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Estimates of above-ground tree carbon after projected land-use and land cover change in a subtropical urban watershed

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Abstract
This study investigates the effect of land-use and land cover (LULC) change on above-ground tree carbon (AGTC) in a subbasin of the Tampa Bay Watershed, Florida. LULC change was integrated with AGTC to project future quantities under three landscape scenarios: baseline, increased and aggressive rates of development. A 12% increase in total landscape AGTC occurred from 2006 to 2011 as agriculture and rangeland were converted to residential, infrastructure and built classes. Scenario projections for 2016 show an additional increase of 11% AGTC under baseline change, 15% under increased development and 18% under aggressive development. These results suggest that residential expansion may cause an increase in AGTC storage as agriculture and rangeland areas are replaced. However, as agriculture and rangeland disappear, LULC change patterns could shift, with residential expansion replacing upland and wetland forested areas causing a long-term decrease in AGTC. These results show that biomass of the tree component of AGTC can be significant, in and of itself, for urban classes and provide insights into its role in AGTC dynamics for these systems. This information can also help decision-makers identify areas as potential carbon sources or sinks.

Keywords Landscape · Land-use change · Land cover change · Urban ecosystems · Ecosystem services · Carbon storage

Introduction

Land-use and land cover change (LULC) causes dramatic increases in deforestation, pollution, habitat destruction and a variety of other environmental problems (Brundtland 1988). This is particularly true in urban areas, which dramatically impact preceding land-uses and land covers. LULC change is also a major contributor to global carbon emissions. It is estimated that during the period 1850–1990 LULC change contributed 33% of total carbon (C) emissions (Houghton 1999, 2003). In the tropics, which are currently experiencing accelerated LULC change, land clearing causes a decrease of 120 Mg C ha^{-1} yr^{-1} vs. 68.3 Mg C ha^{-1} yr^{-1} in the subtropics and 62.9 Mg C ha^{-1} yr^{-1} for temperate regions per year (West et al. 2010). Houghton (2010) points out that while most modeled estimates of C emissions from land-use change point to an upward trend, the variability between estimates is increasing as modeled LULC change is compared to actual change over time. Houghton further states that spatial variability of biomass is not well represented in the literature and is a current limitation in studies relying on these estimates. This highlights the importance of understanding the variation of C storage across the landscape, which for urban ecosystems is apparent in their heterogeneity and fragmentation of land-uses and land covers (Zipperer et al. 2012).

Urban carbon mapping is often excluded from assessments of ecosystems services (ES) because it is poorly understood (Nowak et al. 2013). McPherson et al. (2013) believe this is because urban land is a relatively small proportion of total land area. But current research shows that urban trees contribute up to 14% of C sequestered by all forests and store approximately 7.69 kg C m^{-2} of tree cover, totaling 643 million Mg of C across the US (Heath et al. 2011; Nowak et al. 2013). The United Nation's World Urbanization Prospects report indicates that 54% of the global population and 82% of the North American population reside in urban areas (United

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Nations Department of Economic and Social Affairs 2014). This has led to a greater interest in cities as important ES providers.

A number of studies have shown that urban areas vary significantly in C storage and sequestration, largely dependent on population density, area size and urban forest structure (Escobedo et al. 2010; Strohbach and Haase 2012; Davies et al. 2013). Yet, research suggests that patterns of the distribution and quantity of ES along the urban-to-rural gradient are not consistent between cities. Larondelle and Haase (2013) examined seven ES indicators, including C storage, for LULC classes in Berlin, Helsinki, Salzburg and Stockholm. They found significant heterogeneity in the provisioning of ES and varying patterns of quantities between urban cores and their surrounding regions. This suggests that urban areas do not share a common theory regarding the allocation of ES and should be studied individually to measure the patterns and consequences of LULC change.

Research has been done to describe C storage and sequestration rates of urban forests in peninsular Florida. For instance, the City of Orlando's urban trees cover 22.1% of the city area, contain a total of 640,043 Mg C and sequester 32,237 Mg C yr^{-1} (Empke et al. 2012). Miami-Dade county has a 12% canopy cover, with storage estimates of 1,497,676 Mg C and sequester 564,500 Mg C yr^{-1} across 127,300 ha (Escobedo et al. 2010, 2011). Gainesville's urban tree cover is 51% and was estimated to store 662,648 Mg C in 12,174 ha (Escobedo et al. 2018). Escobedo et al. (2010) concluded that urban forest structure is an important indicator of storage and sequestration potential in Florida.

Objectives

This study extends current research by not only estimating above-ground tree carbon storage (AGTC) per LULC class in a subbasin of the Tampa Bay Watershed (TBW) but also examines LULC impact on the quantity and distribution of AGTC using an ES-centric LULC classification (Lagrosa et al. 2018). Here, future estimates of AGTC as a result of modeled LULC change are derived for three potential landscape scenarios. Results are given for each LULC class in addition to the entire subbasin.

Methods

Study area, LULC and ES data

The study area was a 796-km^2 (79,600 ha) subbasin of the Tampa Bay Watershed that encompasses the entire City of Tampa including its immediate adjacent and surrounding suburban and rural areas (Fig. 1). The climate is classified as subtropical with a mean annual temperature of 23.3 °C and a mean annual precipitation of 127 cm. Over 4.6 million people live in the watershed.

Plot Data

AGTC was estimated using permanent-plot inventory data. To collect inventory data, the study area was divided into a hexagon grid of 1.77 km^2. Within each hexagon, a random sample point was chosen for permanent plot location. A total of 531, 405 m^2 plots were geo-referenced and inventoried. On each plot, diameter at breast height (DBH), tree height, crown width, height-to-base of live crown, percent of crown missing, crown transparency, and percent crown dieback were recorded by species for all woody stems ≥2.5 cm DBH. Definitions of measured attributes and descriptions of analyses can be found in Nowak et al. (2008). In addition, because these are permanent plots, the distance and azimuth from plot center were recorded for each individual tree measured. Woody stems <2.5 cm DBH were excluded from the study due to the difficulty and inaccuracy of obtaining carbon estimates for trees of that size. Plots were sampled in 2006 and resampled in 2011. Any plots that were not resampled were removed from the analyses.

Above ground total carbon (AGTC) was calculated using i-Tree Eco, a suite of models to calculate ecosystem services for urban forests (Nowak et al. 2008, 2013). In the model, biomass for each measured tree is calculated using allometric equations and conversion factors from the literature to estimate whole tree dry-weight biomass and carbon (Nowak et al. 2008). If no allometric equation was identified for an individual species, the average of results from equations of the same genus was used. If no genus allometric equations were found, the average of all broadleaf or conifer equations was used. Carbon was then aggregated at the plot level and averaged by LULC class. The five-year slope for each plot was calculated by assuming a simple linear trend and then averaged by class to derive a trend estimate for 2006–2011. Negative plot values were interpreted as zero. Of the original 531 plots, 409 were used in the analyses.

Initial landscape estimates

To maintain consistency across the study area, we used LULC classes and areas from the Southwest Florida Water Management District (SWFMWD) (Southwest Florida Water Management District 2015). The initial LULC classes were redefined by applying the AGTC classification derived in Lagrosa et al. (2018) to the SWFWMD maps within the subbasin boundaries. Areas were calculated for both 2006 and 2011 using SWFWMD maps from each year to match the time period of the plot data. Nine broad classes were used in the analyses—agriculture, built, forested wetlands, infrastructure, non-forested and mangrove wetlands, rangelands, residential, upland forests and water (Table 1).

