Assessing the global warming potential of human settlement expansion in a mesic temperate landscape from 2005 to 2050

Andrew B. Reinmann*[1], Lucy R. Hutyra[1], Andrew Trlica[1] and Pontus Olofsson[1]

[1]Department of Earth and Environment, Boston University, 685 Commonwealth Ave., Boston, MA 02215.

*Corresponding Author:

Andrew B. Reinmann (T) 617-353-8828; (F) 617-353-8399; email: reinmann@bu.edu

Co-author email addresses:

Lucy R. Hutyra: lrhutyra@bu.edu

Andrew Trlica: atrlica@bu.edu

Pontus Olofsson: olofsson@bu.edu

Abstract:

Expansion of human settlements is an important driver of global environmental change that causes land use and land cover change (LULCC) and alters the biophysical nature of the landscape and climate. We use the state of Massachusetts, United States (U.S.) to present a novel approach to quantifying the effects of projected expansion of human settlements on the biophysical nature of the landscape. We integrate nationally available datasets with the U.S. Environmental Protection Agency's Integrated Climate and Land Use Scenarios model to model albedo and C storage and uptake by forests and vegetation within human settlements. Our results indicate a 4.4 to 14% decline in forest cover and a 35 to 40% increase in developed land between 2005 and 2050, with large spatial variability. LULCC is projected to reduce rates of forest C sequestration, but our results suggest that vegetation within human settlements has the potential to offset a substantial proportion of the decline in the forest C sink and may comprise up to 35% of the terrestrial C sink by 2050. Changes in albedo and terrestrial C fluxes are expected to result in a global warming potential (GWP) of +0.13 Mg CO_2-C-equivalence ha^{-1} yr^{-1} under the baseline trajectory, which is equivalent to 17% of the projected increase in fossil fuel emissions. Changes in terrestrial C fluxes are generally the most important driver of the increase in GWP, but albedo change becomes an increasingly important component where housing densities are higher. Expansion of human settlements is the new face of LULCC and our results indicate that when quantifying the biophysical response it is essential to consider C uptake by vegetation within human settlements and the spatial variability in the influence of C fluxes and albedo on changes in GWP.

Keywords: albedo, carbon, climate change, forests, land use change land cover change, urbanization

1. Introduction:

Human alterations to land use and land cover are important drivers of global environmental change by invoking large perturbations to the terrestrial carbon (C) cycle and surface energy dynamics (Arnfield, 2003; Georgescu et al. 2014; Houghton et al., 2012; Ramankutty & Foley, 1999). Fortunately, global rates of deforestation have stabilized or are declining (Food and Agriculture Organization, 2010), however, human settlements are rapidly expanding (Seto et al., 2012) and becoming the new face of land use and land cover change (LULCC). The spatial extent of many of the world's largest cities increased 16-fold during the 20th century (Angel et al., 2011) due to rapid population growth and migration of people from rural areas to cities (Grimm et al., 2008). Urban lands now cover ~3% of the global land area (Liu et al., 2014) and are expanding twice as fast as their populations (Angel et al., 2010; Angel et al., 2011). The extent of urban land cover is expected to triple between 2000 and 2030 (Seto et al., 2012), but declining densities of metropolitan areas may expedite growth rates (Angel et al., 2011). The United States (U.S.) has the largest urban extent of any country (112,000 km^2; Angel et al., 2011) and developed land is its most rapidly expanding biome (Sleeter et al., 2013; USDA, 2013). While urban areas have more than doubled between 1950 and 2000, the extent of exurban development (i.e., just beyond the urban fringe) has increased five-fold (Brown et al., 2005). Following these trends, by 2025 developed land is projected to comprise 9.2% of the contiguous U.S., an area nearly the size of Texas (Alig et al., 2004).

Expansion of human settlements is of growing concern because it results in complex patterns of intermixed vegetated and impervious surface areas and ecosystem fragmentation that introduce large, and often permanent, shifts in the biophysical composition of the global landscape. For example, human settlements can convert landscapes from a sink to source of C to the atmosphere by reducing biogenic C uptake and increasing fossil fuel combustion (Imhoff et al., 2004; Hutyra et al., 2011). Similarly, shifts in albedo following expansion of human settlements can alter the energy balance and climate at local, regional and even continental scales (Menon et al., 2010; Oke, 1973).

Human settlements are increasingly being recognized as an important part of the terrestrial C cycle (Churkina et al., 2010; Hutyra et al., 2014; Pataki et al., 2006), but their effects can be difficult to quantify due to the heterogeneous nature of development and associated impacts on biogenic C fluxes. In mesic environments, expansion of human settlement tends to reduce vegetation biomass and C storage. For example, Raciti et al. (2014) found that biomass C in the City of Boston, Massachusetts was 75% lower than an intact forest, but there was considerable spatial variation within the city driven by variations in development intensity. Similar effects of development on C storage were observed in the Seattle, Washington where biomass declined over time and with proximity to the urban core (Hutyra et al., 2010). In contrast, expansion of human settlements in arid environments can increase C storage when native vegetation is replaced with trees and lawns (e.g., Golubiewski, 2006).

Growing conditions are also often altered as a landscape is developed. Cultural practices such as watering (Mini et al., 2014) and fertilization as well as increased nitrogen deposition (Rao et al., 2014), CO_2 fertilization (Idso et al., 1998) and a longer growing season associated with the urban heat island effect (Yang et al., 2013) can potentially increase productivity of vegetation in developed landscapes. In contrast, elevated exposure to pollutants such as ozone can reduce productivity (Gregg et al., 2003). While little is known about how these factors interact to affect tree growth, recent work suggests that the productivity of trees can double when the surrounding land is developed (Briber et al., 2015). Across large geographic areas, vegetation biomass and C assimilation generally decrease with increasing development intensity (Zhao et al., 2012) and urbanization has been estimated to reduce U.S. national annual net primary productivity (NPP) by 1.6%, compared to the pre-urban era (Imhoff et al., 2004).

As human settlements expand, vegetation and other natural land covers are replaced with roads, sidewalks, buildings and parking lots. This process creates a mosaic of surfaces with differing albedo characteristics, which in aggregate, can change the surface energy dynamics of the landscape (e.g., Georgescu et al. 2014; Sleeter et al., 1995). For example, LULCC between 1973 and 2000 was estimated to reduce the albedo of the contiguous U.S. (Barnes & Roy, 2010). However, albedo values across the continuum of surfaces that exist within a developed landscape can vary by 50% (Barnes & Roy, 2010; Sailor et al., 1995). As a result, expansion of human settlements can warm or cool the local or regional climate depending on the relative abundance and distribution of different surfaces (Kong et al., 2014).

