The role of economic, policy, and ecological factors in estimating the value of carbon stocks in Everglades mangrove forests, South Florida, USA

Abstract

Old growth mangroves in existing protected areas store more carbon than restored forests or plantations. Carbon storage in such forests has economic value independent of additionality, offering opportunities for policy makers to ensure their maintenance, and inclusion in climate change mitigation strategies. Mangrove forests of the Everglades National Park (ENP), South Florida, though protected, face external stressors such as hydrological alterations due to flooding control structures and agriculture impacts and saltwater intrusion as result of increasing sea level rise. Moreover, decreased funding of Everglades' restoration activities following the recent economic crisis (beginning 2008) threatens the restoration of the Greater Everglades including mangrove dominated coastal regions. We evaluated several economic and ecological challenges confronting the economic valuation of total (vegetation plus soil) organic carbon (TOC) storage in the ENP mangroves. Estimated TOC storage for this forested wetland ranges from 70–537 Mg C/ha and is higher than values reported for tropical, boreal, and temperate forests. We calculate the average abatement cost of C specific for ENP mangroves to value the TOC from \$2–\$3.4 billion; estimated unit area values are \$13,859/ha–\$23,728/ha. The valuation of the stored/legacy carbon is based on the: 1) ecogeomorphic attributes, 2) regional socio-economic milieu, and 3) status of the ENP mangroves as a protected area. The assessment of C storage estimates and its economic value can change public perception about how this regulating ecosystem service of ENP mangrove wetlands (144,447 ha) supports human well-being and numerous economic activities. This perception, in turn, can contribute to future policy changes such that the ENP mangroves, the largest mangrove area in the continental USA, can be included as a potential alternative in climate change mitigation strategies.

27 *Keywords:* carbon storage; carbon economic value; the Florida Everglades; mangroves;
28 ecosystem services.

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1. Introduction

Mangrove wetlands provide a wide array of ecosystem services that benefit society including coastal protection, wood for fuel, water quality improvement, and support of fisheries (Nagelkerken et al., 2008; Alongi, 2011; Barbier et al., 2011). Furthermore, mangrove forests have the potential to store carbon in above and belowground biomass and in the soil (Kauffman et al., 2011). Previous studies (Donato et al., 2012, Alongi et al., 2015) indicate that the mean carbon storage in mangrove forests located in the Indo-Pacific region (i.e., 1,023 Mg/ha) could exceed the mean carbon storage attributed to tropical upland, temperate, and boreal forests (200–400 Mg/ha) (Pan et al., 2011).

Carbon storage (i.e., the stock of carbon stored in plant biomass and soil) and sequestration (i.e., the rate of removal of atmospheric carbon per unit of time by plants and soils) (Chapin, 2006) are recognized ecosystem services that contribute to climate change mitigation (Alongi, 2012; Caldeira, 2012; Siikamaki et al., 2012). Deforestation and land-use change account for 8–20% of the total global anthropogenic CO₂ emissions (IPCC, 2007) including the loss of a third of the global mangrove area over the last 50 years (Valiela et al., 2001). Despite occupying only 0.7% of tropical forest area, mangroves deforestation contributes 10% (0.02–0.12 Pg per year) of the global CO₂ emissions from deforestation (Donato et al., 2011; Siikamaki et al., 2012).

One of the mangrove forests least affected by direct deforestation in subtropical latitudes is the Everglades Mangrove Ecotone Region (EMER) in southern Florida, USA (Rivera-Monroy et al., 2011a). Despite being located in a protected area, the EMER is exposed to several external stressors (e.g., hurricanes, freshwater diversion). Hydrological changes in the past 100 years

have altered drainage patterns and reduced fresh water flows into Everglades estuaries by more than 50% (Smith et al., 1989), affecting nutrient and salinity gradients regulating mangrove structure and productivity (Castañeda-Moya et al., 2013). Although relative sea level has increased by approximately 0.3m over the last century in South Florida (Ross et al., 2009; Zhang et al., 2012), it is seawater intrusion into the main regional aquifer along the Everglades coastline that has altered the spatial distribution and diversity of plant communities, including mangrove forests (Saha et al., 2012).

The Everglades is undergoing a large spatial-scale restoration as part of the Comprehensive Everglades Restoration Plan (CERP) which is expected to increase the quantity, quality, and timing of fresh water throughout the mangrove ecotone, thus potentially altering its productivity and spatial distribution (Rivera-Monroy et al., 2011b; Sklar et al., 2001). CERP is also expected to enhance other significant ecological and economic benefits arising from a range of ecosystem services such as groundwater purification, real estate, park visitation, open space and fishing (McCormick et al., 2010). Yet, the recent economic downturn (beginning in 2008) along with policy changes has led to a significant decrease in funding to continue restoration activities (> 10 billion US dollars) as initially planned (Sklar et al., 2005), threatening the restoration of the Everglades including the EMER. Assessing the quantity and economic value of the legacy C stored in the EMER, which has not yet been investigated, can (a) send a strong signal to concerned policy makers of the importance of long-term allocation of CERP funds for ENP preservation, (b) establish a baseline for future comparisons to evaluate restoration goals, and (c) present opportunities for U.S.A. to include the value of the C stored in ENP mangroves in long-term climate change mitigation strategies, thus fulfilling its commitment to the 2015 Paris Agreement under the United Nations Framework Convention for Climate Change. Carbon

storage in protected areas has an economic value independent of additionality (Zarate-Barrera and Maldonado, 2015), offering opportunities for policy makers to ensure their maintenance, and inclusion in climate change mitigation strategies. Strengthening of these stocks within existing protected areas, especially in old growth forests such as the ENP mangroves, can be an important part of a country's strategy to combat climate change (Campbell et al., 2008; Alongi, 2011).