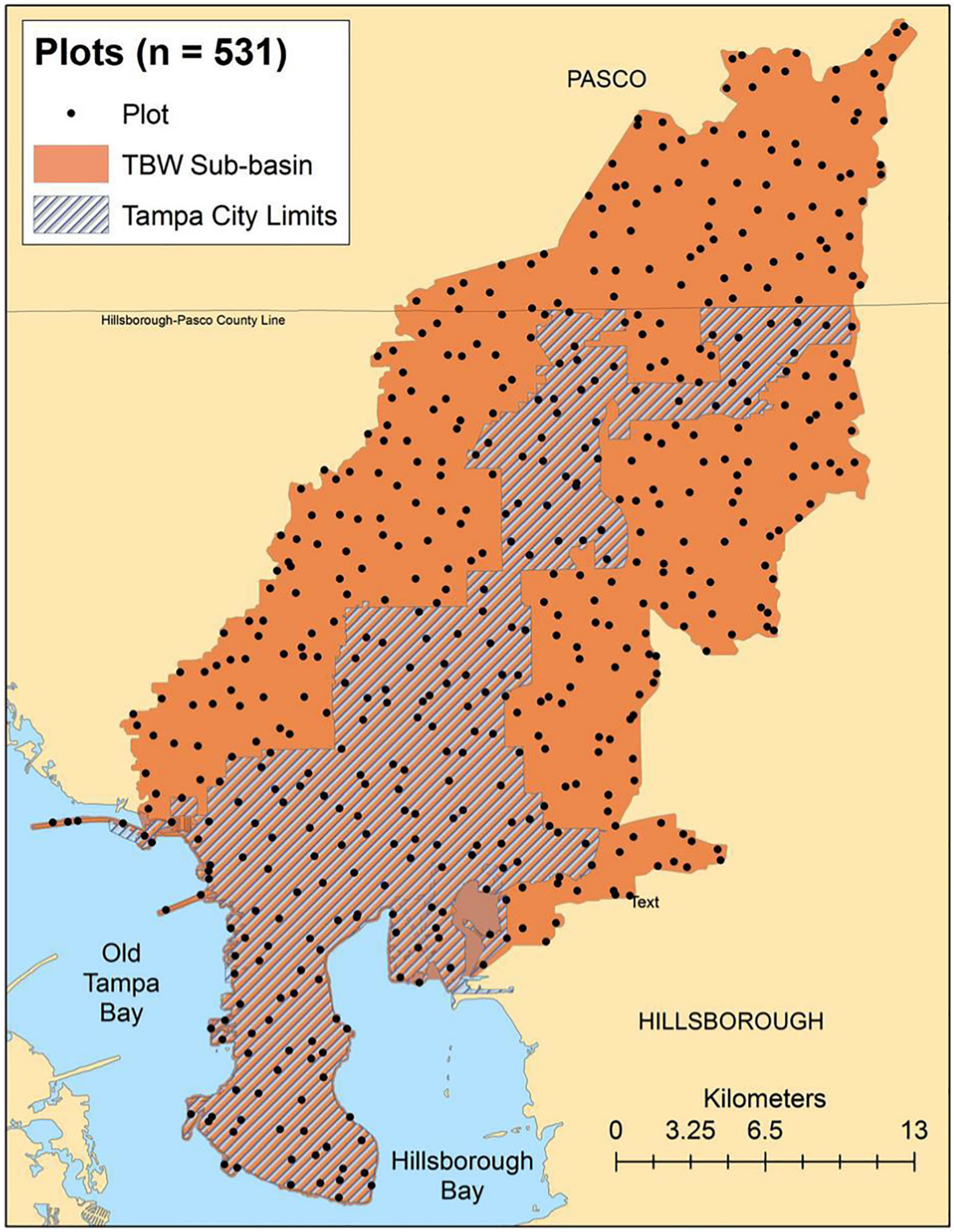

Fig. 1 The study area subbasin located inside the greater Tampa Bay Watershed showing sample plot locations within and around the City of Tampa, Florida

Landscape quantities of AGTC were estimated for 2006 and 2011 by scaling mean plot AGTC estimates with corresponding class areas. These values were then used to establish linear rates of change (slopes) over the five-year period. AGTC estimates, aggregated across the sub-basin, were produced for both 2006 and 2011.

Scenario-based AGTC

LULC change was modeled in the subbasin for three scenarios to simulate baseline, increased and aggressive rates of urban development. Dyna-CLUE (Verburg 2010) was used to model each scenario for the years 2012–2016 as parameterized in Lagrosa et al. (2018). The Conversion of Land-use and its Effects (CLUE) is an LULC change framework that has been used in national, regional, and small-scale studies to project landscape change. The framework uses a hierarchical approach based on systems theory that combines total area analyses with those for individual spatial units. The model integrates top-down allocations with bottom-up dynamics of local drivers. The top-down approach uses an estimate of aggregated land requirements to simulate external, large-scale determinants of LULC change, while the bottom-up approach refers to specific transition characteristics and driving factors of the local area (Verburg and Overmars 2009). A detailed article on the parameterization of the model as used in this study can be

Table 1 Southwest Florida Water Management District land use classification, associated land cover (adapted from Southwest Florida Water Management District (2015)) and number of inventoried plots per classification used in an analysis of a Tampa Bay watershed subbasin

Land use classification	Corresponding land cover	Number of plots
Agriculture	Cropland, pastures, tree crops, nurseries and vineyards	12
Built, non-residential	Commercial and services, industrial, institutional, and recreational	104
Forested wetlands	Bottomland hardwoods, Cypress mounds, wetland forest mixes	63
Infrastructure	Transportation, communication, and utilities	28
Non-forested and mangrove wetlands	Mangrove swamps, freshwater marshes, saltwater marshes, wet prairies, and emergent aquatic vegetation, open lands	21
Rangeland	Shrub and brushlands, mixed rangeland	6
Residential	Single and multiple housing	141
Upland forests	Pine flatwoods, upland coniferous forests, upland hardwood, tree plantations, upland hardwood/coniferous mix	17
Water	Streams and waterways, lakes, reservoirs, bays and estuaries	17

found in Lagrosa et al. (2018). For each model run, a different matrix of land-use requirements and transition sequences were applied. The elasticity and driving factors remained constant. Elasticity is an economic term that refers to an entity's ability (or inability) to respond to an external phenomenon and potentially change to a different class. Driving factors provide information specific to each land-use class. Typical metrics include population density, housing density, median income as well as biophysical attributes such as soil and hydrological characteristics. Driving factors were used to develop a probability equation for change for each LULC class (Lagrosa et al. 2018). After each simulation, the area of each LULC class was calculated on an annual basis. Slopes from the plot AGTC estimates were used to scale AGTC values to the landscape-level by LULC class.

