

Accounting for urban biogenic fluxes in regional carbon budgets

Running title: Urban Biogenic Fluxes

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Abstract

Many ecosystem models incorrectly treat urban areas as devoid of vegetation and biogenic carbon (C) fluxes. We sought to improve estimates of urban biomass and biogenic C fluxes using existing, nationally available data products. We characterized biogenic influence on urban C cycling throughout Massachusetts, USA using an ecosystem model that integrates improved representation of urban vegetation, growing conditions associated with urban heat island (UHI), and altered urban phenology. Boston's biomass density is $\frac{1}{4}$ that of rural forests, however 87% of Massachusetts' urban landscape is vegetated. Model results suggest that, kilogram-for-kilogram, urban vegetation cycles C twice as fast as rural forests. Urban vegetation releases (R_E) and absorbs (GEE) the equivalent of 11 and 14%, respectively, of anthropogenic emissions in the most urban portions of the state. While urban vegetation in Massachusetts fully sequesters anthropogenic emissions from smaller cities in the region, Boston's UHI reduces annual C storage by >20% such that vegetation offsets only 2% of anthropogenic emissions. Asynchrony between temporal patterns of biogenic and anthropogenic C fluxes further constrains the emissions mitigation potential of urban vegetation. However, neglecting to account for biogenic C fluxes in cities can impair efforts to accurately monitor, report, verify, and reduce anthropogenic emissions.

Keywords: urban ecology, carbon emissions, GEE, R_E , NEE, forest, biomass, urban heat island

1.0 Introduction

Fossil fuel emissions of carbon dioxide (FFCO₂) originating from urban areas account for >70% of anthropogenic CO₂ emissions globally (International Energy Agency, 2008; Le Quéré et al., 2013; U.S. Energy Information Agency, 2013). Recent studies have improved spatial and temporal resolution of anthropogenic carbon (C) emissions estimates (Bréon et al., 2015; Gately et al., 2015; McKain et al., 2015; Turnbull et al., 2015), but the influence of urban vegetation on C flows has received less attention and remains poorly constrained (Churkina, 2008; Hutryra et al., 2014; Raciti et al., 2012). Uncertainty in the timing, magnitude, and direction of C fluxes from urban vegetation limits efforts to accurately monitor, report, verify, and mitigate urban anthropogenic C emissions.

Both bottom-up (Briber et al., 2015; Gough and Elliott, 2012) and top-down (Pataki et al., 2007; Turnbull et al., 2015) studies have demonstrated that vegetation is highly active in the urban C cycle. For example, urban biogenic sink strength has been reported to range from 1.8% to 18% of urban fossil fuel emissions (McPherson and Simpson, 1999; Yin et al., 2010; Zhao et al., 2010). Most studies of urban fluxes attribute seasonal patterns to land cover variability (e.g., urban vegetation) only in broad, qualitative terms or to spatially limited areas (i.e., immediately surrounding a flux tower), which limits city-scale understanding of the urban C cycle (Bergeron and Strachan, 2011; Crawford et al., 2011; Helfter et al., 2011; Järvi et al., 2012; Kordowski and Kuttler, 2010). The widespread presence of vegetation in and around cities complicates precise characterization of urban CO₂ budgets using atmospheric observations, particularly due to the spatially heterogeneous arrangement of urban vegetation and seasonality of urban biogenic C fluxes (Bergeron and Strachan, 2011; Crawford et al., 2011; Järvi et al., 2012). Correcting for temporal aliasing of

biogenic and anthropogenic fluxes requires careful partitioning of each to attribute sources using atmospheric measurements (Briber et al., 2013; Gurney et al., 2005; Hutryra et al., 2014; Myeong et al., 2006).

Unique growing conditions facilitate elevated biogenic C cycling rates in urban ecosystems relative to non-urban ecosystems (Hutryra et al., 2014). For example, urban areas experience elevated ambient air temperatures (the “urban heat island” effect; UHI) (Kim, 1992; Oke, 1982), which cause seasonally-dependent changes in C fluxes from urban vegetation and soils (Decina et al., 2016; Pataki et al., 2006; Zhang et al., 2004; Zhao et al., 2012), and extend the urban growing season (Melaas et al., 2016a, 2016b; Zhang et al., 2004). Urban vegetation may thus sequester atmospheric CO₂ at different rates than rural vegetation on a per unit biomass basis (Zhao et al., 2016), while urban soil respiration patterns (spatial and temporal) may differ due to elevated ambient air temperatures (i.e., UHI), impervious surface areas (ISA, i.e. pavement and buildings) that restrict diffusion of CO₂ from soils, and human addition of labile C sources (George et al., 2007; Ziska et al., 2004). Some urban growing conditions negatively impact growth rates; e.g. exposure to ozone reduces photosynthesis rates (Krupa and Manning, 1988; Ollinger et al., 2002). While urban areas are strong emitters of O₃ precursors, Gregg et al. (2003) observed higher [O₃] in rural areas downwind of cities due to transport and competitive interactions that scavenge O₃ precursors. Nevertheless, Briber et al. (2015) reported growth rates of urban trees to be twice those observed in rural forests and documented accelerated tree growth following urbanization suggesting a net positive effect of urban growing conditions. Other inventory studies demonstrate potentially large C sequestration rates in cities across many biomes (Churkina et al., 2010; Jo, 2002; Nowak and Crane, 2002; Zhao et al., 2012). Despite this evidence, biogenic C