Valuation of ecosystem carbon storage faces several challenges. First, there exists a large uncertainty regarding accurate estimations of C storage, particularly in coastal wetlands because of the dynamic nature of C cycling in coastal environments where hydrological conditions regulate the exchange of organic and inorganic carbon between land and adjacent coastal waters (Bouillon et al., 2008, Hopkinson et al., 2012; Rivera-Monroy et al., 2013). Further, these areas are increasingly threatened by human impacts since 10 per cent of the world's population and 13 per cent of the world's urban population live in the coastal zone (McGranahan et al., 2007; Hopkinson et al., 2012). The variability and uncertainty in mangrove primary production and C accumulation through space and time greatly influence the economic valuation of carbon storage (Alongi, 2011). Indeed, C storage in wetland ecosystems, including mangrove forests, could vary between short (month, years) and long (decadal, centennial) temporal scales. This temporal variability is reflected in the role of mangroves as C sinks or sources as a result of dynamic changes in net carbon fluxes that are influenced by the geomorphic setting interacting with latitudinal temperature and precipitation gradients (Barr et al., 2010; Barr et al., 2012; Barr et al., 2013; Fuentes and Barr, 2015; Ho et al., 2014; Maher et al., 2013; Troxler et al., 2015; Rovai et al., 2015). Although the number of studies assessing net C fluxes in mangrove wetlands is limited (e.g., Maher et al., 2013), some studies indicate that mangrove forests could become a net C sink on decadal and centennial timescales if not significantly affected by deforestation or

major changes in land use (Alongi, 2001, Kauffman et al., 2014; Ezcurra et al., 2016). In the case of the ENP, its protected nature (i.e., National Park) has the potential to lock the stored organic C longer into the biological pool, in contrast to carbon in an ecosystem that is under direct and compounding human threats.

Second, the economic valuation of C varies with valuation methodologies (Hessen et al., 2004; Stockmann et al., 2013), not all of which are suitable for valuing the carbon in a particular ecosystem. Carbon prices are also influenced by several technological, regulatory, economic and social factors, and vary across countries and markets. As C sequestration rate for EMER is not currently available and is uncertain (Troxler et al., 2013, Rivera-Monroy et al., 2013), our goal is to value the legacy carbon stored in the ENP mangrove biomass and soil. We therefore investigated different C valuation methodologies to appropriately value the C storage in the Everglades mangroves. Moreover, global estimates of mangrove C storage or estimates derived from tropical mangroves cannot be directly transferred to the Everglades region because of its distinct geographical location, ecogeomorphic characteristics, and the nature of the pressures (i.e., sea level rise) and pulses (e.g., hurricanes) influencing the functional properties of mangroves in the Everglades (Collins et al., 2011, Castañeda-Moya et al., 2010).

While recognizing the above challenges, the purpose of our study is to conduct a first economic valuation of total (vegetation plus soil) organic carbon (TOC) storage in the mangrove forests of the Everglades National Park (ENP). To achieve this goal, we first estimated the physical quantity of stored TOC, and then performed an economic valuation of this ecosystem service. Although a number of mangrove carbon stock assessments are now available for neotropical regions (e.g., Kauffman et al., 2014; Hamilton and Lovette, 2015), there is a lack of information establishing the physical and economic baseline of C values to strengthen emerging

carbon mitigation initiatives, particularly given the wide difference in the degree of carbon storage magnitude (above and belowground) provided by different mangrove ecotypes (Ewel et al., 1998, Twilley and Rivera-Monroy, 2009, Rovai et al., 2015). We expect the economic value of the stored organic carbon in mangroves per unit area to be significantly higher than other types of forests. Presently, despite the fact that the Everglades has the largest area of mangroves in the continental USA (1,445 km²) (Simard et al., 2006), there are no regional estimates of the TOC (aboveground, belowground, and soil carbon) stored and its associated economic value (Rivera-Monroy et al., 2013, Troxler et al., 2013). We expect that the estimate of TOC in the Everglades mangroves and its economic valuation will encourage similar quantifications in other mangrove wetlands.

As a protected area, some of the major threats to the ENP and its ecosystem services come from external stressors located outside the park boundaries (e.g., change in the hydrological regimes upstream; Sklar, 2005). Managing such external factors is often a complex process and requires a large scale regional assessment study. For instance, as Friess et al. (2015) note, designing a Payment for Environmental Services (PES) mechanism to handle external factors needs to carefully consider a series of complex steps—from the evaluation of onsite large-scale impacts of stressors to willingness on the part of external stakeholders to take mitigation action and/or to pay for such actions. Evaluating all those steps is beyond the scope of this paper. Instead, our methodology focuses on the first logical step of estimating and valuing the stored onsite carbon. Such valuation will provide a baseline of stored C (in physical and monetary terms), serving as a benchmark for evaluating the impacts of restoration programs that might occur over a long period of time. Because of the protected nature of our study site and year to year variability and uncertainty in C sequestration (Alongi, 2011), it may neither be necessary

or feasible to implement a PES system for legacy carbon. PES systems have often proved more appropriate for services that are measurable in short intervals (e.g., C sequestration instead of C storage) and with less uncertainty (Alongi, 2011).