The baseline scenario was modeled with class growth rates from the most recent five-years of available LULC data (2007–2011). The second scenario applied increased rates of change reflecting the period 1999–2004 which coincided with a population growth of nearly 100,000 annually (overall rate of 8.8%) (United States Census Bureau 2000, 2010). During this time the area saw a significant reduction in agriculture and rangeland classes with a corresponding increase in residential and infrastructure. The third scenario introduces a hypothetical situation where aggressive residential and infrastructure development reduces the agriculture and rangeland classes to zero by 2016.

Results

2006/2011 AGTC estimates

Analysis of plot AGTC data showed distinct patterns. Forested wetlands had the highest mean plot AGTC (Table 2) for both time periods (97.59 Mg ha^{-1} and 103.82 Mg ha^{-1} respectively). Upland forests (36.42 Mg ha^{-1} in 2006 and 41.87 Mg ha^{-1} in 2011) had the second highest mean value, whereas the lowest value was measured for infrastructure (1.56 Mg ha^{-1}) in 2006 and agriculture (0.35 Mg ha^{-1}) in 2011. At the plot level, eight of the nine classes had at least one plot with zero AGTC (Table 2). Only upland forests had a minimum value greater than zero with 0.07 Mg ha^{-1} in 2006 and 0.42 Mg ha^{-1} in 2011. Forested wetlands also had the single highest AGTC by plot at 414.62 Mg ha^{-1} in 2006 and 336.70 Mg ha^{-1} in 2011. Residential plots had the second highest maximum AGTC with a high of 235.12 Mg ha^{-1} in 2006 and 246.53 Mg ha^{-1} in 2011. Standard deviations for seven of the nine plots exceeded mean AGTC values. Only forested wetlands and upland forests had means greater than one standard deviation. This was consistent for both time periods with means less than two standard deviations for 2006 and 2011.

Apart from the agriculture and water classes, all seven of the other classes exhibited the same relative ranking in terms of the minimum, maximum, mean and standard deviation of AGTC between the two time periods. For example, rangeland was ranked sixth for maximum AGTC in both 2006 and 2011. Similarly, residential had the second highest standard deviation in 2006 and 2011.

Change between 2006 and 2011

Change metrics from 2006 to 2011 were calculated for plots in each LULC class including means and standard deviations of change in AGTC (Tables 3 and 4). Forested wetlands had the highest mean absolute positive change of all classes with 6.22 Mg ha^{-1} followed by upland forests with 5.48 Mg ha^{-1}. Infrastructure had the least absolute positive change with 0.07 Mg ha^{-1}. Agriculture had the only absolute negative change (−5.55 Mg ha^{-1}). Also listed are the proportions of plots that had positive change and the proportion of plots that had negative change. Except for agriculture, LULC classes showed greater proportion of positive than negative change. The final column shows the proportion of plots that had zero AGTC in both time periods. Reflecting land use, agriculture and water had the greatest number of plots with zero AGTC

Table 2 Range, mean and standard deviation (Mg/ha) of above ground total carbon (AGTC) for each LULC class for 2006 and 2011 in Tampa Bay watershed subbasin

	Class	Min AGTC (Mg/ha)	Max AGTC (Mg/ha)	Mean AGTC (Mg/ha)	SD AGTC (Mg/ha)
2006	Agriculture	0.00	54.27	5.90	15.78
	Built, non-industrial	0.00	140.14	11.41	23.90
	Forested wetlands	0.00	414.62	97.60	67.58
	Infrastructure	0.00	18.57	1.56	4.20
	Non-forested and mangrove wetlands	0.00	100.29	15.78	26.10
	Rangeland	0.00	25.48	4.48	10.15
	Residential	0.00	235.12	31.04	40.44
	Upland forests	0.07	85.55	36.42	28.59
	Water	0.00	23.58	1.60	5.73
2011	Agriculture	0.00	4.12	0.35	1.19
	Built, non-industrial	0.00	145.10	13.73	26.67
	Forested wetlands	0.00	336.70	103.82	58.17
	Infrastructure	0.00	19.88		
	Non-forested and mangrove wetlands	0.00	108.69	20.22	29.97
	Rangeland	0.00	29.31	6.72	11.58
	Residential	0.00	246.53	35.28	45.23
	Upland forests	0.42	104.27	41.87	30.34
	Water	0.00	30.39	3.01	7.93

(75% and 82%, respectively, Table 3). By comparison, forested wetlands and upland forests had the least number of plots without AGTC (2% and 0%, respectively).

Positive and negative minimum and maximum change values for each respective plot were calculated to illustrate some of the variation within each class (Table 4). For example, forest wetlands had the largest, maximum positive (99.12 Mg ha^{-1}) and negative change (−111.87 Mg ha^{-1}) of all the land use classes. Mean AGTC (Mg ha-1) were plotted to show the linear trend of change in AGTC from 2006 to 2011 by LULC (Fig. 2). Except for agriculture and water, infrastructure had the smallest and forested wetlands had the largest mean annual changes (0.02 and 1.25 Mg ha^{-1}, respectively) (Table 4).

Subbasin AGTC

Land-use area projects from 2011 to 2016 showed a dynamic landscape for the different scenarios (Table 5). From 2011 to 2016 agriculture was reduced 19% for the baseline (44.9 to 37.6 km^2) and scenario 1 (44.9 to 34.0 km2) and a 100% decline for scenario 2 (as modeled). Rangeland and upland forests also showed a similar decline from 2011 to 2016 for baseline and scenario 1. Rangeland decreased from 19.9 to 16.4 km^2 (21%) for the baseline and to 14.0 km^2 (21%) for scenario 1. Interestingly, the model projected rangeland to decline to 0 km^2 for scenario 2, whereas upland forests was unchanged for scenario 2. Forested wetlands were basically unchanged for baseline and scenario 2 but increased 5% (11.7

Table 3 Means (Mg ha^{-1}) and standard deviations (Mg ha^{-1}) of AGTC change for plots in LULC classes of a Tampa Bay watershed subbasin. Also shown are the percentage of plots in each class with an increase, decrease, or contained no AGTC from 2006 to 2011

LULC Class	Mean Δ AGTC	SD Δ AGTC	% + Δ	% -Δ	% 0 AGTC
Agriculture	−5.55	15.90	8%	17%	75%
Built, non-residential	2.32	10.10	47%	12%	41%
Forested wetlands	6.22	28.19	75%	24%	2%
Infrastructure	0.07	3.90	29%	11%	61%
Non-forested & mangrove wetlands	4.44	6.96	65%	20%	15%
Rangeland	1.85	2.69	50%	0%	50%
Residential	4.25	16.32	69%	18%	13%
Upland forests	5.48	8.49	94%	6%	0%
Water	1.41	3.60	18%	0%	82%

Table 4 Minimum and maximum change in AGTC (Mg ha^{-1}) and mean annual change in AGTC (Mg ha^{-1} yr^{-1}) by respective plots in each LULC class between 2006 and 2011 for a Tampa Bay watershed subbasin

LULC Class	Min + Δ AGTC	Max +Δ AGTC	Min -Δ AGTC	Max -Δ AGTC	Mean ΔC Mg ha^{-1} yr^{-1}
Agriculture	1.92	1.92	−14.41	−54.85	0.00
Built, non-residential	0.01	59.55	−0.01	−51.81	0.46
Forested wetlands	0.34	99.12	−0.20	−111.89	1.25
Infrastructure	0.05	9.03	−0.05	−17.43	0.02
Non-forested & mangrove wetlands	0.09	19.17	−0.01	−3.66	0.89
Rangeland	0.84	3.82	–	–	0.37
Residential	0.00	75.70	−0.02	−63.64	0.85
Upland forests	0.35	23.95	−15.73	−15.73	1.09
Water	3.74	13.42	–	–	0.00

to 12.3 km2) under scenario 1. As expected, residential, infrastructure and built (non-residential) increased under all scenarios. The expansion of residential and infrastructure occurred principally in the northern portion of the subbasin with