fluxes from urban vegetation are often treated as known, neutral, or negligible (Gurney et al., 2005; Kennedy et al., 2012) introducing biases of unknown magnitude into the measurement and modeling of anthropogenic emissions.

To improve understanding of the influence vegetation exerts on the urban C cycle, we combined existing land cover products with field estimates of urban vegetation biomass to produce an improved map of urban biomass density. This was compared with spatially and temporally resolved model estimates of biogenic and anthropogenic C fluxes. We estimated biogenic C fluxes (gross ecosystem exchange [GEE] and ecosystem respiration [R_E]) using the Vegetation Photosynthesis and Respiration Model (VPRM) (Mahadevan et al., 2008), a remote sensing-based light use efficiency model that we modified to incorporate the altered phenology, higher air temperatures, and ISA in urban ecosystems. These fluxes are compared with new inventories of anthropogenic emissions to produce a comprehensive C budget for the state of Massachusetts. Importantly, we restricted our analysis to use nationally available data sources so that these methods can be extended to other urban areas.

2.0 Methods

To investigate the role of vegetation in the urban C cycle, we combined an improved map of vegetation in urban areas, an ecosystem model of biogenic C flows that incorporates urban growing conditions normally ignored by ecosystem models, and a novel, comprehensive inventory of spatially and temporally resolved anthropogenic CO₂ emissions.

2.1 Study Area

Our study focused on the state of Massachusetts (MA), USA with three 20km x 30km (600 km²) focal areas corresponding to communities spanning a gradient from low to high urban development intensity: Petersham (42.54°N, 72.17° W; 44 persons km⁻²), Worcester (42.27°N, 71.84° W; 340 persons km⁻²), and Boston (42.356°N, -71.062°W; 2,049 persons km⁻²) (population density from ORNL, 2014). All three focal areas contain atmospheric CO₂ sampling sites. MA is predominantly covered by northern mixed-deciduous forest (60%) and developed areas (25-38%), with small areas of agriculture, grasslands, and wetlands (Homer et al., 2015; US Census Bureau, 2010). Climate in MA is temperate with mean summer (JJA) and winter (DJF) temperatures of 20°C and -4°C, respectively, and mean annual precipitation of 1125mm (National Climatic Data Center, 2015).

2.2 Biomass Map

To develop a map depicting the quantity and distribution of aboveground biomass (AGB) across the state, we used the National Biomass and Carbon Dataset (NBCD; Kellndorfer et al., 2013) as a base estimate. While NBCD AGB estimates are tuned to be consistent with county-scale FIA data, it tends to overestimate AGB and underestimate vegetation extent in urban areas (Raciti et al., 2014). To correct this bias, we revised the NBCD urban biomass estimates

using a linear statistical relationship between field measurements of AGB in urban plots (n=299, see S1 for additional details on field plots), growing season mean EVI (Enhanced Vegetation Index, a satellite-derived metric of land-surface greenness), and forest canopy cover from the 2011 National Land Cover Database (NLCD) (Homer et al., 2015). While we applied this relationship using a linear model, the pattern likely asymptotes at high biomass values (Huete et al., 2002). However, high biomass areas are rare in most urban areas, and so the impact of this mis-parameterization is minimal. The relationship between biomass observed in ground plots throughout the greater Boston area and corresponding predicted values, with associated 95% confidence and predictive intervals is shown in Figure S1-1. We produced the Better Urban Biomass Map (BU-BioM) at 30m spatial resolution by applying this statistical relationship (SI Eqn. 1, model adjusted $R^2=0.51$, $p<0.01$; see S1 for additional detail) to the intersection of NLCD-defined urban areas (classes 21-24) and the areas covered by the urban areas/urban clusters (UA/UC) in the US Census (US Census Bureau, 2010). The total affected area was 20.2% of the state, or 4,220 km². For pixels outside of the UA/UC that were classified by NLCD as urban, we retained the AGB value reported by the NBCD. Pixels inside UA/UC classified as non-urban by NLCD were also unchanged from the NBCD AGB values.