2. Methodology

2.1. Study Area: Everglades Mangrove Wetlands

Mangrove forests in South Florida are located within the protected areas of Everglades National Park (ENP), Florida (Figure 1). The EMER lies on a carbonate platform between the freshwater marshes of the Everglades and the marine waters of the Gulf of Mexico and Florida Bay (Chen and Twilley, 1999a). Two major drainage basins, the Shark River Slough (SRS) and Taylor Slough (TS) discharge water through the western and eastern regions of the Everglades, respectively (Saha et al., 2012; Blanco et al., 2013). In contrast to other nutrient rich estuaries, the EMER mangroves are influenced by a highly oligotrophic and phosphorus-limited coastal area (Noe et al., 2001).

Multigradient patterns of resources, regulators, and hydroperiod control mangrove community structure and ecosystem function in the region (Castañeda-Moya et al., 2013). The tall (~18 m) riverine mangroves along SRS store the highest aboveground biomass values, particularly close to the mouth of the estuary. The high aboveground productivity of the SRS mangroves is attributed to the dominant tidal hydrology and higher phosphorus availability in the soil (Chen and Twilley, 1999b; Castañeda-Moya et al., 2010). The mangrove forests throughout TS have lower aboveground productivity and are dominated by scrub mangroves (< 1.5 m) with biomass < 50 Mg/ha (Coronado-Molina et al., 2004; Simard et al., 2006; Castañeda-Moya et al., 2013). These scrub forests are permanently flooded throughout the year with negligible tidal

frequency (Castañeda-Moya et al., 2011; Rivera-Monroy et al., 2011a). A significant interaction of hydrological and fertility gradients results in distinct mangrove ecotypes (i.e., riverine vs. scrub) with differences in the allocation of aboveground and belowground biomass across spatial scales throughout ENP-EMER (Castañeda-Moya et al., 2013).

2.2. Total Organic Carbon in the EMER

The total organic carbon (hereafter referred to as TOC and/or C) present in the Everglades mangroves was estimated using data and methodologies from previous studies of tree height spatial distribution and associated biomass in both the Everglades and other tropical regions (Simard et al., 2006; Simard et al., 2008; Rivera-Monroy et al., 2013). The stored TOC included three main C pools: (1) aboveground biomass (C_{AGR}), (2) belowground live roots (C_{BGR}), and (3) peat soils (C_{BGS}).

2.2.1 Aboveground TOC (C_{AGR})

We used landscape level tree height data to determine total mangrove biomass in the EMER. A detailed description of the methods to estimate height can be found in Simard et al. (2006). Briefly, tree height was estimated using Shuttle Radar Topography Mission (SRTM) (February 2000) and Light Detection and Ranging (LiDAR) data. The SRTM data was calibrated using airborne LiDAR data and a high resolution USGS digital elevation model (Desmond 2003). Space Shuttle Endeavour was equipped with two radar antennas separated by a 60 m boom (baseline) operating at C-band and X-band. Airborne LiDAR data were collected using an Optech ALTM 1233 system operated by Florida International University on 15–16 April and 14–16 May 2004. With a flight altitude of 500 m above the ground, flight speed of 210 km/hr, field of view angle of 40°, and pulse rate of 33 kHz, a 360 meter-wide swath of 13 cm wide laser

footprints spaced approximately every 1.5 m beneath the flight path was produced; for the location of LiDAR survey transects see Simard et al. (2006). For SRTM calibration purposes, the elevation data was filtered with a 31x 31 mean filter (i.e., 30 m x 30 m area) and co-registered the LiDAR data with the SRTM map with nearest neighbor interpolation. The Root Mean Square Error (RMSE) of the LiDAR versus SRTM-L regressions was 1.8 m and a filter window of 0.9 ha (5 x 5 pixels) was used to preserve sharp tree height edges that are characteristic of the mangrove forests (Simard et al., 2006).

Forest stand-level structural properties including tree height, basal area, and diameter at breast height (DBH) were collected in several sites using the variable plot method across 22 sampling points. A total of 289 trees were sampled over approximately 2.4 km. Moreover, plot data in scrub mangrove forests in Biscayne Bay and Taylor Slough were incorporated in the dataset to include biomass estimates across the entire range of mangrove canopy heights in the EMER. Initial landscape level estimates of tree height were converted to biomass values using published species-specific allometric equations for neotropical mangroves (Fromard et al., 1998; Ross et al., 2000) and later updated using regional species-specific allometric equations obtained in 2005-2006 (Smith and Whelan, 2006). For specific values of statistical variance and error propagation in above- and belowground biomass estimates see original sources (Simard et al., 2006; Castañeda-Moya et al., 2011, 2013). Aboveground biomass was then converted to carbon values based on plant carbon content of ~ 44% (Ewe et al., 2006). This conversion factor is an average value ($44\% \pm 2$ SE) of all species-specific tree components (e.g., tree trunk, leaves, roots, etc.) across all mangrove sites in our study region (Castañeda-Moya et al., 2013), and is similar to estimates in a wide range of mangrove forest types and latitudes (Alongi, 2012).

2.2.2 Belowground Carbon in Roots (C_{BGR})

Castañeda-Moya et al. (2011) estimated total (0–0.9 m depth) root biomass (g m^{-2} ; <20 mm in diameter) for the six FCE-LTER sites (SRS-4, 5, and 6 and TS/Ph-6, 7 and 8) (Table 1). Overall, a total of 229 soil cores were collected across all six mangrove sites to estimate root biomass. Root cores (0–90 cm) were collected at each site using a PVC coring device (10 cm diameter x 45 cm length). Cores were sectioned into 0–45 and 45–90 cm depths and processed individually in the laboratory. Root cores were rinsed with water through a 1-mm synthetic mesh to remove soil particles, and live roots were separated by hand picking and sorted into size classes (< 2 mm, 2–5 mm, 5–20 mm). Each root sample was oven-dried at 60°C to a constant mass, and weighted. As part of the Florida Coastal Everglades Long Term Ecological Program (FCE-LTER, <http://fce.lternet.edu/>) mangrove studies have been implemented since 2001 on these six study sites encompassing the Taylor (TS/Ph-6, 7, 8) and Shark River sloughs (SRS-4, 5, 6) (Figure 1). Similar to the aboveground biomass carbon estimates, belowground biomass was converted to carbon using the plant carbon content of ~ 44% (Table 1).