Fig. 2 Trend lines for plot AGTC (Mg ha-1) for all LULC classes for 2006–2011

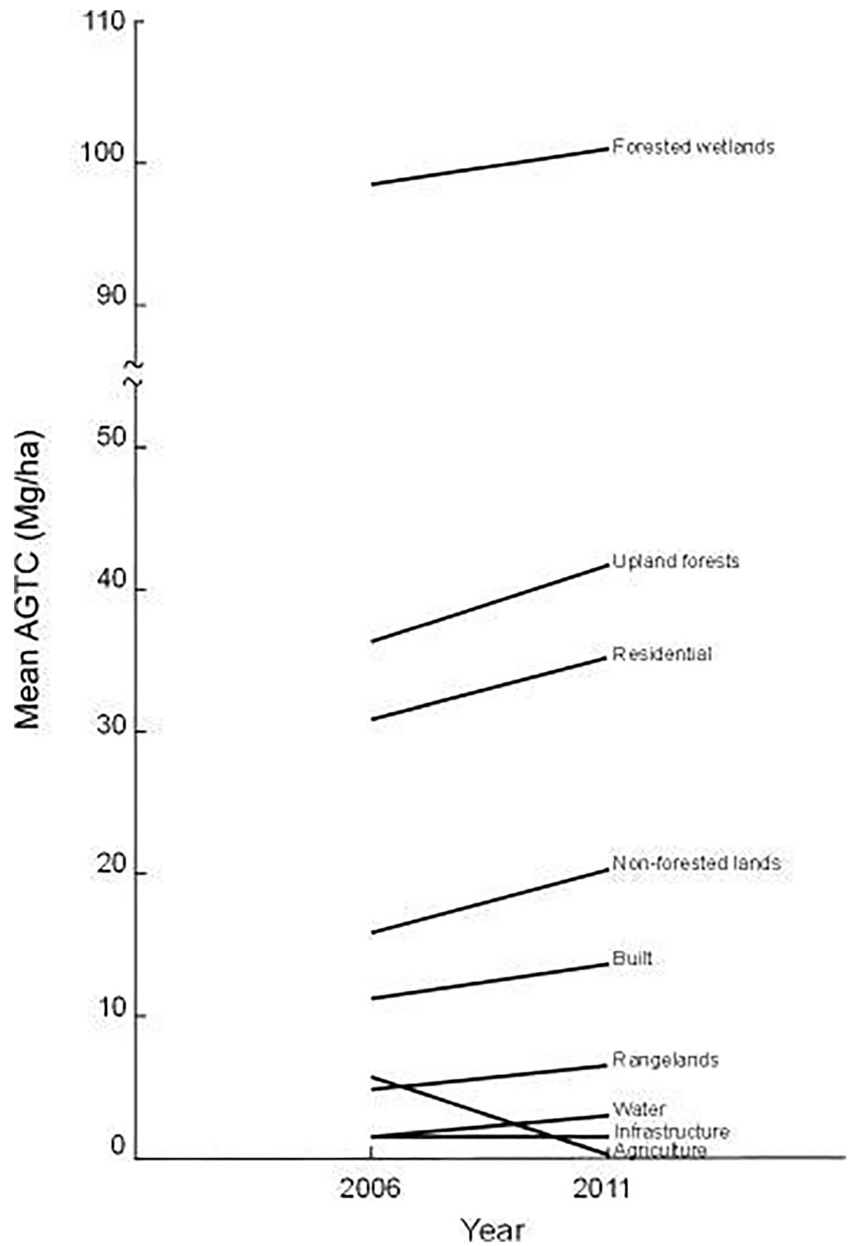

Table 5 Observed and projected class areas (km^2), by year, as a result of LULC change scenarios in the Tampa Bay watershed subbasin

Class	2006	2011	2016 Base	2016 Incr.	2016 Aggr.
Agriculture	52.8	44.9	37.6	34.0	0.0
Built, non-residential	172.0	171.4	175.2	178.0	181.0
Forested wetlands	116.6	116.6	116.3	123.0	117.0
Infrastructure	44.3	47.9	51.2	52.0	57.0
Non-forested & mangrove wetlands	37.9	39.6	38.8	39.0	40.0
Rangeland	23.4	19.9	16.4	14.0	0.0
Residential	253.9	261.7	273.3	283.0	307.0
Upland forests	45.9	44.1	39.4	37.0	44.0
Water	50.0	50.8	48.7	50.0	50.0

losses of agriculture and rangeland (Figs. 3, 4, 5, 6, and 7). Originally, this portion of the subbasin had the least area of grey infrastructure.

Forested wetlands had the highest amount of AGTC with 1,138,313 Mg in 2006, 1,210,788 Mg in 2011 and 1,279,398 Mg in 2016 under baseline conditions (Table 6, Fig. 8). This represents between 43% and 48% of total subbasin AGTC across the time scale of the study. In addition, the model projected an increase in AGTC stored in forested wetlands of 11.8% or an additional 142,897 Mg for scenario 1 and 7.5% (90,864 Mg) for scenario 2. Residential showed the second highest amount of AGTC 788,253 Mg in 2006, 923,632 Mg in 2011 and 1,080,734 Mg in 2016 under baseline conditions (Table 6, Fig. 8). This represents approximately 34% of the total subbasin AGTC. With respect to the different scenarios, baseline and scenario 1 are similar in AGTC, but with scenario 2 there is an increase of approximately 4% of total AGTC for the subbasin.

In addition to agriculture, rangeland was the only other land-use class to be zeroed out in scenario 2, corresponding to a loss of only 0.4% of the total AGTC for the subbasin. The least amount of total AGTC in a land-use class was infrastructure with approximately 0.3%.

Subbasin estimates showed that in 2006 there was a total of 2,367,921 Mg AGTC, which grew to 2,655,707 Mg by 2011. Subbasin-level estimates under LULC scenarios projected 2,946,225 Mg for baseline conditions and 3,050,493 MG for scenario 1. Increased rates similar to those that occurred from 1999 to 2004 caused an accumulation of 104,268 Mg (20,853 Mg yr^{-1}) for a total of 3,050,493 Mg. AGTC under aggressive development, in which agriculture and rangeland disappeared, was projected to increase to 3,122,643 Mg for scenario 2 (Table 6). Overall, subbasin-level results indicated that from 2006 to 2011, the conversion of agriculture and rangeland to residential, infrastructure, and built (non-residential) resulted in a projected 12% increase in total