2.3 UrbanVPRM

We estimated biogenic C fluxes over MA for 2013 at hourly time step and 500m resolution using a modified version of the Vegetation Photosynthesis and Respiration Model (VPRM), a data-driven, spatially-explicit model for ecosystem fluxes based on remote sensing and climate data (Mahadevan et al., 2008). For this study, we modified VPRM to incorporate some urban-specific influences on C dynamics. Due to the dearth of urban parameterizations within ecosystem

models, we treated MA urban areas as similar to the deciduous broadleaf forest predominant in the region. We evaluated inclusion of multiple landcover types, but the available suite of choices is unrepresentative of contemporary or historical landcover patterns in this region and ultimately we chose to represent the region as uniformly forested—a decision supported by analyses of historical landcover in the region (Foster, 2009; Fuller et al., 1998). As urban areas appear in a wide variety of biomes, it is unlikely that a single parameterization could exist for a generic urban ecosystem; as such, it is important to consider the background eco-regional context of a city and the natural vegetation that urban land use displaces when estimating ecosystem behavior in a given city (Imhoff et al., 2004). This modified model, henceforth referred to as UrbanVPRM, incorporates a large and highly variable fraction of ISA as a measure of urban intensity from the NLCD (Homer et al., 2015). Emission of respired CO₂ from paved soils is restricted to diffusion through adjacent unpaved areas or cracks in pavement (Lorenz and Lal, 2009; Nehls et al., 2006; Scalenghe and Marsan, 2009). The model did not allow soil respiration from paved areas but did include autotrophic respiration from overhanging vegetation (see also S2.4). High ISA fraction also modifies ambient air temperature. We modeled the spatially and temporally varying UHI effect using two years of surface air temperature measurements from a sensor network deployed across an urban gradient in the study region ($n = 25$). This correction modified air temperatures by up to 2.5°C and extended the growing season, with corresponding influences on photosynthesis and ecosystem respiration rates. Luus and Lin (2015) provide a rigorous analysis of uncertainty and error in a version of VPRM modified for polar ecosystems. We ran the model both with and without the UHI-modified temperature drivers to evaluate the influence of UHI on C fluxes from urban

vegetation. Parameterization, data sources, and additional modifications made to VPRM are further described in S2.

2.4 Anthropogenic C emissions

We built an inventory of anthropogenic CO₂ emissions at 1km spatial resolution and hourly time-steps for the year 2013. This inventory was produced by assimilating data from the EPA National Emissions Inventory (U.S. Environmental Protection Agency, 2014) and the EPA Greenhouse Gas Reporting Program (GHGRP) to calculate CO₂ emissions from non-road transportation (railroads, marine vessels, off-road vehicles, airport taxiing, take-off and landing operations), residential, commercial, industrial, and electric power generation sectors. On-road emissions were obtained from the Database of Road Transportation Emissions (DARTE; Gately et al., 2015). We calculated mean hourly CO₂ fluxes from human respiration using population data from the national census-derived LandScan product (Bright et al., 2012; ORNL, 2014), with an emission rate of 257 g/C/person/day (Prairie and Duarte, 2007). To facilitate comparison across our three study areas, sectors were aggregated into three groups: on-road emissions, point source emissions (electric generation, airports, and industrial emissions), and other sources (industrial, residential, commercial, human respiration, and off-road vehicle emissions). Full details of the emissions calculations are reported in S3 and Gately and Hutyra (in review) conduct a comprehensive analysis of variations across inventories. All biological and anthropogenic C fluxes are reported using atmospheric sign convention, with negative values indicating biogenic uptake by the land from the atmosphere.

3.0 Results and Discussion

3.1 Refining Representation of Urban Biomass

Typical land cover maps and ecosystem models do not estimate C fluxes for urban areas (e.g. the MODIS NPP product, MOD17, see Figure 1) or do not incorporate urban influences (e.g. NBCD, sensu Raciti et al., 2014). Our method of estimating AGB in urban areas differs fundamentally from that used to produce the NBCD, and our results reveal substantial differences in the spatial patterns of AGB relative to the NBCD (Figure 1A-B). In areas of the state for which we re-estimated biomass, mean (standard deviation) AGB was 21.7 (25.8) MgC ha⁻¹ -- roughly 25 percent lower than the NBCD density of 27.2 (36.5) MgC ha⁻¹ -- and the areal extent of urban vegetation increased statewide by 62.3%. Large standard deviations in AGB reflect the inherent heterogeneity in urban land cover. Where present, AGB ranges from 1-192 and 47-132 MgC ha⁻¹ in the BU-BioM and NBCD, respectively (Figure S1-2), with interquartile ranges of 23 and 69 MgC ha⁻¹ for BU-BioM and NBCD, respectively. While the two estimates of mean biomass are broadly in agreement, BU-BioM corrects for both underestimation of urban vegetation extent and pixel-level overestimation of urban AGB in the NBCD, patterns that are consistent with Raciti et al. (2014).