While the expansion of human settlements clearly affects the terrestrial C cycle and surface energy budgets at local to global scales, most of the developed land that will exist by 2050 has yet to be built. While this may mean that the largest impacts of development are yet to come, there is also the opportunity for scientists, policymakers and land managers to shape the form and magnitude of these impacts (Georgescu et al. 2014; Lawler et al. 2014). In recent years, several studies have improved our understanding of the potential impacts of future human settlement expansion on U.S. land covers across a range of development trajectories obtained from the IPCC Special Report on Emissions Scenarios (Bierwagen et al., 2010; Nakicenovic & Swart, 2000; Sohl et al., 2012; Sohl et al., 2014), econometric models (e.g., Radeloff et al., 2012; Strengers et al., 2004), projections of cropland demand (e.g., Lawler et al., 2014) and recent patterns of development (e.g., Thompson et al., 2011). However, these studies did not explicitly project changes in C fluxes, surface energy dynamics and global warming potential (GWP) associated with expansion of human settlements. Seto et al. (2012) projected changes in the global extent of urban areas, but primarily focused on the C implications of urbanization in tropical regions.

The objectives of this study are to a) present an approach to quantifying the effects of projected changes in human settlements on terrestrial C storage and fluxes, and surface albedo at a spatial resolution sufficient to aid in policy decision making at the municipal scale, and b) assess the GWP of these biophysical changes to the landscape. We integrate nationally available datasets on land cover and forest biometrics with the U.S. Environmental Protection Agency's (EPA) Integrated Climate and Land Use Scenarios (ICLUS) model (Bierwagen et al., 2010). The state of Massachusetts located in the northeastern U.S. is used as an initial case study to develop this approach because of the existence of high quality data sets, rapid rates of development in recent history and its high proportions of both forested and developed land covers, Massachusetts is simultaneously the eighth most forested and third most densely populated state in the U.S.

2. Methods:

2.1. Study Area and Land Use and Land Cover History

Massachusetts has a population of 6.7 million people and five cities with more than 100,000 people with Boston being the most densely populated city (5,151 people km^{-2}; U.S. Census Bureau). Massachusetts has a humid, continental climate characterized by warm summers and cold, snowy winters with seasonal temperature ranges generally increasing from east to west. The capital city, Boston is located on the east coast of the state has mean monthly temperatures of -1.7° C in January and 23.3° C in July and receives approximately 1,100 mm of precipitation, evenly distributed throughout the year (National Climatic Data Center, 2010). Mixed-deciduous temperate forest is the dominant natural land cover type.

Massachusetts, similar to most of the eastern U.S., was nearly entirely forest prior to European colonization (ca. 1600), but rapid agricultural expansion reduced forest cover to <30% of the land area by the middle of the 19^{th} century (Foster & Aber, 2004; Jeon et al., 2014; Fig. 1). Agricultural abandonment allowed forest cover to increase during the latter half of the 19^{th} century, in parallel with population growth, until it peaked at nearly 70% of the land area in 1970. A new wave of deforestation began around 1970 as expansion of human settlements began to directly compete with forests for land. Between 1971 and 1999 forest cover declined by ~4,000 ha yr^{-1} (U.S. Forest Service Forest Inventory and Analysis Program; FIA), almost entirely for development of residential housing (Thompson et al., 2014). This trend continued in the 1990s and 2000s; Jeon et al. (2014) estimated the rate of forest loss driven by residential and commercial development to be 3,100 ha yr^{-1} between 1990 and 2000, and 1,700 ha yr^{-1} between 2000 and 2005. Currently, the dominant land cover types (as a proportion of total land area excluding water) are forest (65%) and developed (26%), with forest cover declining from west to east, inversely related to population density (Fig. A.1; Fig. 2a,b). Agricultural land comprises 7% of total land area.