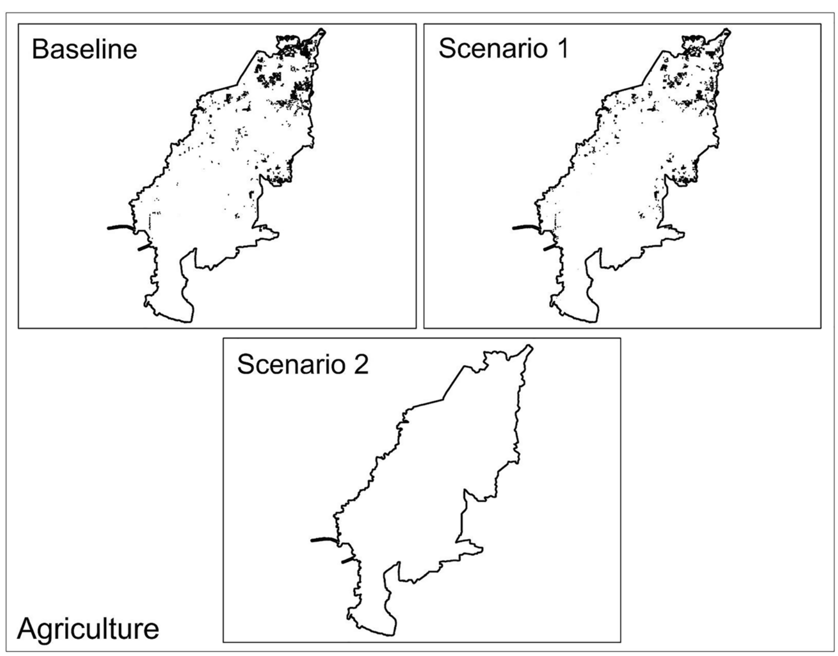

Fig. 3 Modeled LULC change for 2016 under baseline rates, increased rates (Scenario 1), and aggressive rates (Scenario 2) of urban development for the agriculture class

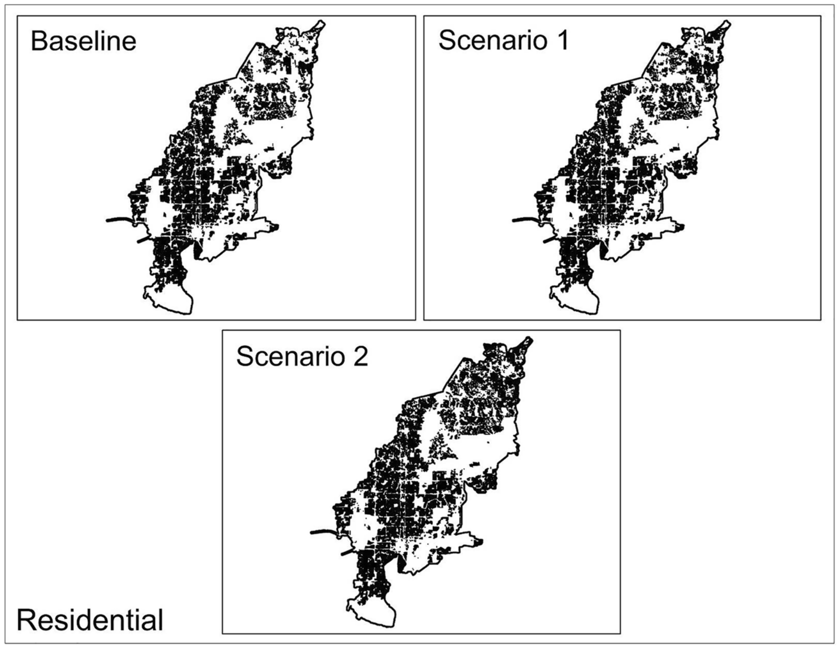

Fig. 4 Modeled LULC change for 2016 under baseline rates, increased rates (Scenario 1), and aggressive rates (Scenario 2) of urban development for the residential class

AGTC. Projections for 2016 suggested an additional increase of 11% under baseline, 15% under scenario 1 and 18% under scenario 2.

Discussion

This analysis paired ecological tree data with LULC projections under three scenarios to estimate changes in AGTC by LULC class over time. Previous estimates showed that in 2006, the City of Tampa stored 467,200 Mg AGTC (Andreu et al. 2009, 2019). According to these analyses, rural, suburban and urban land within the study area, but outside and adjacent to the City of Tampa, provided an additional 1.9 million Mg AGTC in 2006 for a total of 2,367,923 Mg AGTC throughout the 79,683 ha subbasin. On average, the TBW subbasin stored 31.5 Mg AGTC ha^{-1} compared to 11.76 in Miami-Dade County, 24.6 in Orlando and 54.4 in Gainesville (Escobedo et al. 2010; Empke et al. 2012). There may be a number of reasons for differences between these cities, including cultural perceptions on the benefits and costs associated with urban forests (Wyman et al. 2012).

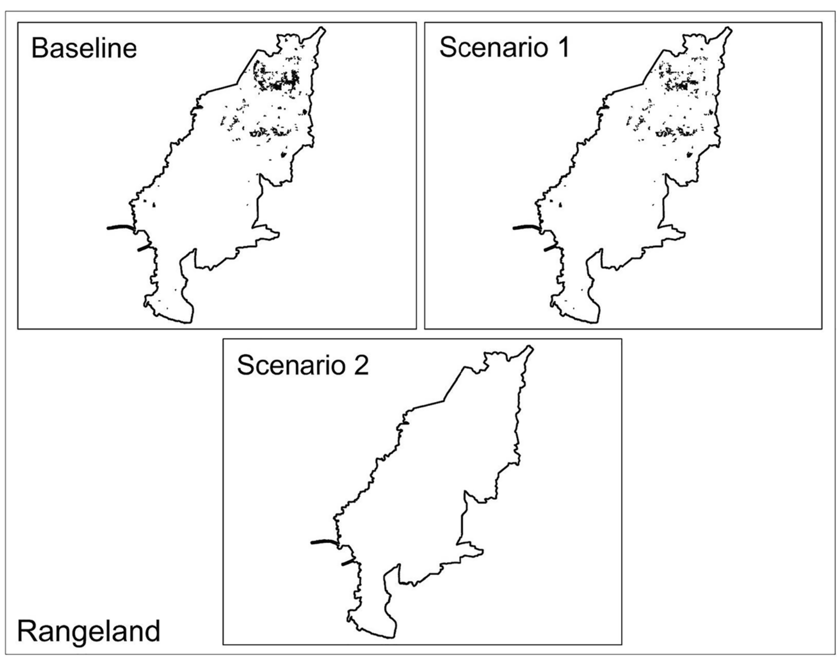

Fig. 5 Modeled LULC change for 2016 under baseline rates, increased rates (Scenario 1), and aggressive rates (Scenario 2) of urban development for the rangeland class