AGB density declined with urban intensity, but was still widespread even in the densest urban areas (Figure 1A, Figure 2). For example, AGB in rural Petersham averaged 72.7 (37.6) MgC ha⁻¹, while AGB in Worcester and Boston had mean AGB that was 68% and 25% of rural AGB, respectively. Only 13.4% of Boston was estimated to be devoid of large stature vegetation (i.e., trees), indicating that C sources and sinks form a heterogeneous mosaic throughout the urban landscape.

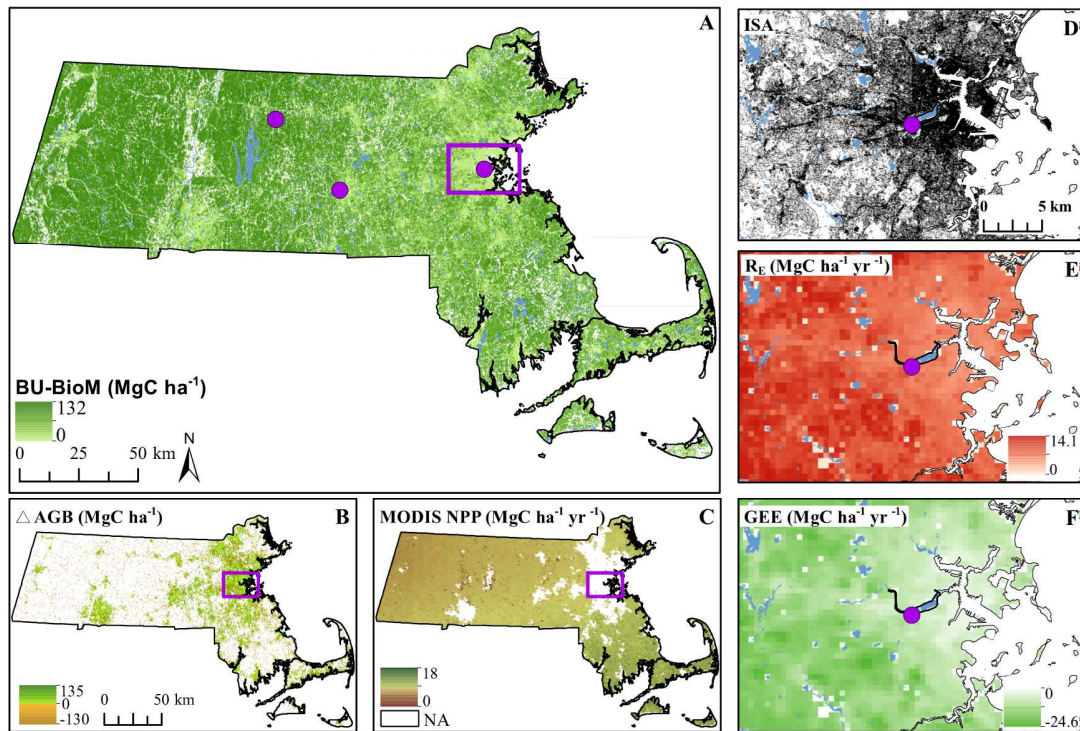


Figure 1. The Better Urban Biomass Map (BU-BioM) provides refined estimates of vegetation biomass in urban ecosystems (A), demonstrating that existing biomass products under-represent the extent and quantity of vegetation in urban ecosystems (B). Urban biogenic C exchange is strongly influenced by urban factors such as impervious surface area (ISA) (D), which alters spatial patterns of ecosystem respiration (E) and gross ecosystem exchange (F). Carbon fluxes in (E) and (F) are the mean of all noon hours for July 2013 but are reported with annualized units for ease of comparison to Panel C. Purple dots show the center of focal area locations and purple boxes correspond to view extent in Panels D-F.

3.2 Biogenic urban carbon flows

As with AGB, most model- or remote sensing-based estimates of landscape-level biogenic fluxes omit urban areas from their analysis (Figure 1C). Further, current approaches that do model biogenic urban C dynamics often do not account for unique elements of urban growing conditions, but instead represent urban areas as the unmodified “background” biome (Churkina, 2008; Imhoff et al., 2004; Nowak and Crane, 1998). UrbanVPRM used for this work incorporates urban influences such as UHI, phenological shifts, and ISA that alter spatial and temporal patterns of biogenic C fluxes (GEE and R_E) compared to natural systems (Figure 1D-F).