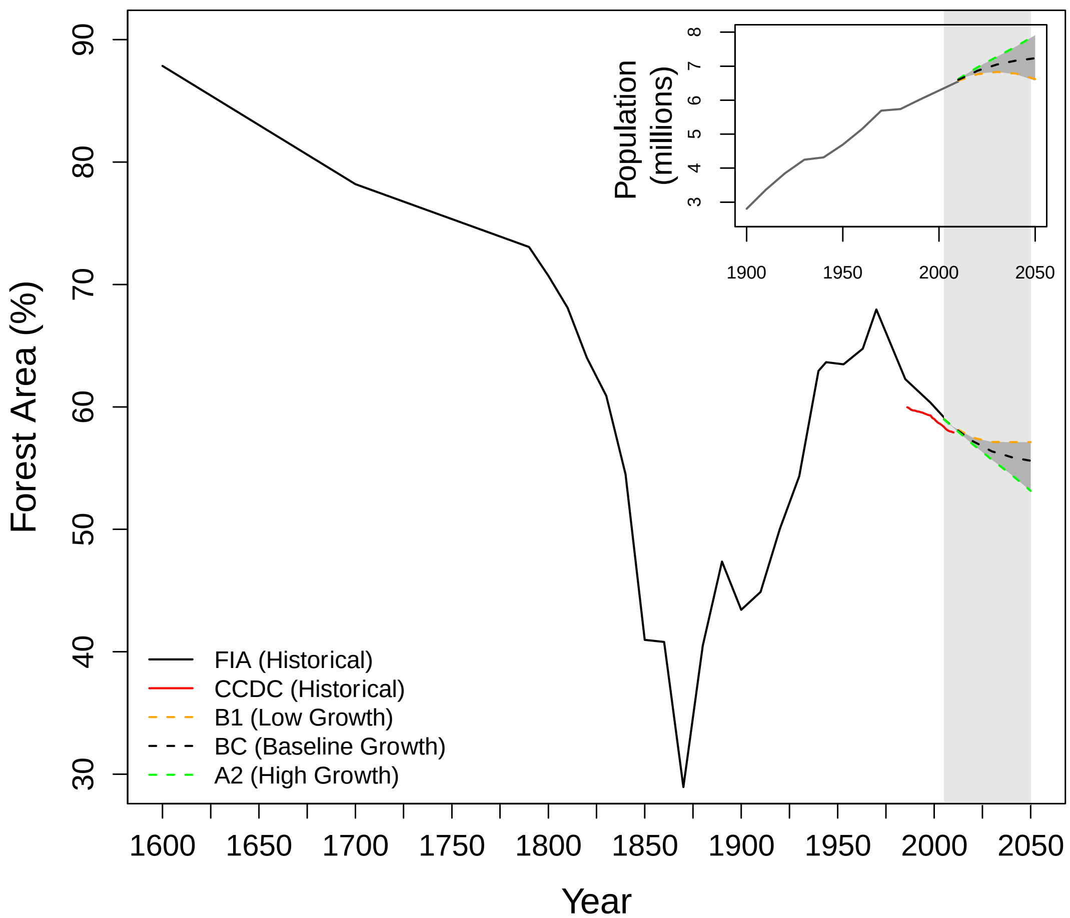


Figure 1. Historic and projected future changes in forest cover and population (inset) in Massachusetts. Note, data presented here from the Continuous Change Detection and Classification (CCDC) algorithm represent the area of forest lost to development and not the net change in total forest area. Light grey shading highlights the time period modeled

and the dark grey shading spans the projected range in change in forest area and population across the development trajectories.

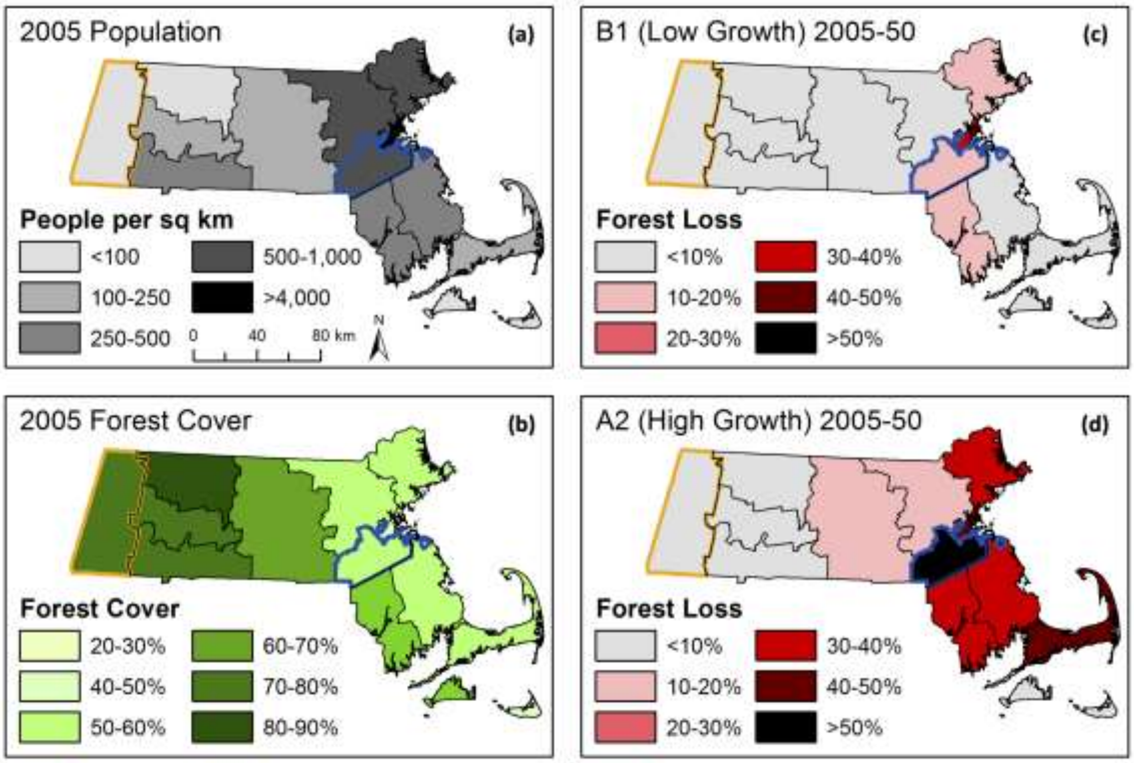


Figure 2. County-level 2005 population density (a), 2005 forest cover (b) and loss of forest area between 2005 and 2050 for the B1 (low growth; c) and A2 (high growth; d) development trajectories. Rural Berkshire County and rapidly developing Norfolk County are outlined in bold orange and blue lines, respectively.

2.2. Historical and Projected Changes in Human Settlements

We used rates of forest lost to development estimated from satellite data and reference observations as independent sources for comparison with the ICLUS-based projections in land cover change. Time series of Landsat data at 30 m resolution covering

Massachusetts were analyzed using the Continuous Change Detection and Classification (CCDC) algorithm (Zhu & Woodcock, 2014) to map the annual rate of forest loss and residential development between 1986 and 2012 (data through 2010 are used in the present study; Olofsson et al., submitted). The map was used to stratify a sample of reference observations from which rates of land cover change were estimated using stratified estimation (Olofsson et al., 2013; Olofsson et al., 2014).

Development through 2050 was modeled using the U.S. EPAs ICLUS; ICLUS is freely available online (www.epa.gov/ncea/global/iclus) and has been previously described by Bierwagen et al. (2010). This model uses county-level population growth projections from the U.S. Census Bureau, standard demographic approaches and the Spatially Explicit Regional Growth Model to develop five different scenarios to project changes in housing density in the U.S. at a 1 ha resolution. These scenarios follow the main storylines of the IPCC's Special Report on Emissions Scenarios (Nakicenovic & Swart, 2000). The model blocks out land that is considered undevelopable due to legal (e.g., parks and conservation land) or land cover (e.g., water) restrictions (area modeled by ICLUS covers 73% of Massachusetts and is hereafter referred to as 'ICLUS domain') and divides the rest of the country into 13 different housing density categories broadly designated as 'rural' (categories 1-4), 'exurban' (categories 5-8), 'suburban' (categories 9-10) and 'urban' (categories 11-13). Using the ICLUS tools for ArcGIS (v.10.1; ESRI 2012), we ran the model from 2005 to 2050 using the baseline growth (BC), high growth (A2) and low growth/sustainability focused (B1) development trajectories, which together encompass the range in housing density projections. The first projected year is 2010 and we ran at 10-year time steps thereafter. All projected changes in land cover and

C pools/fluxes refer to only the land area within the Massachusetts ICLUS domain, unless otherwise noted.

