 $km^2.yr^{-1}$	1.5 (0.2)	53.7 (4.0)	562.6 (27.4)	151.2 (11.2)
Restore to 1975 levels: 0.25 $km^2.yr^{-1}$	1.5 (0.2)	45.8 (3.9)	310.0 (24.0)	128.7 (11.1)

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