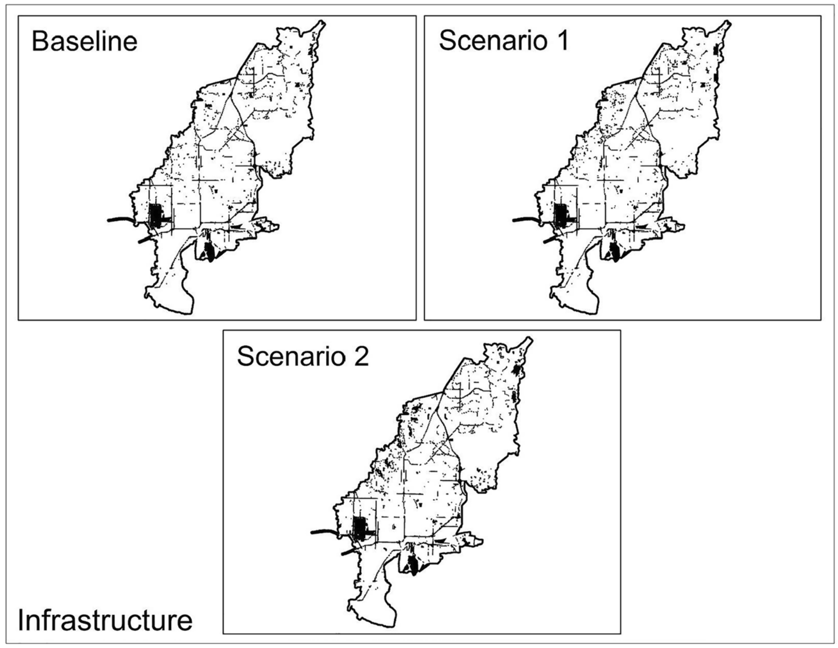

Fig. 6 Modeled LULC change for 2016 under baseline rates, increased rates (Scenario 1), and aggressive rates (Scenario 2) of urban development for the infrastructure class

Contemporary patterns of LULC change in the Tampa Bay area show the replacement of fringe rural lands with urban built classes.

This study concluded that, at least in the short-term, LULC change in urban areas causes an increase in AGTC storage because of the conversion of LULC classes containing AGTC to those with higher storage values. This suggests that urbanization may cause an increase in AGTC storage as agriculture and rangeland areas are converted to urban land-uses. Over time, as these classes disappear, it is possible that a shift in land change patterns may occur with the encroachment of urban classes on upland and wetland forested areas. This could lead to a net loss in AGTC since urban classes have been shown, on average, to contain less AGTC than forests in the TBW subbasin.

Other studies conducted in Florida also concluded that urban areas stored more C than non-urban, albeit under different landscape dynamics. A study conducted in the Apalachicola region of the Florida panhandle investigating soil and vegetative carbon storage found that urban LULC classes stored more carbon per ha than pine plantations and natural pine forests (Nagy et al. 2014). They found that urban areas stored approximately 140 Mg C ha^{-1} compared to 127 Mg C ha^{-1} in pine forests and 820 Mg C ha^{-1} in forested wetlands. Overall,

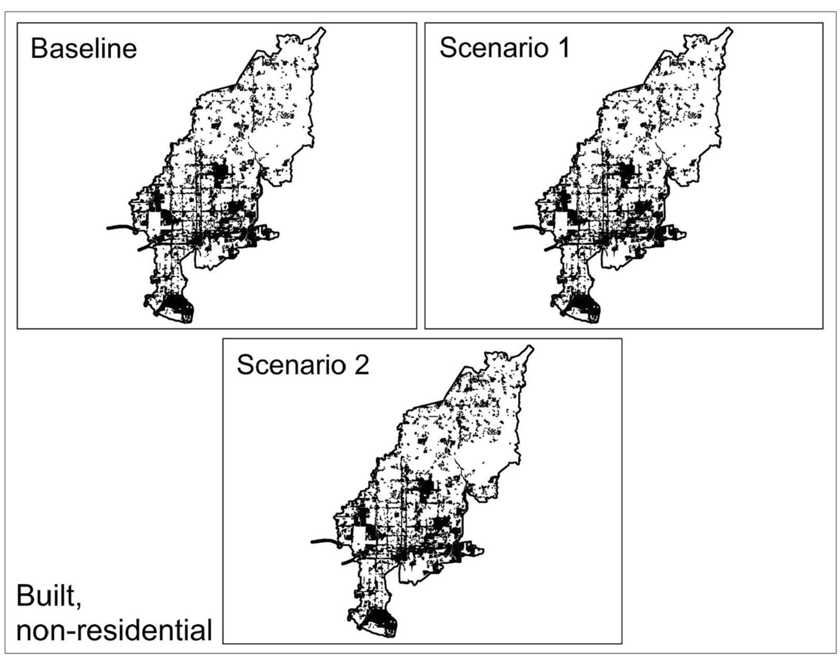

Fig. 7 Modeled LULC change for 2016 under baseline rates, increased rates (Scenario 1), and aggressive rates (Scenario 2) of urban development for the built, non-residential class

Table 6 Observed and projected class AGTC (Mg), by year, as a result of LULC change scenarios in the Tampa Bay watershed subbasin

Class	2006	2011	2016 Base	2016 Incr.	2016 Aggr.
Agriculture	0	0	0	0	0
Built, non-residential	196,089	235,234	281,042	285,534	290,346
Forested Wetlands	1,138,313	1,210,788	1,279,398	1,353,685	1,301,652
Infrastructure	6883	7839	8819	8957	9818
Non-forested & Mangrove Wetlands	59,728	80,075	95,689	96,133	98,598
Rangeland	11,408	13,361	14,055	11,998	0
Residential	788,253	923,632	1,080,734	1,118,969	1,213,863
Upland Forests	167,247	184,778	186,488	175,217	208,366
Water	0	0	0	0	0
Total	2,367,921	2,655,707	2,946,225	3,050,493	3,122,643

forested wetlands made up 30% of the landscape and stored 21,910,000 Mg C of the 27,369,000 stored across the study area. Further, LULC change projections indicate an expansion of urban areas with a corresponding increase in landscape C storage as urban classes replace pine plantations and natural pine forests (Nagy et al. 2014). The authors suggest this is likely due to differences in fire regimes between urban and forested areas in the region and caution that care should be taken interpreting these results in management plans focused on the promulgation of one ES. This highlights the importance of a systems perspective to ES management. Other studies in the Apalachicola region have shown that woody biomass productivity rates for both urban pine and urban oak forests were higher than those in natural forests of each type (Enloe et al. 2015). While internal dynamics, i.e. drivers of change, within LULC classes were not examined in this study, these results may provide a foundation for future work to look at what factors within a given class cause shifts in AGTC over time. One challenge in comparing the results of both these studies to those presented here are differences in LULC classification schemes. In the first example, only five classes of general pine-centric descriptions were used to differentiate classes while in the second, only forest stands in urban areas (as opposed to any urban vegetation) were compared to natural forested areas.