2.3. Inferring Land Cover Change from ICLUS Housing Densities

Using ArcGIS, we intersected the 2006 National Land Cover Database (NLCD; Jin et al. 2013) 30 m resolution land cover layer with the 2005 ICLUS housing density layer to empirically define land cover composition for each of the 13 different ICLUS housing density categories (Fig. A.2a,b). NLCD uses the Anderson Land Cover Classification System to define 20 land covers, which we consolidated into five categories: forest, agriculture (i.e., grassland, pasture and cropland), urban/developed, water and other. Land cover composition of each housing density category (i.e., coverage of each land cover as a proportion of land area, excluding water) was quantified separately for each of the 14 counties in Massachusetts. Land cover proportions within a given housing density were assumed to remain static over time and future changes in land cover were inferred by applying these land cover proportions to ICLUS projected changes in housing density. This approach tended to underestimate the annual rate of forest lost to development obtained from the NLCD 2001 and 2006 land cover layers. To adjust for this bias we multiplied the ICLUS-based projected changes in forest area using county-specific scalars (Table A.1) that were calculated by dividing the NLCD-derived rate of forest lost to development by the ICLUS 2005 to 2010 estimate.

2.4. Terrestrial Carbon Pools and Fluxes

We developed an empirically based bookkeeping approach, similar to those used by Houghton et al. (1983; 1999), to model the effects of timber harvesting and projected changes in land cover on aboveground terrestrial C storage and fluxes. The model tracks 10 different C pools including C accumulation in biomass of forests and human settlements and C losses associated with timber harvesting and conversion of forests to human settlements (Fig. 3). Conversion of agricultural land to human settlements comprises a small proportion of the land developed and we assume no net decline in aboveground C storage from this land conversion trajectory following Hutyra et al. (2011).

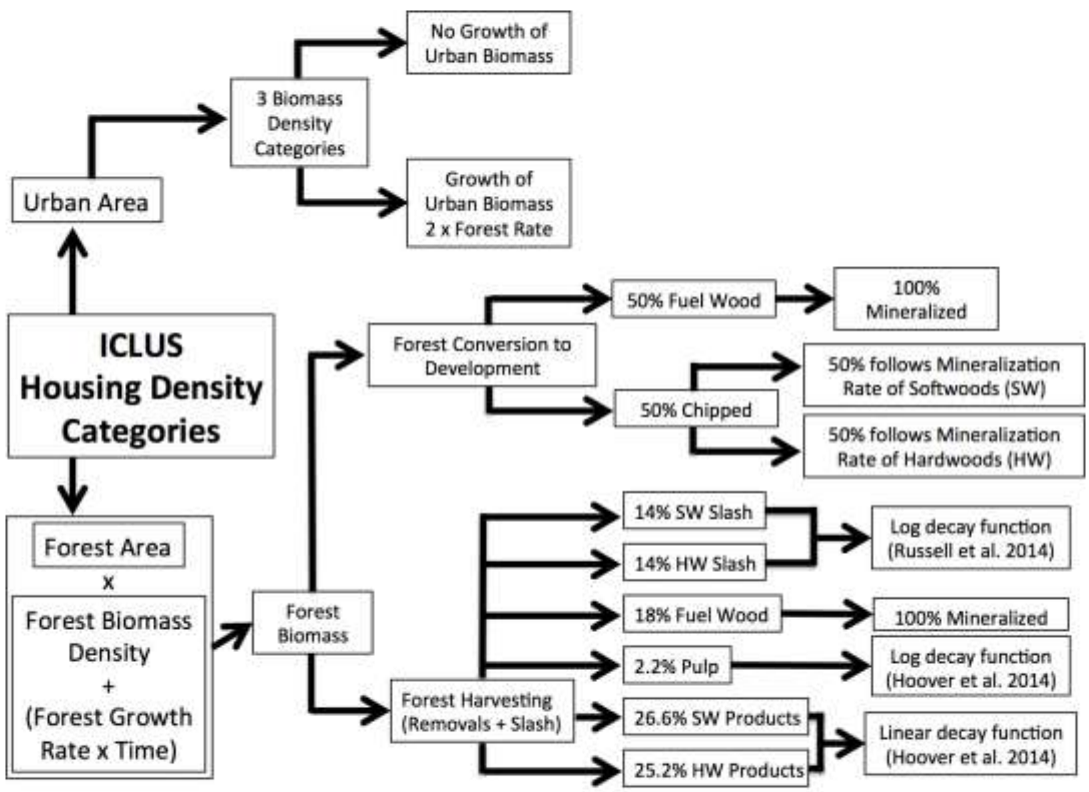


Figure 3. Expansion of human settlements carbon bookkeeping model framework.

Forest aboveground live tree biomass density (Mg C ha^{-1}) was obtained for each county using the Carbon OnLine Estimator tool (COLE v. 2.0; www.ncasi2.org/GCOLE/gcole.shtml; Van Deusen and Heath 2015), which is a web suite of applications that uses FIA plot-level data to generate a range of user-defined forestry statistics and C estimates (Heath 2012; Proctor et al. 2005). The COLE output includes means, sample size, and standard error for each county. Rates of C accumulation in live aboveground forest biomass between 2005 and 2050 were projected using forest growth curves that were calculated using a space-for-time substitution approach and quantifying the relationship between stand age and aboveground biomass from data extracted with the COLE tool. To account for differences in forest type and area-weighted site-indices between the 11 counties comprising Mainland Massachusetts (hereafter 'Mainland') and the three counties comprising Cape Cod and adjacent islands (hereafter 'Cape and Islands') we developed a separate growth curve for each of these two regions. The FIA forest growth rates (Table A.2) were linear for both the Mainland and Cape and Islands for the projected ranges in forest age across Massachusetts. Site indices varied widely among the three counties comprising the Cape and Islands. Therefore, forest growth rates for each county were adjusted based on how that county's site index deviated from the Cape and Islands area-weighted site index (Table A.2). For each county and time step, C density of forest aboveground live tree biomass and growth rates were multiplied by forest area to quantify changes in forest C storage and uptake, respectively. Forest biomass removed during expansion of human settlement was quantified by multiplying C density by the area of forest lost between time steps. In Massachusetts it is generally not economically viable to commercially harvest and process timber from parcels being developed and biomass removed is generally burned or chipped (D. Kittredge, personal communication). For our bookkeeping model, it was assumed that half of the biomass removed during development was burned as firewood and half was left to decompose as slash and wood chips.