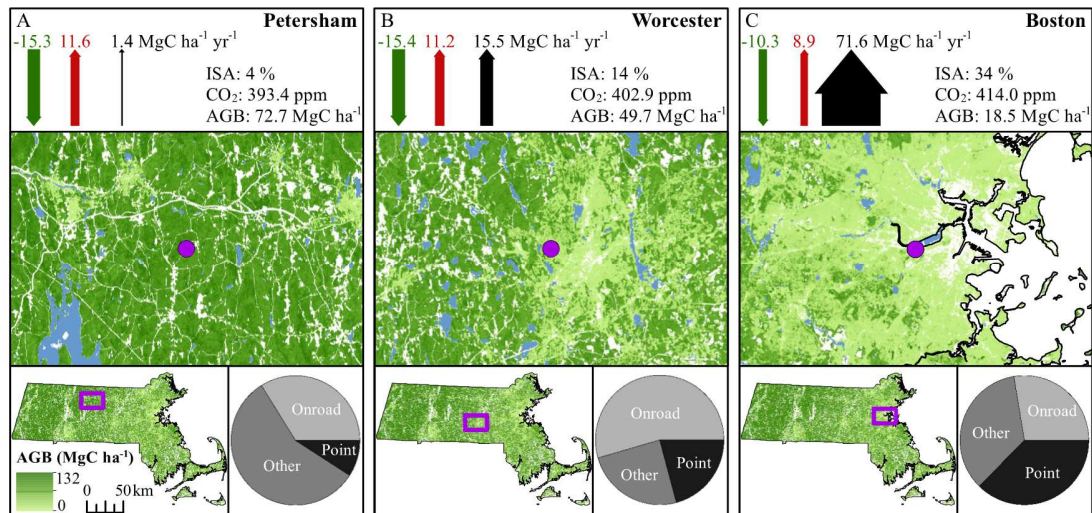


Figure 3. Spatial patterns of biomass and C fluxes in areas of increasing urban density in Petersham (A), Worcester (B), and Boston (C). Arrows correspond to mean annual C fluxes for 2013 (GEE-green, R_E-red, Anthropogenic-black) with arrow direction indicating C movement into (down) or out of (up) the ecosystem. NEE is the difference of GEE and R_E. All arrow widths are proportional to Boston's anthropogenic emissions. Fraction of impervious surface area (ISA), atmospheric CO₂ concentration, and mean aboveground biomass (AGB) density from BU-BioM are listed for each 20x30km area surrounding an atmospheric monitoring tower (purple dots). Insets (purple boxes) show the location of each focal area within the state. Pie charts illustrate the change in relative contribution of each anthropogenic emissions sector across this gradient of urban intensity.

Modeled annual mean GEE was similar in Petersham and Worcester (-15.3 and -15.4 MgC ha⁻¹ yr⁻¹, respectively; Figure 2), even though Worcester had 32% less AGB and 21.6% less canopy cover than Petersham. Worcester and Petersham had 56.5% and 72.0% canopy cover, respectively. Previous estimates for GEE of urban vegetation range from -2.3 MgC ha⁻¹ yr⁻¹ based on upscaled plot measurements in Boston (Nowak *et al* 2013) to -7.1 MgC ha⁻¹ yr⁻¹ in Detroit, USA based on a LUE model (Zhao *et al.*, 2007). Our estimate of Boston's GEE (-10.3 MgC ha⁻¹ yr⁻¹) is higher than these published estimates, but similar to that observed by an eddy covariance tower at nearby Petersham (-14.0 MgC ha⁻¹ yr⁻¹; Urbanski *et al.*, 2007). GEE in Boston was 67.6% of that in Petersham, but possessed 75% less AGB (Figure 2). GEE per unit AGB (GEE/AGB) was 1.5 and 2.5 times higher in Worcester and Boston, respectively, than in Petersham. Reinmann and Hutrya (Reinmann and Hutrya, 2017) documented 89% higher growth rates near edges

of urban forest fragments and much higher sensitivity of growth rates to climate drivers (temperature & heat stress), results consistent with our model estimates of elevated growth rates in response to urban growing conditions. Lower vegetation density in urban Boston coupled with per-capita growth rates double those at Harvard Forest (Briber et al., 2015) suggests that our estimates are reasonable. These results suggest that urban ecosystems may annually assimilate C at a proportionally higher rate relative to the quantity of AGB they contain, likely due to a lengthened growing season, reduced soil respiratory losses, and enhanced spring and autumn photosynthesis. Other urban growing conditions (e.g., ozone stress, reduced light competition, fertilizer application, irrigation, and altered species composition in urban forests) are not specifically included in our model, but several of these likely contribute to elevated C cycling rates observed by several studies. Thus, our modeled GEE per AGB values indicate a conservative, lower bound for potential urban C cycling.