2.2.3 Soil Carbon (C_{BGS}) and Estimation of Total Organic Carbon

The soil total (inorganic + organic fractions) carbon (TC) values to a depth of 0.45 m for the six FCE mangroves sites were reported by Castañeda-Moya et al. (2013). In that study, four soil cores were collected at each site using a PVC suction-coring device (15 cm diameter x 45 cm length; Meriwether et al., 1996). Cores were extruded and sectioned into 0–2 cm intervals using a custom-made incremental soil core extruding apparatus. Soil sections were oven-dried at 60°C to a constant mass and weighted to estimate soil bulk density. Samples were ground with a Wiley Mill (250 μm mesh screen) and duplicate subsamples were analyzed for TC concentrations using an ECS 410 Elemental Analyzer (Costech Analytical Technologies, Inc.,

Valencia, CA). To determine the TOC fraction for all study sites, a concentrated (12M HCL)
acid fumigation procedure was applied to all soil samples to remove the inorganic (carbonate)
fraction (Harris et al., 2001). Soil TOC values were then extrapolated to values per unit area
(Table 1). Soil and peat formation is strongly linked to mangrove production in the Everglades
(Chen and Twilley, 1999a) where peat deposits vary in thickness from 0.5 to 6.5 m across the
landscape (Ewe et al., 2006). In the Shark River region, pollen analysis in a 5.25 m depth soil
core at the mouth of this estuary (4 km from SRS-6 site) indicates that the underlying peat started
accumulating about 5250 cal yr BP (Yao et al., 2015). Mangrove vegetation in this region started
to appear from 3,800 cal yr BP and became dominant by 1150 cal yr BP (at approximately 1.4 m
depth). Prior to mangrove establishment, the vegetation community in the study area was
apparently dominated by upland taxa and marsh vegetation (~5700 to 3800 cal yr BP). In
contrast, mangroves in the Taylor Slough region are younger forests and more recently
established by colonizing this region. For instance, mangroves in TS/Ph-6 have encroached
inland approximately 1.5 km from Florida Bay during the past 50 years due to reductions in
freshwater drainage into this region along with a gradual increase in sea level (Ross et al., 2000).
As a result, the mangrove peat overlying marl at this site is less than 0.3 m depth, in contrast to
peat depths of about 0.5 to 1 m at TS/Ph-7 & 8 (Ewe et al., 2006). Thus, because of the large
spatial variation in peat depth and difference in age of soil formation between regions, and
current level of certainty in C origin (upland/freshwater/brackish vegetation vs. mangrove), we
estimated soil TOC content down to 0.9 m depth in the Shark River region and down to 0.45 m
in the Taylor River region to provide a conservative estimate of landscape level mangrove
organic C storage in the EMER (Table 1).

We estimated the TOC mean and median values for above (C_{AG}) and belowground root biomass (C_{BGR}) and soil organic carbon (C_{BGS}) using all data sets available for each study site (Table 1). We summed both the above- and belowground C and soil C pool (limited to 0.9 m for Shark River and to 0.45 m for Taylor River region) and derived a median carbon storage value for the ENP-EMER mangroves (Table 1, $335.6 \pm 88.1 \text{ MgC ha}^{-1}$) to account for intrinsic variability due to sample size and as performed in other studies (e.g., Alongi et al., 2015, Murdiyarso et al., 2015). We also assessed the uncertainty of the median value by calculating its standard error considering a non-normal, skewed distribution of C values for each site and calculated the standard error of the mean value for each C source using all sites ($n=6$) (Taylor, 1997; Chave et al., 2004) (Table 1).

3. Selection and Development of Carbon Prices

Upon analysis of different valuation methodologies, we performed a comparison of C prices that included estimates based on social costs of carbon (SCC), marginal abatement costs (MACs), and market prices. All values were converted to the common unit of 2015 U.S. dollar per ton of C.

Social cost of carbon, also referred to as marginal damage cost, reflects what society is theoretically willing to pay to avoid the damages caused by increased C emissions (Price et al., 2007). We derived an estimate for the SCC by using social cost of CO_2 estimates published by the U.S. Government Interagency Working Group for regulatory analysis of projects that reduce emissions (Interagency Working Group on Social Cost of Carbon, United States Government, 2013). We used the SCC of CO_2 for the year 2015 (\$36/t CO_2 in 2007\$), inflated it to 2015 USD

(\$46/tCO₂) by applying an annual inflation rate of 3%, and thereafter calculated the social cost of carbon (\$167/tC) (conversion factor: 1tC = 3.67 tCO₂).