Scenario 2 was hypothetical and meant to investigate the overall effect of LULC change on landscape AGTC if the complete conversion of agriculture and rangeland were to

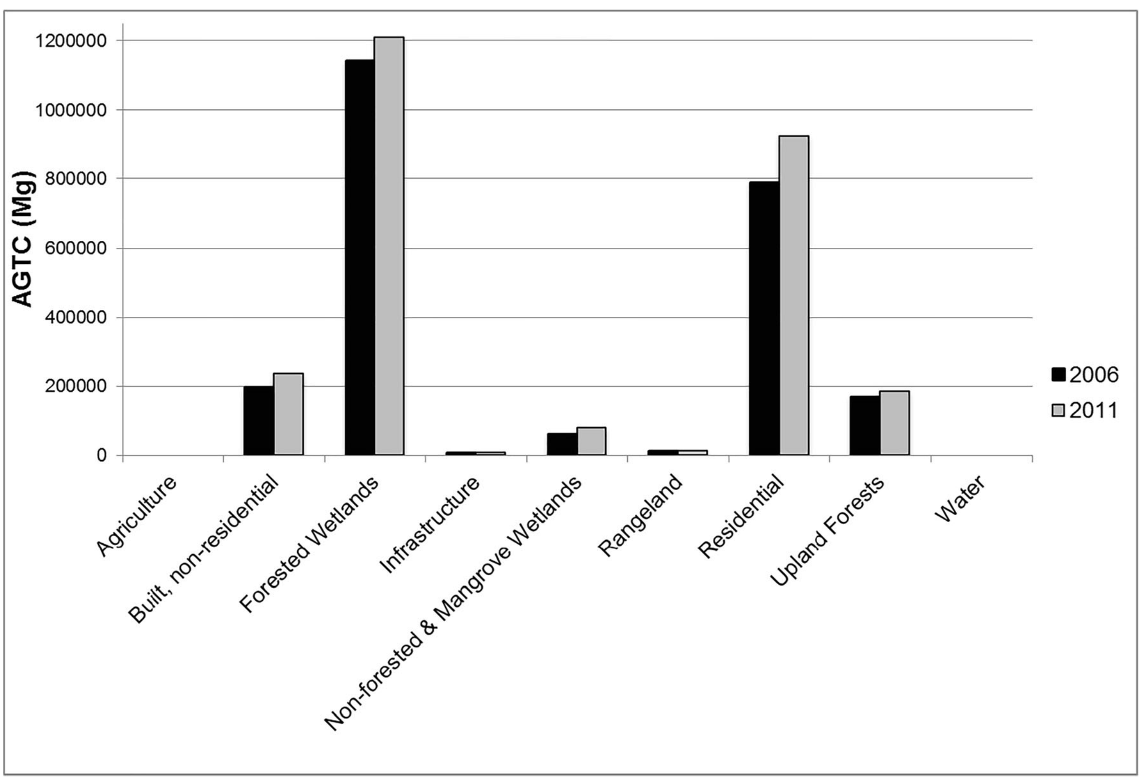

Fig. 8 Observed landscape AGTC estimates (Mg) for each LULC class

occur regardless of the timeframe. It was not meant to suggest that farm and grasslands could disappear in only five years. Rather, to highlight the effects of such a complete transition in LULC composition. Still, the overall scenario is not without precedent. As an example, the complete removal of forests, wetlands, and farmlands occurred in western Long Island in an area of 18,000 ha corresponding to modern day Kings County (Brooklyn), NY (Linder and Zacharias 1999). In the 1870's Brooklyn was described as bucolic, with forests to the north and west and over six miles of farmland to the northeast and other natural areas to the south. It ranked second only to Queens as the largest producer of vegetables in the United States, at a time before high yield industrial agriculture. In 1860 Kings County had a population of 279,000. By 1900 this number grew to 1.16 million with the near-complete conversion of agriculture to urban land during the 20 year period between 1890 and 1910 (Linder and Zacharias 1999; United States Census Bureau 2010). In 50 years the area underwent almost complete urbanization and today is the most populated in absolute terms (2.6 million) and most densely populated (14,182 km^{-2}) borough of New York City (United States Census Bureau 2010).

The proportion metrics for plot data (Table 1, Table 2) aligned with the SWFWMD class descriptions (Southwest Florida Water Management District 2015). Infrastructure had 61% of its plots with zero AGTC in both time periods. Infrastructure includes roads and other areas often kept clear of vegetation such as power lines or pipeline corridors. Still, continued maintenance of these areas can vary, so it is reasonable for some to have significant vegetation. The rangeland class had a proportion of 50% zero AGTC with a sample size of six and could have been depreciated to zero based on the criteria used for the agriculture and water classes. However, unlike those classes in which positive-value estimates are more likely to be outliers based on SWFWMD definitions, it is expected that rangeland has a higher degree of plot AGTC, even if much smaller compared to some of the AGTC-rich classes such as forested wetlands. Based on these considerations the decision was made to keep the sample data estimates to incorporate some indication of the rangeland C profile.

Standard deviations of class plot data were much larger than means. This indicates that some variation within each class was not captured in the plot data. This may be an issue with the variables collected or simply reflecting the highly heterogeneous nature of urban landscapes. In either case, further investigations of class dynamics could improve landscape-level estimates. For example, ages of trees or of the plot itself were not collected during sampling. Stand and tree age profiles are important variables for C studies because they are critical in determining C sequestration and storage rates (Coomes et al. 2012).

The temporal scale of the plot data limited modelling of AGTC change to simple linear. Non-linear techniques were not possible given a two-step time series. The continued collection of data would allow for the exploration of other time-series regression techniques.

The i-Tree methodology was tested in New Zealand by Dale (2013) who found that model estimates were all within standard error ranges of other C model estimates derived specifically for New Zealand. However, Dale notes that i-Tree does not include allometric equations for palm species but instead substitutes those derived for hardwood species to estimate palm growth. This may lead to miscalculations of landscape AGTC for areas like the TBW that have a high number of palms (Andreu et al. 2009).

Although upland forests experienced some decrease in area, LULC change in the TBW was primarily due to the conversion of agriculture and rangeland forests into urban built classes. This context is critical when interpreting these results because agriculture and rangeland areas initially contained little to no AGTC. Even though residential areas are not AGTC-rich compared to forested classes, they still contain higher amounts on average than agriculture and rangeland and so the end result is an increase in landscape AGTC. This contrasts with contemporary urbanization in the Seattle area which is driven by the conversion of forested classes. Between 1986 and 2007 Seattle saw a 100% increase in the extent of urban land. The conversion of these forests, which included old growth stands, led to an average decline of 1.2 Mg ha^{-1} yr^{-1} in C storage that contributed 15% of total C emissions for the area (Hutyra et al. 2011). This allows for some speculation on what could happen in the TBW if Scenario 2 were to occur. Contemporary patterns of LULC change in the Tampa area show the replacement of fringe rural lands with urban built classes. This has important implications for the future of C storage as these areas are further reduced in size. If urbanization were to continue beyond the complete removal of agriculture and rangeland areas, it is likely that LULC change would shift to the conversion of upland and wetland forests, which contain large amounts of AGTC. This is supported by historical observations and future projections in other studies which concluded that wetland and forested areas are most significantly impacted by exurban growth (Brown et al. 2005; Theobald 2010). The result could reverse the conclusions of this study and lead to a decrease in landscape AGTC over a longer time period of urban development. It is likely that a similar situation already occurred for the TBW upon initial settlement. Ellis et al. (2010) coupled historical LULC data with GIS analysis to show how LULC composition evolved globally from 1700 to 2000. They suggest that in 1700 over half of all available land was wild, 45% semi-natural with only slight impacts from agriculture with the remainder anthropogenic. Over the next three-hundred years, LULC composition transitioned to over 50% anthropogenic with 39% of ice-free land converted to agriculture (Ellis et al. 2010). Therefore, it is not impossible to hypothesize a scenario in which the clearing of forests and wetlands to produce farmland was likely a primary driver of historical LULC change in the TBW. If this were the case, it would almost certainly lead to a decrease in landscape storage of AGTC in future time periods if further reduction of land with high AGTC storage occurred.