Timber harvesting annually affects a small percentage of the forested area in Massachusetts (<1.5%; McDonald et al., 2006), it was included for completeness and comparison with expansion of human settlements. We assume the area harvested, intensity of harvest, and composition and fate of removals follow the patterns reported by McDonald et al. (2006).

Carbon emissions associated with biomass removed during development and harvesting were quantified for each county and time step using unique C mineralization rates for each C pool. We assumed all C in burned biomass was emitted into the atmosphere within one year. Biomass that was left onsite to decompose (i.e., slash and wood chips) as well as softwood products, hardwood products and pulp (i.e., paper) were assumed to lose mass following decay functions described by Russell et al. (2014) and Hoover et al. (2014), respectively. For all C pools that were not burned, it was assumed that two-thirds of the C was emitted into the atmosphere while the remaining one-third was converted to a long-term turnover pool (Nakane et al., 1996) that was stable for the duration of the model run.

While the expansion of human settlements removes forest biomass, some biomass remains onsite as remnant trees and planted ornamentals (Raciti et al., 2012). We assume that the developed proportion of each housing density category within a county has a biomass C density proportional to that county's forest biomass C density that is commensurate with those reported by Raciti et al. (2012) for a range in residential development intensities along an urban to rural gradient in Massachusetts. For example, the residential biomass density proportions that we applied to the 13 different ICLUS housing density categories are: 0.44 ± 0.01 for rural and exurban residential (i.e., < 0.5 housing units ha^{-1}; ICLUS categories 1 to 8), 0.36 ± 0.01 for suburban residential (i.e., 0.5 to 4 housing units ha^{-1}; ICLUS categories 9 and 10) and 0.16 ± 0.04 for urban residential (i.e., > 4 housing units ha^{-1}; ICLUS categories 11 to 13). While the effects of human settlement on the growth rates of remnant trees is poorly understood, we explored the likely range in residential tree biomass growth scenarios: a) intensive management of trees (i.e., pruning and mortality) offsets growth and results in no net C accumulation in trees of residential areas (hereafter 'No Urban Tree Growth' scenario) and b) no intensive management coupled with enhanced growing conditions in residential areas (e.g., increased light and nitrogen availability and a longer growing season) doubles the rate of net C accumulation in remnant trees relative to forest trees (hereafter '2x Urban Tree Growth' scenario; following results from Briber et al., 2015).

2.5. Fossil Fuel Emissions

County-level emissions for off-road, residential and commercial sectors were calculated using data from the US EPA 2011 National Emissions Inventory version 1 (U.S. EPA, 2013). Off-road CO_2 emissions are estimated directly and residential and commercial CO emissions were converted to CO_2 using emissions factors from the US EPA WEBFire database and the Vulcan 2.0 Methodology Documentation (Gurney et al., 2009). On-road emissions were obtained from Gately et al. (2015). Emissions from these four sectors comprised 91% of the total 2010 fossil fuel emissions in Massachusetts and were combined into one per capita value for fossil fuel emissions for each county. Fossil fuel CO_2 emissions were then projected out to 2050 as a function of population growth. We include emissions from only these four sectors because we assume they scale with population at a county-scale while emissions from other sectors such as industrial, energy production and air/sea travel do not. The net result is likely a conservative estimate of emissions, although it assumes no changes in efficiency.

2.6. Albedo

The 500 m resolution MODIS albedo product (MCD43A3) was used to quantify growing season (June 1 to August 31) albedo of each housing density category. Because ICLUS (100 m) is at a higher resolution than the MODIS albedo product we only used MODIS pixels that had at least 66% coverage of a single ICLUS housing density category to calculate mean and standard error of the albedo value for each housing density category. Changes in albedo between 2005 and 2050 were converted to radiative forcing using incoming global solar radiation data measured from 1991 to 2005 at Harvard Forest in central Massachusetts (Fitjarrald & Sakai, 1999). We assumed no geographic variation in incoming solar radiation.

2.7. Global Warming Potential

We used the BC (baseline) trajectory to calculate a first approximation of the GWP from biophysical changes to the landscape associated with projected expansion of human settlements. Following the approach in Muñoz et al., (2010) we first calculated top-of-atmosphere radiative forcing (RF_{TOA}) based on the change in surface albedo from 2005–2050 ($\Delta\alpha$), the average solar radiation as measured at Harvard Forest (158.529 W m^{-2} for whole-year average, 232.386 W m^{-2} for June-August), and assuming an average global atmospheric transmittance factor of 0.854. We then calculated the global CO_2 equivalent emissions based on the area of land affected (m^2) and RF_{TOA}, assuming a global airborne fraction of 0.48 (100 year time horizon) and 0.908 W kg CO_2^{-1} marginal radiative forcing of CO_2 emissions at current atmospheric concentrations. Emissions were normalized to the area of land affected (Mg CO_2-C-eq. ha^{-1}) and to the time horizon of the land-use change (Mg CO_2-C-eq. ha^{-1} yr^{-1}).

2.8. Uncertainty

The ICLUS model itself does not report uncertainty in projected changes in housing density, but rather provides a series of development trajectories. We quantified the uncertainty in our conversion of changes in housing density to changes in land cover, C fluxes and albedo associated with each development trajectory. Briefly, 95% confidence intervals for aboveground biomass C from the COLE tool and Raciti et al. (2012) were calculated using a normal distribution and the reported means and standard errors of these variables. Confidence intervals for land cover proportions and albedo values for each housing density category were obtained by bootstrapping 1000 times from the full sample of each housing density category. A root mean square approach was used to propagate uncertainty for each projected variable. All analysis was conducted in R version 3.0.2 (R Core Team). Unless noted otherwise, all reported errors represent 95% confidence intervals.