Marginal abatement cost (MAC) indicates the cost of maintaining and/or reducing C emissions or loss from the storage (Ramirez et al., 2002). To derive an abatement cost of mangrove C relevant to the Everglades, we used the cost associated with restoration of the historic sheet flow of water across the Everglades by the Comprehensive Everglades Restoration Plan (CERP) (Carter, 2002). The primary goal of CERP is to capture and store freshwater that is currently discharged into the ocean and use 80% of that freshwater for the preservation of the natural habitat and species. The initial cost estimated by the South Florida Ecosystem Restoration Task Force was \$7.8 billion with an annual cost of \$182 million (Carter, 2002). We converted the 1998 costs to 2015 USD (initial cost: \$12.8 billion, annual cost: \$300 million) by compounding at 3% annual rate. As only 80% of these costs were allocated for the preservation of the natural habitat (Carter, 2002), initial 2015 costs amounted to \$10.3 billion and annual costs were \$240 million. The present value of the annual costs over the project's lifetime was estimated as \$8 billion (at 3% discount). The total cost of preserving the natural habitat in Everglades, calculated as the sum of the initial and annual costs, was \$18.3 billion. As the mangrove forests (144,447 ha) encompass 14.86% of the total Everglades area (971,660 ha) covered under CERP, the cost of preserving the Everglades mangroves was calculated as \$2.72 billion. Thus, we estimated the average abatement cost of the mangrove C in Everglades as \$56/tC (Table 2).

We selected the Regional Greenhouse Gas Initiative (RGGI), a market-based regulatory program in the U.S.A. to reduce greenhouse gas emissions, to represent the market price for C for our study area. To estimate the market value for C, we used RGGI's 2014 average auction

clearing price (\$4.72/tCO₂) (Potomac Enterprises, 2015), converted it to 2015\$ (\$4.86/tCO₂), and then calculated the market price of C (\$18/tC) (conversion factor: 1tC = 3.67 tCO₂). For comparison, we also estimated the C value by using the 2014 international voluntary market price of \$3.80/tCO₂ (Forest Trends Ecosystem Marketplace, 2015), and converted it to 2015 price of \$14.36/tC. The 2014 year was dominated by the Reduction of Emission by Deforestation and Degradation (REDD+) market, which sells carbon offsets based on avoided deforestation.

Market prices are generally lower in value as consumers participating in C markets are currently not in a position or are not willing to pay a full price required to supply the benefits of C storage or sequestration. Estimates for SCC and MACs are derived using quantitative economic models that combine scientific knowledge on climate change with socio-economic aspects of economic growth under possible climate change scenarios (Ding et al., 2010). The presence of uncertainties of key parameters in the climate change models and the type of model used to estimate C prices contribute significantly to the uncertainty of model outputs. Nevertheless, the average abatement cost of mangrove C calculated in our study (\$56/tC) is based on cost estimates of an existing functional project (CERP) and not on future projects. Additionally, the above MAC-based price estimated using the CERP-related costs is more conservative than the SCC-based price.

4. Results and Discussion

4.1 Carbon Storage

There were significant differences in aboveground biomass (Castañeda-Moya et al., 2013) and carbon storage ($p < 0.005$, Table 1) between Shark and Taylor rivers because of the differences in forest structure characterized by different mangrove ecotypes (riverine vs. scrub),

with aboveground C values 12.4 times higher in Shark River sites relative to Taylor sites (Table 1). In contrast, belowground root C values were relatively similar between these two basins ranging from 10.6 to 20.6 Mg C ha⁻¹ across all sites. However, when comparing total vegetation (aboveground + belowground roots) C storage between basins, Shark River allocated 77.4% of the carbon to aboveground (i.e., trunk), while scrub mangroves in Taylor River stored more C belowground (78.5%) relative to aboveground (21.5%). This differential allocation in carbon between above-and belowground components across sites and basins is also reflected in the root:shoot ratios, with lower ratios in the Shark River sites (range: 0.2 to 0.5) compared to the Taylor River sites (range: 1.9 to 9.7). The mean total (vegetation + soil) ecosystem TOC was 2.8 times higher in Shark River (473 Mg C ha⁻¹) relative to Taylor River (169 Mg C ha⁻¹), with the lowest TOC allocation at TS/Ph-6 (Table 1). The vegetation C pool accounted for 15% of the total ecosystem C storage for both basins, whereas the soil C pool represented the largest fraction of C storage in each site ranging from 77–90%. These results show that riverine mangroves along Shark River store more C relative to scrub mangrove forests, which are P-limited and influenced by long flooded hydroperiods restricting growth and development (Castañeda-Moya et al. 2013). Our results also support the findings from other studies (e.g., Donato et al., 2011; Kauffman et al., 2014, Adame et al., 2013) indicating that regardless of the dominant mangrove ecotype (riverine vs. scrub), the majority of carbon is stored in the soil. The higher C allocation in the western region (Shark River) of Everglades National Park is associated with the presence of deep soil peat layers compared to the eastern region where shallow peat deposits and scrub mangrove forests dominate the Everglades landscape (Simard et al., 2006; Castaneda-Moya et al., 2013; Yao et al., 2015).

4.2 Comparison of C storage with other forests

Current estimates of carbon stock density in tropical, boreal, and temperate forests are 242, 239, and 155 Mg C/ha, respectively (Pan et al., 2011). In the case of mangrove forests, a number of studies have shown variable values depending on the geomorphic setting, but overall mean C stocks are significantly higher than those for terrestrial forests. For example, Donato et al. (2012) estimated an average total (aboveground + belowground + soil) carbon storage of 1,023 Mg C/ha at variable soil depth in the Indo-Pacific region in forest stands classified as oceanic and estuarine mangroves. Similar values were obtained in mangrove forests from the tropical Pacific (Micronesia, Republic of Palau: 830–1218 Mg C ha⁻¹) (Donato et al., 2012). Our median C storage estimates for the EMER indicate that mangrove forests in this region store 335.6 ± 88.1 Mg C/ha (Table 1) (range: 0.45–0.9 m soil depth). We compared our values to those obtained in other locations in the Americas including the Caribbean region (Sian Kaan) (Adame et al., 2013), the Pacific coast of Mexico (Bahia Magdalena, Baja California; Ezcurra et al., 2016) and the Dominican Republic (Kauffman et al., 2014). Carbon stocks in the Sian Ka'an Biosphere Reserve, a karstic-dominated environment similar to the ENP, were significantly different in tall (10–14 m: 127.0–145.6 MgC/ha), medium (5–11m: 80.8–82.7 MgC/ha), and scrub (0.1–2 m: 4.9–11.7 MgC/ha) mangrove forests. C stocks in mangrove wetlands in Bahía Magdalena, a non-topographically constrained lagoon on a flat coastal plain under a semi-arid climate were < 150 MgC/ha, which were obtained at a range of 0.9–1.6 m soil depth. The highest values reported by Ezcurra et al. (2016) were obtained in tropical latitudes where riverine mangroves were widely variable and ranged from 100 to 800 MgC/ha (La Encrucijada, Mexico). Mangrove wetland carbon stock estimates (~1 m depth) obtained in coastal regions in the Dominican Republic also varied depending on forest stand stature; the highest C stock (580 ± 40