Modeled R_E , driven primarily by temperature and the amount of pervious ground cover in UrbanVPRM, peaks in summer (Figure 3 A3, B3) but is >0 throughout the year in both rural and urban areas, which is consistent with local observations (Luus and Lin, 2015; Urbanski et al., 2007) (Fig 3 A2, B2). In Boston, spatial variation of summer R_E mirrored the spatial variability in ISA (Figure 1 D&E), with low R_E co-occurring with high ISA. Spatial variation in annual modeled R_E , measured as the coefficient of variation in R_E across model pixels, was similar in Petersham and Worcester (0.24 and 0.25, respectively) and much higher in Boston (0.35), indicating that variation in ISA contributes substantially to variation in modeled R_E . Further, while average soil respiration was lower across the Boston area than in the Worcester or Petersham areas due to high ISA (Figure 2), average respiration rates were elevated in pervious portions of urban

areas due to higher air temperatures arising from the urban heat island effect (Wang et al., 2016). Our estimates of summer R_E rates across the Boston area ($0-0.2 \text{ gC m}^{-2} \text{ hr}^{-1}$, Fig 1 E) are consistent with soil respiration field measurement across a 25 km transect in Boston ($0.09 \pm 0.04 \text{ gC m}^{-2} \text{ hr}^{-1}$) (Decina *et al* 2016, Figure 3 E).