3. Results:

3.1. Recent Trends in Expansion of Human Settlements

Expansion of human settlements has resulted in a decline in forest cover in Massachusetts since the 1970s. The CCDC algorithm suggests that 3.4% (42,926 ± 8,883 ha) of Massachusetts' forestland was converted to residential development between 1986 and 2010 (Fig. 1). However, the rate of land conversion was not constant over time and more than doubled from 1,180 (± 244) ha yr^{-1}, between 1986-1998, to 2,397 (± 496) ha yr^{-1}, between 1998 and 2010. By comparison, our projections using the ICLUS model indicate slightly higher rates of forest conversion to development of 2,620 to 2,902 ha yr^{-1} between 2005 and 2010 (Fig. 1). Forest conversion to residential development generally increased from west to east. For example, between 1986 and 2010 < 0.02 (± 0.004)% yr^{-1} of the forest area was converted to residential development in sparsely populated Berkshire County in western Massachusetts, while 0.31 (± 0.06)% yr^{-1} of the forest area was converted in densely populated Norfolk County in eastern Massachusetts.

3.2. Projected future changes in population, housing density and land cover

Massachusetts population is projected to increase from 6.4 million people in 2005 to 7.2 (0.3% yr^{-1} increase) and 7.9 (0.5% yr^{-1} increase) million people by 2050 under the BC (baseline) and A2 (high growth) trajectories, respectively (Fig. 1). However, under the B1 (low growth) trajectory, the population is projected to peak at 6.8 million people by 2030 before slowly declining to 6.6 million people by 2050. Within the state, the population of Berkshire County in western Massachusetts is projected to decrease by up to 44% (54,000 people; -1 % yr^{-1}) while the population of Norfolk County in eastern Massachusetts is projected to increase by up to 10% (70,000 people; 0.2 % yr^{-1}).

In 2005, most of the land area within the ICLUS domain of Massachusetts was comprised of exurban housing densities (61%) followed by suburban (20%), rural (12.8%) and urban (4.8%) housing density categories (Fig. 4a). The distribution of housing densities varies across the state and the rural and exurban categories are most prevalent in western Massachusetts (Fig. 4b), while suburban and urban categories dominate in eastern Massachusetts (Fig. 4c). Eastern Massachusetts is projected to develop more rapidly than western Massachusetts between 2005 and 2050.

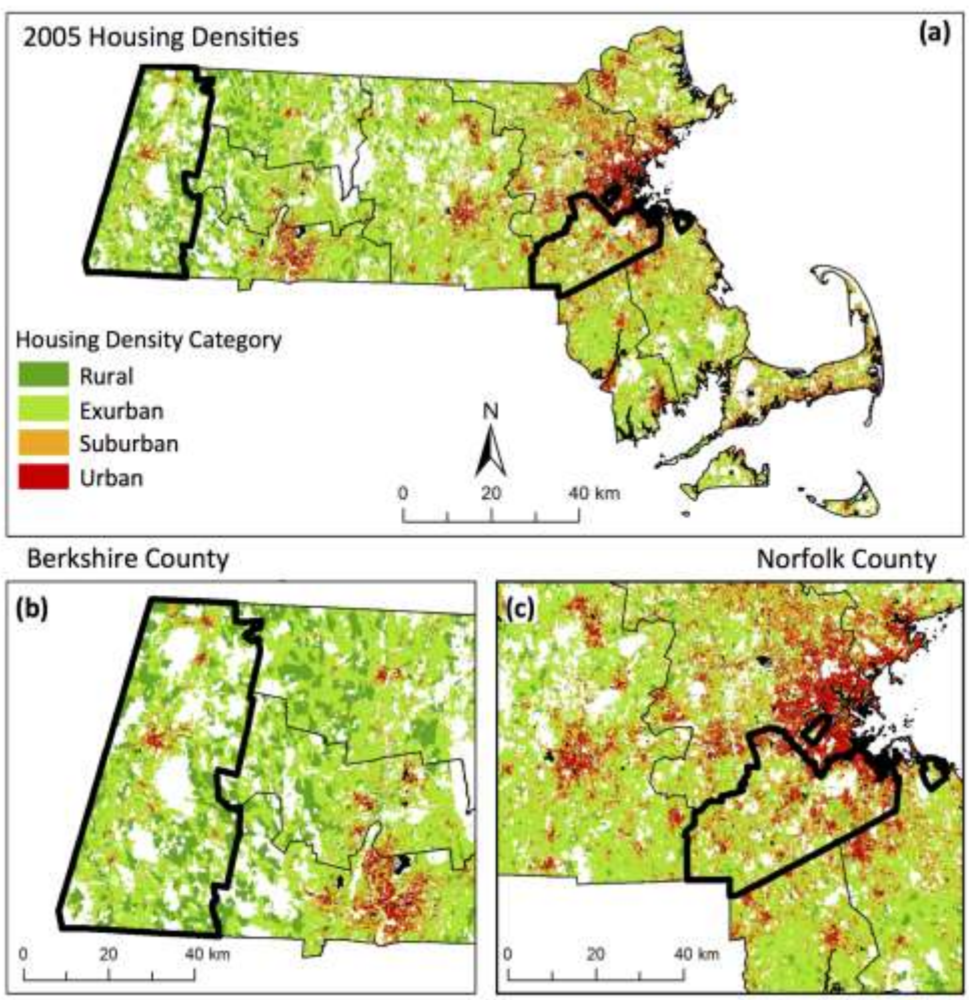


Figure 4. Distribution of 2005 ICLUS housing densities in Massachusetts (a), Berkshire County (b, outlined in bold) and Norfolk County (c, outlined in bold). Land excluded from development projections in the ICLUS model are indicated in white.

3.3. Projected changes in land cover

Development associated with expansion of human settlements between 2005 and 2050 is expected to reduce forest cover within the Massachusetts ICLUS domain by 4 to 14% across the development trajectories (Fig. 1; Table 1). Rates of forest loss and development for the B1 (low growth) and BC (baseline) trajectories are projected to be highest between 2005 and 2030, but moderate thereafter. In contrast, rates of forest loss under the high growth A2 trajectory are projected to be linear from 2005 to 2050. Because we modeled forest loss only associated with expansion of human settlements, the increase in developed land area (34% of 2005 ICLUS domain) was commensurate with forest loss reported here.