MgC/ha) was measured in tall mangrove forests (>10 m) and the lowest (320 ± 210 MgC/ha) in dense stands of scrub forests <3 m, which showed a high variability in C stock values (Kauffman et al., 2014). These neotropical sites, including our study area, are strongly impacted by tropical storms, which can influence total aboveground biomass, tree stature and productivity (Farfán et al., Smith et al., 2009).

About 4% percent of the EMER carbon storage value is partitioned in the aboveground biomass and 96% is belowground (roots + soil). This belowground value is comparable to estimates from other mangrove areas in the Indo West Pacific (IWP, old world) and Atlantic East Pacific (AEP) (Neotropics). For instance, in the IWP mangroves soil C ranges between 71–98% of the total value in estuarine mangrove sites and 49–90% in oceanic sites (Donato et al., 2012). In the AEP region, values are similar, ranging from 65 to 99% (Adame et al., 2013, Kauffman et al., 2014). In comparison, tropical forests store 56% C in biomass and 32% in soil, while boreal forests store 20% in biomass and 60% in soil (Pan et al., 2011). These results underscore the significant contribution of soil C storage in these forested wetlands worldwide compared to other terrestrial biomes.

4.3 Economic Criteria for Valuation of Mangroves

The estimated values for C storage in ENP mangroves are \$4,819/ha, \$6,041/ha, and \$56,045/ha based on the international voluntary market (\$14.36/tC), RGGI (\$18/tC), and the social cost of carbon (\$167/tC) derived by the US. Interagency Working Group 2013 estimate, respectively. The average abatement cost based on CERP costs (\$56/tC), yields a value of \$18,794/ha for the C stored in ENP (Table 2). Given the variability in C prices (as seen in Section 3.2), we developed a simple set of general economic criteria for valuing ecosystem C,

based on the type of carbon (stored or sequestered), the robustness of C markets, and the stage at which a country's policy cycle is at a given time. These criteria can act as a guide for setting the appropriate C price to value mangrove forests in general and those in South Florida in particular.

The methodology for the economic valuation of stored carbon differs from that for sequestered carbon (Ramirez et al., 2002). The social cost of carbon, which reflects the economic value of the damage associated with incremental increase in C emissions (Price et al., 2007), would be appropriate for valuing periodic C sequestration (Ramirez et al., 2002). However, our goal in this study is to value the legacy carbon stored in the Everglades biomass and soil at the centennial scale. Marginal abatement costs represent the cost of preserving the existing mangrove forest and the associated stored carbon. Since the primary goal of CERP is to preserve and maintain the natural habitat in the Everglades, including mangroves, by restoring the historic flow of freshwater, we consider the average abatement cost derived from CERP costs (\$56/tC) as the most appropriate for carbon valuation in this study.

Carbon markets and REDD+ schemes consider C sequestration, and not storage, for pricing ecosystem carbon (Alongi, 2011; Zarate-Barrera and Maldonado, 2015). Moreover, the low C market price in current U.S. markets (RGGI: \$18/tC) reflects the disparity between what people are actually paying and what people might be willing to pay. The current price (\$14.36/tC) of carbon in the international markets like REDD+ is even lower. Carbon markets do not include C storage in protected areas as yet (Zarate-Barrera and Maldonado, 2015), and until they do so, the current market prices may be unsuitable for policy development. However, these tentative/preliminary prices could at best provide a benchmark for stable and robust C prices. Thus, our study uses the average abatement cost of carbon (\$56/tC) for the valuation of C stored in ENP (Table 2).

4.4 Ecological Criteria for Valuation of Mangroves

In this section, we offer general recommendations for ecological factors to be considered when setting the price for C storage (as in this study) and sequestration in mangroves.

(1) Forest age. A mature and intact mangrove forest, such as the EMER, has more interconnectivity and functional redundancy, and provides a variety of ecosystem services in comparison to younger, single-species plantations (Alongi, 2012; Rivera-Monroy et al., 2013). For mature, older forests like the EMER, where periodic sequestrations are uncertain and their estimates are limited (e.g., Troxler et al., 2013, Rivera-Monroy et al., 2013), C valuation may include C stored over a long period of time (decades, centuries). As the C sequestration rate specific for ENP mangroves is not available in current literature, we only valued its C storage. It is critical not to use market prices alone, the framework of payment for ecosystem services (Alongi, 2011), or social costs of carbon (Ramirez et al., 2002) to value the C storage in such mature forests. In our study, we therefore used the abatement cost of C based on CERP costs.