Conclusions and limitations

Care must be taken when interpreting the results of this study. Residential development should not be seen as a method in which to counteract C emissions in urban areas. It must be understood that all landscapes are systems, and that alteration of one process invariably affects another. Negative consequences in hydrology, nutrient cycles, habitat loss, among many others, are associated with the development of any landscape. Also, as discussed above, in the absence of agriculture land, any further development would replace forested areas that hold more AGTC than the residential class. It is also important to remember that this study only looks at above-ground carbon stored in trees. Soils store significant quantities of C, which can exceed amounts stored above-ground. Also, while we did not include the herbaceous and shrub layers, our results are novel in that they show quantitative evidence that the biomass of the tree component of AGTC can be significant, in and of itself, for urban landscapes when compared to forested land cover. In addition, using only the tree component provides insights into the dominant carbon component of the above-ground portion of these systems. Herbaceous and shrub layers will add additional, and significant, amounts of carbon, but the primary component of the above-ground portion of these systems is that of trees.

Finally, initial LULC estimates were derived over a five-year period due to data availability. It is hoped that additional data collection efforts will expand the timeframe and types of ES used in future iterations of this analysis.

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References

Andreu MG, Friedman MH, Northrop RJ (2009) Environmental Services Provided by Tampa's Urban Forest Univ Fla IFAS Ext EDIS 2009:

Andreu MG, Friedman MH, Landry SM, Northrop RJ (2019) City of Tampa urban ecological analysis. Univ Fla IFAS Ext EDIS

Brown DG, Johnson KM, Loveland TR, Theobald DM (2005) Rural land-use trends in the conterminous United States, 1950-2000. Ecol Appl 15:1851–1863

Brundtland GH (1988) Our common future: a climate for change. In: Proceedings of the world conference on the changing atmosphere: implications for global security, pp 27–30

Coomes DA, Holdaway RJ, Kobe RK, Lines ER, Allen RB (2012) A general integrative framework for modelling woody biomass production and carbon sequestration rates in forests. J Ecol 100:42–64

Dale MJ (2013) Evaluation of methods for quantifying carbon storage of urban trees in New Zealand. Unpubl Manuscr

Davies ZG, Dallimer M, Edmondson JL, Leake JR, Gaston KJ (2013) Identifying potential sources of variability between vegetation carbon storage estimates for urban areas. Environ Pollut 183:133–142

Ellis EC, Klein Goldewijk K, Siebert S et al (2010) Anthropogenic transformation of the biomes, 1700 to 2000. Glob Ecol Biogeogr 19:589–606

Empke EK, Becker E, Lab J et al (2012) Orlando, Florida's urban and community forests and their ecosystem services. Univ Fla IFAS Ext EDIS

Enloe HA, Lockaby BG, Zipperer WC, Somers GL (2015) Urbanization effects on leaf litter decomposition, foliar nutrient dynamics and aboveground net primary productivity in the subtropics. Urban Ecosyst 18:1285–1303

Escobedo F, Varela S, Zhao M, Wagner JE, Zipperer W (2010) Analyzing the efficacy of subtropical urban forests in offsetting carbon emissions from cities. Environ Sci Pol 13:362–372

Escobedo F, Klein J, Pace M et al (2011) Miami-Dade county's urban forests and their ecosystem services. Univ Fla IFAS Ext EDIS

Escobedo F, Seitz J, Zipperer WC, Iannone B (2018) Gainesville, Florida's urban tree cover. Univ Fla IFAS Ext EDIS

Heath LS, Smith JE, Skog KE et al (2011) Managed Forest carbon estimates for the US greenhouse gas inventory, 1990—2008. J For 109: 167–173

Houghton RA (1999) The annual net flux of carbon to the atmosphere from changes in land use 1850–1990. Tellus B 51:298–313

Houghton RA (2003) Revised estimates of the annual net flux of carbon to the atmosphere from changes in land use and land management 1850–2000. Tellus B 55:378–390

Houghton RA (2010) How well do we know the flux of CO2 from land-use change? Tellus Ser B Chem Phys Meteorol 62:337–351

Hutyra LR, Yoon B, Hepinstall-Cymerman J, Alberti M (2011) Carbon consequences of land cover change and expansion of urban lands: a case study in the Seattle metropolitan region. Landsc Urban Plan 103:83–93

Lagrosa JJ, Zipperer WC, Andreu MG (2018) Projecting land-use and land cover change in a subtropical urban watershed. Urban Sci 2:11

Larondelle N, Haase D (2013) Urban ecosystem services assessment along a rural–urban gradient: a cross-analysis of European cities. Ecol Indic 29:179–190

Linder M, Zacharias LS (1999) Of cabbages and kings county: agriculture and the formation of modern Brooklyn. University of Iowa Press

McPherson EG, Xiao Q, Aguaron E (2013) A new approach to quantify and map carbon stored, sequestered and emissions avoided by urban forests. Landsc Urban Plan 120:70–84

Nagy RC, Lockaby BG, Zipperer WC, Marzen LJ (2014) A comparison of carbon and nitrogen stocks among land uses/covers in coastal Florida. Urban Ecosyst 17:255–276

Nowak DJ, Crane DE, Stevens JC et al (2008) A ground-based method of assessing urban forest structure and ecosystem services. Arboric Urban For 34:347–358

Nowak DJ, Greenfield EJ, Hoehn RE, Lapoint E (2013) Carbon storage and sequestration by trees in urban and community areas of the United States. Environ Pollut 178:229–236

Southwest Florida Water Management District (2015) Current SWFWMD Managed land. Southwest Florida Water Management District, Brooksville, FL

Strohbach MW, Haase D (2012) Above-ground carbon storage by urban trees in Leipzig, Germany: analysis of patterns in a European city. Landsc Urban Plan 104:95–104

Theobald DM (2010) Estimating natural landscape changes from 1992 to 2030 in the conterminous US. Landsc Ecol 25:999–1011

United Nations Department of Economic and Social Affairs (2014) 2014 revision of the World Urbanization Prospects. https://www.un.org/en/development/desa/publications/2014-revision-world-urbanization-prospects.html

United States Census Bureau (2000) The 2000 United States Census. https://www.census.gov/programs-surveys/decennial-census/decade.2000.html

United States Census Bureau (2010) The 2010 United States Census. http://www.census.gov/2010census/

Verburg PH (2010) The CLUE modelling framework: the conversion of land use and its effects. University Amsterdam. Inst Environ Stud, Amsterdam

Verburg PH, Overmars KP (2009) Combining top-down and bottom-up dynamics in land use modeling: exploring the future of abandoned farmlands in Europe with the Dyna-CLUE model. Landsc Ecol 24:1167–1181

West PC, Gibbs HK, Monfreda C, Wagner J, Barford CC, Carpenter SR, Foley JA (2010) Trading carbon for food: global comparison of carbon stocks vs. crop yields on agricultural land. Proc Natl Acad Sci 107:19645–19648

Wyman M, Escobedo F, Stein T, Orfanedes M, Northrop R (2012) Community leader perceptions and attitudes toward coastal urban forests and hurricanes in Florida. South J Appl For 36:152–158

Zipperer WC, Foresman TW, Walker SP, Daniel CT (2012) Ecological consequences of fragmentation and deforestation in an urban landscape: a case study. Urban Ecosyst 15:533–544