		Change from 2005-2050							
	Scenario	Forest Cover (ha)		Forest Biomass (Tg C)		Urban Biomass (NG*; Tg C)		Urban Biomass (2x**; Tg C)	
Massachusetts	B1	-39,118 ± 670	(-0.09% yr^{-1})	+31.1 ± 2.6	(+1.1% yr^{-1})	+1.21 ± 0.10	(+0.3% yr^{-1})	+13.3 ± 3.40	(+2.8% yr^{-1})
	BC	-70,955 ± 1,240	(-0.18% yr^{-1})	+27.8 ± 2.3	(+1.0% yr^{-1})	+2.13 ± 0.20	(+0.5% yr^{-1})	+13.9 ± 3.50	(+3.0% yr^{-1})
	A2	-122,630 ± 2,233	(-0.31% yr^{-1})	+22.4 ± 1.9	(+0.8% yr^{-1})	+3.48 ± 0.40	(+0.7% yr^{-1})	+14.7 ± 3.70	(+3.1% yr^{-1})
Berkshire County	B1	-297 ± 2.7	(-0.01% yr^{-1})	+4.9 ± 0.5	(+1.1% yr^{-1})	+0.01 ± 0.001	(+0.05% yr^{-1})	+0.54 ± 0.19	(+2.3% yr^{-1})
	BC	-551 ± 5.0	(-0.01% yr^{-1})	+4.9 ± 0.5	(+1.1% yr^{-1})	+0.01 ± 0.002	(+0.05% yr^{-1})	+0.54 ± 0.19	(+2.3% yr^{-1})
	A2	-530 ± 4.8	(-0.01% yr^{-1})	+4.9 ± 0.5	(+1.1% yr^{-1})	+0.01 ± 0.002	(+0.05% yr^{-1})	+0.54 ± 0.19	(+2.3% yr^{-1})
Norfolk County	B1	-3,860 ± 158	(-0.2% yr^{-1})	+0.8 ± 0.2	(+0.8% yr^{-1})	+0.12 ± 0.04	(+0.25% yr^{-1})	+1.3 ± 0.30	(+2.7% yr^{-1})
	BC	-7,259 ± 320	(-0.5% yr^{-1})	+0.4 ± 0.2	(+0.4% yr^{-1})	+0.23 ± 0.07	(+0.48% yr^{-1})	+1.4 ± 0.31	(+2.8% yr^{-1})
	A2	-16,958 ± 1,101	(-1.0% yr^{-1})	-0.7 ± 0.4	(-0.7% yr^{-1})	+0.51 ± 0.15	(+1.06% yr^{-1})	+1.5 ± 0.32	(+3.1% yr^{-1})

*'NG' refers to the No Urban Tree Growth scenario

**'2x' refers to the 2x Urban Tree Growth scenario

Table 1. Changes in forest cover and aboveground biomass within the ICLUS domain between 2005 and 2050. Values are means ± 95% confidence intervals associated with the conversion of projected changes in housing density to forest area, urban area and

biomass. Parenthetical values represent the mean annual rate of change between 2005 and 2050.

Similar to patterns of changes in housing density, the highest rates of forest loss are projected to occur in eastern Massachusetts with little change in western Massachusetts (Fig. 2c,d). Counties in eastern Massachusetts are projected have light to moderate forest loss (<10%) between 2005 and 2050 under the B1 trajectory, but high rates of forest loss (>20%) under the A2 trajectory. In particular, Norfolk County is projected to lose 13% (3,861 ± 444 ha) to 56% (16,959 ± 1,952 ha) of its forested land (Fig. 2c,d). In contrast, forest area in Berkshire County is projected to decline by <0.5% (530 ± 41 ha).

3.4. Effects of expansion of human settlements on terrestrial C cycle

At the start of the model run in 2005, 86% of aboveground C storage within the ICLUS domain in Massachusetts was in forest biomass (63.3 ± 4.4 Tg C) while the remaining 14% was within human settlements (10.5 ± 0.8 Tg C). Aboveground biomass within human settlements comprised a substantial proportion of aboveground C storage in eastern Massachusetts, but played only a small role in aboveground C storage in rural western Massachusetts. For example, in Norfolk County, 32 ± 1.3% of the 3.3 ± 0.5 Tg C was stored in aboveground biomass of human settlements compared to 5.1 ± 0.1% of the 10.1 ± 1.1 Tg C in Berkshire County.

Our analysis indicates a net increase in forest aboveground biomass within the ICLUS domain of 35 ± 0.9% to 49 ± 2.9% between 2005 and 2050 across all development trajectories and that accumulation of forest biomass from tree growth will outpace losses in forest biomass from land conversion (Table 1). Rates of aboveground C sequestration by forest biomass are projected to decline from 0.53 ± 0.01 Mg C ha^{-1} land yr^{-1} in 2005 to between 0.46 ± 0.01 (A2) and 0.50 ± 0.01 (B1) Mg C ha^{-1} land yr^{-1} in 2050 (Fig. 5a). High forest cover and slow rates of forest lost to development in Berkshire County are projected to result in a strong C sink relative to the more rapidly developing Norfolk County (Fig. 5a).