(2) Variability in ecosystem functions and services. Carbon valuation must take into account not only the distinct geomorphic settings (e.g., delta vs. karstic) where mangroves develop, but also the influence of landscape level disturbances when monitoring C storage in the long-term. Both mangrove sequestration capabilities and limits are non-linear and vary significantly with time and space, especially in coastal regions subject to major natural (e.g., tropical storms) and human disturbances (Ewel et al., 1998; Twilley and Rivera-Monroy, 2009).

(3) The status of mangrove forests as a protected area. Although protected areas are primarily designated for biodiversity conservation, they are also recognized as an alternative to regulate climate through C storage (Campbell et al., 2008), independent of additionality. If the

regulations are properly enforced, forests in protected areas could become significant sinks of carbon. The valuation of C sequestration and storage services in protected areas depends on the magnitude of C stored, the level of management and enforcement, the amount of resource extraction/use permitted, and the governance and land-use change pressures that determine the capability of the system to store carbon (Campbell et al., 2008). For example, mangroves in the EMER are a unique case in comparison to other mangrove forests in the Americas since they have not been influenced by direct (human settlements) or indirect extractive uses (e.g., wood, silviculture), or significant eutrophication impacts during the last 150 years.

The carbon storage in ENP mangroves is high and compares to other mangrove forests in developing countries in the Americas (Adame et al., 2013; Suárez-Abelenda et al., 2014; Duque-Estrada et al., 2015). Mangrove-dominated coastal regions are under constant pressure derived from human activities despite their designation as protected areas and as part of a network of national parks system or under the assignments of RAMSAR sites or Biosphere Reserves (Rivera-Monroy et al., 2004). To compound the problem, it is estimated that of the total global mangrove area estimated using Landsat imagery (137,760 km²) in 2000, only 6.9% is protected under the Convention on Biodiversity (Giri et al., 2011). In the case of mangroves in the Americas, protected areas that lie in more restrictive International Union for Conservation of Nature (IUCN) management category (e.g., categories I–II) are more effective in reducing deforestation (Clark et al., 2008). Yet, protected areas also face net loss when deforestation pressures are avoided in the core zone because large spatial scale impacts are transferred into the park boundaries as result of changes generally occurring in adjacent, unprotected areas that are hydrological and functionally connected to the protected/park area (Rivera-Monroy et al., 2004).

4.5 Economic Valuation of the Carbon Stored in EMER

Considering both the ecological nature of the study region and the economic criteria for valuation, we use the average cost of abatement derived from CERP costs (\$56/tC) as the most appropriate to value the Everglades mangrove C since this cost indicates the cost of preserving the stored carbon within these forests. SCC-based valuation is more appropriate for C sequestration (Ramirez et al., 2002) and not so much for C storage in the Everglades where mangrove wetlands are protected, and deforestation and the attendant carbon loss are not an immediate threat to the ecosystem. Until robust carbon markets that consider C stored in protected areas evolve and the resultant carbon market prices represent a balance between what the society would be willing to pay and what it costs to supply C, the current market prices may not be appropriate for valuation of stored carbon in EMER. Moreover, the main purpose of the valuation in this study is *not* to make an economic and ecological justification for receiving substantial market-based payments. As long as the CERP policy focuses on preventing the loss of legacy C, the abatement cost would be a more appropriate valuation tool. Accordingly, the value of the C stored in EMER is \$18,794/ha or \$2.7 billion (Table 2). This value reflects the potential loss of C in monetary terms if society fails to prevent the degradation of mangroves because of external threats in the future.

One should exercise caution when comparing the above per-hectare C value with those reported in other studies because of the variation on the basis of pricing. Also, there is a wide range of assumptions and environmental processes (e.g., geomorphic, hydrological) underlying C estimation in studies reported in section 4.2. For the sake of illustration, we multiply our abatement cost price of \$56/tC with the C estimates of other studies to derive the following results. Based on the Pan et al. (2011) C storage estimate, the value of C stock in tropical, boreal,

and temperate forests would be \$13,552/ha, \$13,384/ha, and \$8,680/ha, respectively. Using Donato et al. (2012) estimate of C storage of 1,023 Mg C/ha in the Indo-Pacific oceanic and estuarine mangroves, the C value is \$57,288/ha. Similarly, the C stock in mangrove forest with a medium tree height (5–11m tall, (e.g., Sian Ka'an Biosphere Reserve) would be worth \$4,525/ha to \$4,631/ha; whereas in the mangrove wetlands in the Dominican Republic (Kauffman et al., 2014), the C value would be \$32,480/ha and \$17,920/ha for tall mangrove forests and scrub forests, respectively. Clearly, the above values would be lower if we base our valuation on the international voluntary market price or regulatory market price.

5. Conclusion

The aim of our study was to evaluate C pricing methodology and develop an appropriate economic value of aboveground and belowground C pools in mangrove wetlands of the Florida Everglades. The estimation of clear, quantifiable carbon stored in the Everglades mangrove ecosystem can have several implications. Our carbon storage estimates may be considered as a benchmark for establishing the costs of inaction associated with the failure to prevent such losses in C stock. In addition, mangrove restoration programs will be increasingly needed as a consequence of human and natural impacts in mangroves outside the ENP. In such cases, estimates for the value of C storage in ENP mangroves can be based on MACs or more specifically, the average abatement cost estimated in this paper (\$56/tC) to perform the cost-benefit analysis of mangrove restoration.