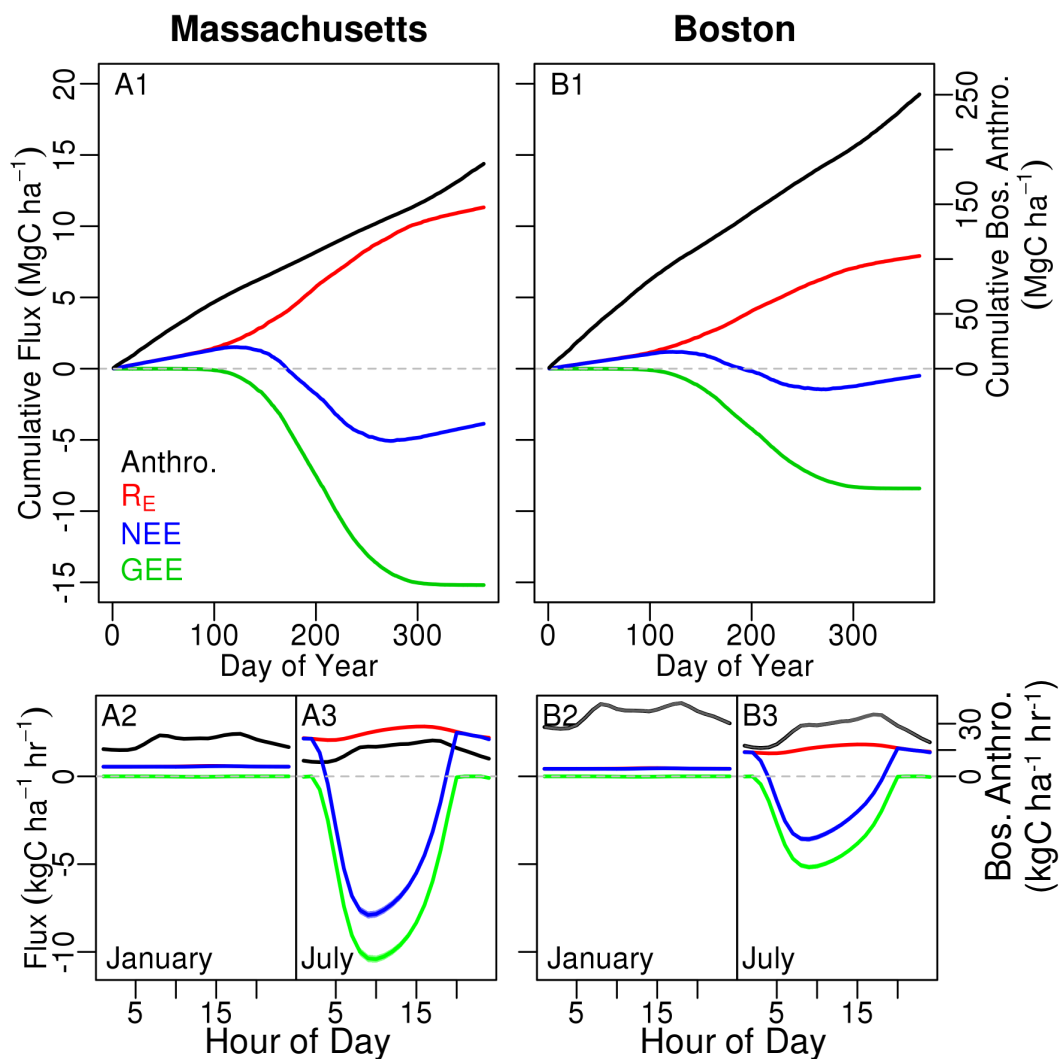


Figure 3. Cumulative C fluxes (MgC ha^{-1}) for 2013 indicate that MA is a net biogenic C sink and that on an area-basis statewide anthropogenic emissions are of similar magnitude as biogenic fluxes (A1). Both biogenic and anthropogenic fluxes ($\text{kgC ha}^{-1} \text{ hr}^{-1}$) follow daily cycles, the amplitude of which varies seasonally (A2, A3). Boston's biogenic fluxes are dwarfed by anthropogenic C emissions (B1) but follow similar patterns as seen at the state-level (B1, B2). Anthropogenic fluxes in all B panels are plotted on a second y-axis to facilitate comparison with biogenic fluxes.

Statewide, forests comprised a net biogenic C sink of $-3.9 \text{ MgC ha}^{-1} \text{ yr}^{-1}$ (Figure 3 A1), substantially greater than previously reported (Lu et al., 2013), but broadly consistent with flux tower observations (Urbanski et al., 2007). Vegetation in Boston constituted a net biogenic C sink in 2013, storing an estimated annual total of $-1.5 \text{ MgC ha}^{-1} \text{ yr}^{-1}$ (Figure 3 B1). This value is substantially lower than non-urban temperate deciduous-dominated forests (Gough et al., 2016; Urbanski et al., 2007), but similar to rates reported for urban Boston: -1.7 ± 0.23 and $-1.8 \pm 0.4 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$, by Nowak *et al* (2013) and Briber et al (2015), respectively. Among the three focal areas in this study, modeled annual mean NEE was highest in Worcester, where vegetation stored three times more C ($-4.2 \text{ MgC ha}^{-1} \text{ yr}^{-1}$) than Boston (Figure 2), which is consistent with field observations from the Boston area (Briber et al., 2015). The Petersham rural site had 46% more AGB than Worcester (72.7 MgC ha^{-1}), but was a slightly weaker C sink with annual NEE equal to $-3.7 \text{ MgC ha}^{-1} \text{ yr}^{-1}$ (12% lower than Worcester). Biogenic R_E in Worcester ($11.2 \text{ MgC ha}^{-1} \text{ yr}^{-1}$) was only 40% of total emissions ($R_E + \text{Anthropogenic} = 26.8 \text{ MgC ha}^{-1} \text{ yr}^{-1}$) because of extensive paved areas under tree canopies. Relative to Petersham and Boston, Worcester's vegetation represents a 'Goldilocks Zone' in which extensive ISA suppresses soil respiration, while other urban growing conditions have neutral or positive effects on GEE.

The UHI extends the growing season in Worcester and Boston relative to Petersham. Mean growing season length (defined as the portion of the year when $\text{EVI} \geq 50\%$ of maximum EVI) was 10.8 and 20.3 days longer in Worcester and Boston, respectively, than Petersham's growing season (140.2 ± 8.8 days). This is consistent with the findings of Melaas et al. (2016a); that the growing season was 18-22 days longer in Boston relative to adjacent rural areas. Spring arrived 3.1 ± 3.4 and 6.5 ± 5.1 days earlier and fall arrived 7.7 ± 5.4 and 13.8 ± 6.3 days later in

Worcester and Boston, respectively, than in Petersham. Longer growing seasons in Worcester and Boston may increase annual C storage, but the UHI may also impose increased temperature stress on urban vegetation. To quantify the C consequences of the UHI, we ran two versions of the UrbanVPRM for Boston and Worcester: one that applied the empirical UHI adjustment to air temperature, and a second based on unaltered air temperatures. The influence of UHI on biogenic C flux estimates was estimated as the difference between the outputs of these model runs. Urban air temperatures were consistently higher than rural air temperatures and increased R_E by $0.07 \text{ MgC ha}^{-1} \text{ hr}^{-1}$ throughout the year. UHI enhanced Boston GEE estimates by $0.05 \text{ MgC ha}^{-1} \text{ hr}^{-1}$ in the spring and fall, but this was more than offset by a $0.11 \text{ MgC ha}^{-1} \text{ hr}^{-1}$ reduction in summer GEE associated with heat stress. The net influence of UHI on GEE and R_E reduced Boston's estimated annual C storage by 22.8%. The same analysis for Worcester indicates a 2.9% reduction in annual NEE associated with UHI, suggesting minimal influence of UHI on annual C storage by vegetation in a less intensely urbanized/more heavily vegetated ecosystem. These results suggest that growing season extension via UHI does not necessarily enhance C storage by urban vegetation.

3.3 Fossil fuels compared to gross and net fluxes

Statewide, MA is a net C source, with biogenic NEE offsetting 27% of anthropogenic emissions annually. Most C storage occurs in the rural, western portion of the state, and sink strength declines with increasing urban intensity (Figure 2). Urban NEE accounted for 8.3% of all of MA's cumulative NEE. In the Boston focal area, urban vegetation offset just 2.1% of anthropogenic emissions, consistent with previous conclusions that urban vegetation is unlikely to directly

offset a significant fraction of urban anthropogenic emissions (McPherson and Simpson, 1999; Pataki et al., 2011; Zhao et al., 2010).

Both biogenic and anthropogenic C fluxes follow daily and seasonal patterns that are evident statewide and within the Boston area (Figure 3). Peak daily anthropogenic emissions coincide with morning and evening commuting periods (circa 8AM and 5PM), whereas biogenic fluxes peak during mid-day (Figure 3 A2-3, B2-3). Anthropogenic emissions rates were higher in winter than in summer (Figure 3) due to local combustion associated with heating demands, which contrasts sharply with seasonal biogenic flux patterns. Daily and seasonal asynchrony between biogenic C fluxes and anthropogenic emissions suggests that the urban biogenic C sink is phenologically incapable of mitigating emissions during their peak. However, these sink/source temporal dynamics require careful consideration when interpreting urban atmospheric observations from towers (McKain et al., 2012) or satellite observations of urban column CO₂ concentration (e.g. from OCO-2; Duren and Miller, 2012) since biogenic C fluxes do significantly alter atmospheric CO₂ mixing ratios, both daily and seasonally.

3.4 Urban-rural vegetation differences and implications for the future

Rural forests are not reliable analogs for estimating urban biogenic C fluxes. Kilogram-for-kilogram, urban vegetation exchanges C with the atmosphere faster than rural forests (Briber et al., 2015; Reinmann and Hutyyra, 2017; Zhao et al., 2016). As urban intensity increased, AGB-normalized fluxes increased two- and three-fold for GEE and R_E, respectively (Figure 3), suggesting that while NEE is small, vegetation in urban areas cycles C significantly faster than in rural forests, and that urban GEE and R_E are not linearly proportional to AGB. Further, the majority of urban-specific growing conditions not yet included in our model

(e.g. N fertilization, irrigation, elevated CO₂) are likely to boost vegetation C uptake (Rao et al., 2014; Zhang et al., 2014), which could potentially offset the influence of UHI. Additional discussion of factors affecting urban biogenic C dynamics is provided in S4. Future work will examine the influence of these additional drivers on biogenic C fluxes. More generally, these results underscore the importance of accounting for urban growing conditions that are not encountered in rural vegetation (Zhao et al., 2016), and suggest that understanding the ecology of rural forests is not sufficient to understand the role of vegetation in the urban C cycle. Ecosystem models that treat urban forests as identical to rural forests are likely to underestimate urban ecosystem C exchange.

3.5 Conclusions

Our results indicate that neither biomass nor biogenic C fluxes are negligible in urban ecosystems. GEE and R_E per unit AGB in urban vegetation are elevated relative to rural forests. Multiple distinct urban growing conditions suggest that our understanding of forest ecology in rural areas is necessary, but not sufficient, to assess the role of vegetation in the urban C cycle. Urbanization promotes heterogeneity of C sources and sinks at fine spatial scales and the near-ubiquitous (albeit low density) vegetation biomass we observed in urban areas poses challenges to monitoring, reporting, and verification of anthropogenic C emissions using conventional ecological models, remote sensing, or inverse atmospheric models based on daytime observations. Further, the significant diurnal and seasonal variation in sources and fluxes of C needs to be resolved at sub-urban scales to accurately attribute sources and sinks, which requires relatively high resolution remote sensing data. Our methods rely on nationally available data sources coupled with local ground observations and can be readily

applied in other cities to improve understanding of the role of vegetation in urban ecosystem C flows across a broad range of biomes and urban forms.

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