Finally, we highlight limitations of the study. The C values estimated in this paper have a wide range for two main reasons. First, the estimation of the C storage itself is influenced by significant variances in the C stocks of different forest components. There is a large spatial

gradient in mangrove biomass distribution that reflects the hydrological and nutrient fertility heterogeneity, and thus of the C stocks across the landscape (Childers et al., 2006, Rivera-Monroy et al., 2011a, Castañeda-Moya et al., 2013). Still, we consider our already conservative values a first-rate approximation and a robust baseline that could be improved upon as more long-term sampling points are added across the region in the future to improve the C value spatial resolution. Second, there is no single ideal economic value for carbon, which varies widely depending on the political and economic environment of the natural resource in question. Therefore, the values estimated in this study must be used with caution and in reference to appropriate contexts. Further, carbon storage is only one of the several ecosystem services that the EMER mangroves provide. Our estimated economic value of carbon storage is just a component of the total economic value of the mangrove ecosystem. Future studies on the valuation of other ecosystem services in the region can help in determining a spatially explicit comprehensive value.

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Tables

Table 1. Mean (± 1 SE) above-(C_{AG}) and below-ground carbon (C_{BGR}) and soil carbon (C_{BGS}) storages in mangrove sites throughout the Everglades Mangrove Ecotone Region (EMER). Soil stocks are estimated to a depth of 0.9 m except where noted (i.e., Taylor River sites).

FCE-LTER Study Site	Dominant Species	Aboveground Carbon ^a (MgC/ha) (C _{AG})	Belowground Root Carbon ^c (MgC/ha) (0.9 m) (C _{BGR})	Soil Organic Carbon (MgC/ha) ^d (0.9 m) (C _{BGS})	Total Organic Carbon (MgC/ha)
SRS-4	<i>Rhizophora</i> <i>mangle</i>	47.5 $\pm$ 1.5*	14.1 $\pm$ 1.8 (n=5)	373.8 $\pm$ 23.8 (n=4)	435.4*
SRS-5	<i>Rhizophora</i> <i>mangle</i>	42.0 $\pm$ 0.3*	19.3 $\pm$ 1.4 (n=5)	475.7 $\pm$ 14.2 (n=4)	537*
SRS-6	<i>Laguncularia</i> <i>racemosa</i>	71.4 $\pm$ 0.9*	11.1 $\pm$ 1.9 (n=5)	365.6 $\pm$ 7.7 (n=4)	448.1*
TS/Ph-6	<i>Rhizophora</i> <i>mangle</i>	5.5 ^b	10.6 $\pm$ 1.5 (n=10)	53.5 $\pm$ 6.2 (n=4)	69.6
TS/Ph-7	<i>Rhizophora</i> <i>mangle</i>	5.5 ^b	20.6 $\pm$ 3.3 (n=7)	224.8 $\pm$ 13.9 (n=4)	250.9
TS/Ph-8	<i>Rhizophora</i> <i>mangle</i>	1.97 $\pm$ 0.2	19.2 $\pm$ 3.2 (n=5)	163.5 $\pm$ 5.1 (n=3)	184.7
<i>Median</i>		23.8	16.7	295.2	335.6 $\pm$ 88.1
<i>Mean $\pm$ SE</i>		29.8 $\pm$ 11.8 (n=6)	15.8 $\pm$ 1.8 (n=6)	276.1 $\pm$ 63.8 (n=6)	321 $\pm$ 73.6

^aCastañeda-Moya et al. (2013);

^bCoronado-Molina et al. 2004, values for TS/Ph-6 and TS/Ph-7 do not have standard error in original source;

^cCastaneda-Moya et al 2011: a total of 229 ingrowth cores were analyzed for belowground root carbon estimation since it considers repeated measurements; in this table, we include only the total N value representing the number of sampling points at two depths (0-0.45 m and 0.45-0.90 m);

^dCastañeda-Moya et al. 2013; Soil Organic C at the Shark sites and TS/Ph-6, TS/Ph-7, and TS/Ph-8 were measured at 0.45m; values included in the table were extrapolated to 0.9 m depth for the Shark River sites only- see Methods section;

*ANOVA significant differences between Shark River and Taylor:

Aboveground Carbon: Shark River = 53.5 ± 9.0 (SE), Taylor River = 4.3 ± 1.2 (SE). Total Organic Carbon (MS = 3647.1, F ratio = 29.3, P >F, 0.005)

Total Organic Carbon: Shark River = 473.5 ± 31.9 (SE), Taylor River = 168.4 ± 52.9 (SE). Total Organic Carbon (MS =139629, F ratio = 24.3, P >F, 0.007)

Table 2. Economic Valuation of C Storage in the Mangrove Forest of the Everglades National Park, Florida (in 2015 US\$)

	Total Organic Carbon in ENP Mangroves (MgC/ha)	Average Abatement Cost for ENP mangroves (\$/tC ^a)	Economic Valuation of C stored in ENP mangroves per ha (\$/ha)	Economic Valuation of total C storage in ENP mangrove forests ^b (\$)
Median - SE Value	247.5		13,860	2,002,035,420
Median Value	335.6	56	18,794	2,714,679,139
Median + SE Value	423.7		23,727	3,427,322,858

Notes:

^a1tC = 1 metric ton Carbon = 1 MgC;

^bENP mangrove forest area: 144,447 ha.

Total Organic Carbon (median value) in ENP mangroves: 335.6 MgC/ha * 144,447 ha = 48,476,413 MgC.

The median economic value for ENP mangroves based on:

SCC (\$167/tC) = \$56,045/ha.

RGGI market price (\$18/tC) = \$6,041/ha.

International voluntary market price (\$14.36/tC) = \$4,819/ha.

Figure

Figure 1: Map of South Florida showing spatial distribution and location of the Taylor and Shark River sloughs in the Everglades National Park, Florida (Source: FCE LTER, <http://fcelter.fiu.edu/>)