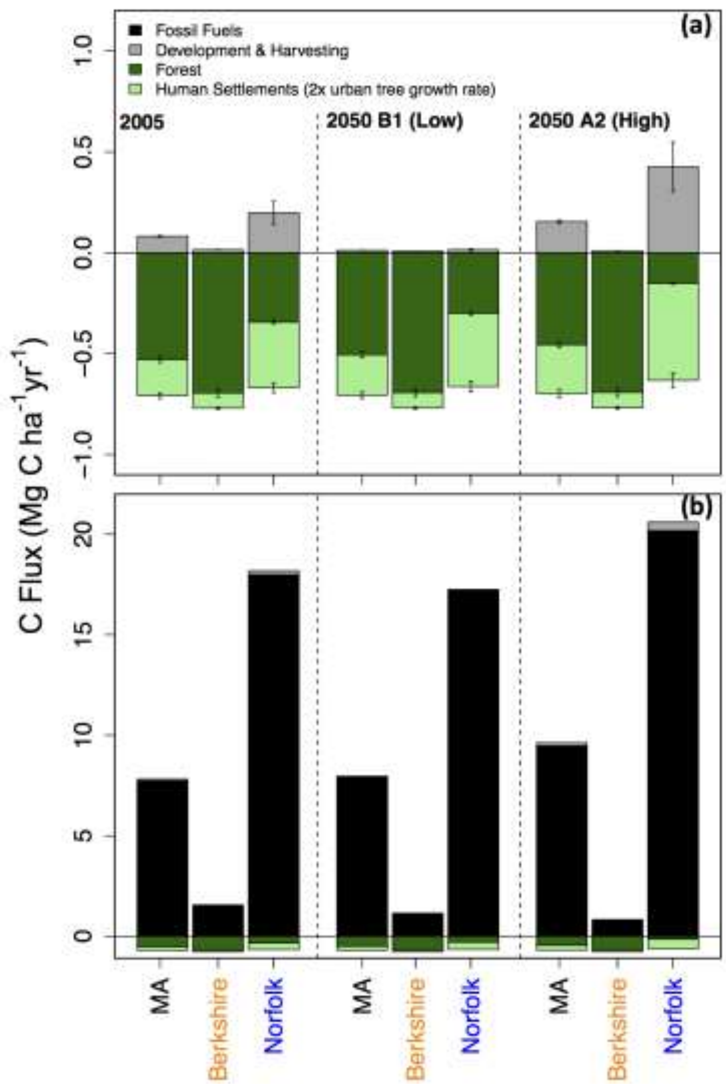


Figure 5. 2005 and 2050 annual rates of C uptake by aboveground biomass in forests and human settlements and C emissions from fossil fuel combustion and decomposition of biomass removed during harvesting and development. Panel (a) includes only biogenic fluxes to highlight changes associated with land cover change from development. Panel (b) also includes C emissions from fossil fuel combustion, which are 1-2 orders of magnitude larger than biogenic fluxes. Values are fluxes ± 95% confidence intervals associated with the conversion of projected changes in housing density to C fluxes.

Aboveground biomass C storage within human settlements is projected to increase by 11.5 ± 0.01 to 140 ± 35% across development trajectories and urban tree growth scenarios (Table 1) with rates of biomass C sequestration within human settlements increasing from 0.18 ± 0.01 Mg C ha^{-1} yr^{-1} in 2005 to up to 0.24 ± 0.02 Mg C ha^{-1} yr^{-1} for the 2x Urban Tree Growth scenario (Fig. 5a). There was considerable geographic variation within the state, which followed both spatial patterns of development and amount of aboveground biomass within human settlements at the start of the model run. Low rates of development and a small area of human settlements resulted in low rates of C sequestration by aboveground biomass in human settlements of Berkshire County compared to Norfolk County, which had both high area of human settlements and rates of development (Fig. 5a).

Between 2005 and 2050, forestland is projected to remain the largest C sink and reservoir of aboveground C under all development trajectories, but vegetation in human settlements is expected to comprise up to 35% of the annual aboveground terrestrial C sink under the 2x Urban Tree Growth scenario (Table 1; Fig. 5a). Across development trajectories, state C emissions associated with the burning and decomposition of aboveground biomass removed during development are projected to be equivalent to 3 ± 0.001% to 34 ± 0.02% of the aboveground forest C sink (Fig. 5a). By 2050, C emissions associated with land cover change are projected to be negligible relative to the mean annual forest C sink in Berkshire County, but will be up to nearly three times the annual forest C sink in Norfolk County (Fig. 5a).

The net terrestrial biogenic C flux (i.e., sequestration in forest and human settlement aboveground biomass minus emissions from land cover change) in Massachusetts is projected to decline from 0.63 ± 0.03 in 2005 to as low as 0.54 ± 0.04 Mg C ha^{-1} yr^{-1} under the 2x Urban Tree Growth scenario. Expansion of human settlements is projected to have little effect on terrestrial C sequestration in rural western Massachusetts, and Berkshire County is expected to continue to be a strong C sink between 2005 and 2050 (Fig. 5a). In contrast, under the A2 (high growth) development trajectory Norfolk County in eastern Massachusetts is projected to become a weak C sink. For comparison, fossil fuel CO_2 emissions in Massachusetts are projected to increase from 7.4 Mg C ha^{-1} yr^{-1} in 2005 to up to 8.7 (A2) Mg C ha^{-1} yr^{-1} by 2050 based on population growth trajectories (Fig. 5b). Geographic variability in projected fossil fuel emissions follows patterns in population density and growth (Fig. 5b).

3.5. Global warming potential of biophysical changes to the landscape

The MODIS albedo data indicate a decline in growing season albedo from 0.1471 ± 0.0008 (housing density category 1) to 0.1001 ± 0.0018 (housing density category 13; Table A.3) with increasing housing density. Massachusetts land surface albedo was 0.1389 ± 0.0008 in 2005 and is projected to decline by 0.3 ± 0.002 to 0.8 ± 0.006% across development trajectories by 2050. The land surface albedo of Norfolk County in 2005 was 0.1302 ± 0.0011 and is projected to decline by 0.21 ± 0.003 to 0.97 ± 0.01% by 2050 across development trajectories. In contrast, the land surface albedo of Berkshire County in 2005 was 0.1428 ± 0.0005 and is projected to decline by less than 0.001% by 2050.

The mean growing season incoming global solar radiation was 206.3 W m^{-2} resulting in absorption of 176 ± 0.9 to 185.6 ± 3.3 W m^{-2} for housing density categories 1 and 13, respectively. Massachusetts land surface absorption of incoming solar radiation during the growing season was 177.6 ± 1.0 W m^{-2} in 2005 and is projected to increase by 0.14 ± 0.001 to 0.23 ± 0.002 W m^{-2} by 2050. Absorption of incoming solar radiation in 2005 was higher in Norfolk County (179.4 ± 0.4 W m^{-2}) than Berkshire County (176.8 ± 0.5 W m^{-2}). Absorption in Norfolk County is projected to increase to up to 0.26 ± 0.003 W m^{-2}.

Expansion of human settlements in Massachusetts reduced the strength of the forest C sink, provided a source of C emissions to the atmosphere from biomass removals and reduced land surface albedo. For the BC (baseline) trajectory, these biophysical changes to the landscape resulted in a GWP of +0.13 Mg CO_2-C-eq. ha^{-1} yr^{-1} between 2005 and 2050 (Fig. 6). There was large geographic variability within the state ranging from negligible change in Berkshire County to a GWP of +0.3 Mg CO_2-C-eq. ha^{-1} yr^{-1} in Norfolk County (Fig. 6). These GWPs are equivalent to 17% and 70% of the GWP associated with the projected increases in fossil fuel emissions in Massachusetts and Norfolk County, respectively. While reductions in the forest C sink and C emissions associated with biomass removals often made the largest contributions to GWP, declines in albedo made significant contributions across much of the state and comprised the entire GWP associated with expansion of human settlements in Suffolk County (i.e., Boston; Fig. 6).

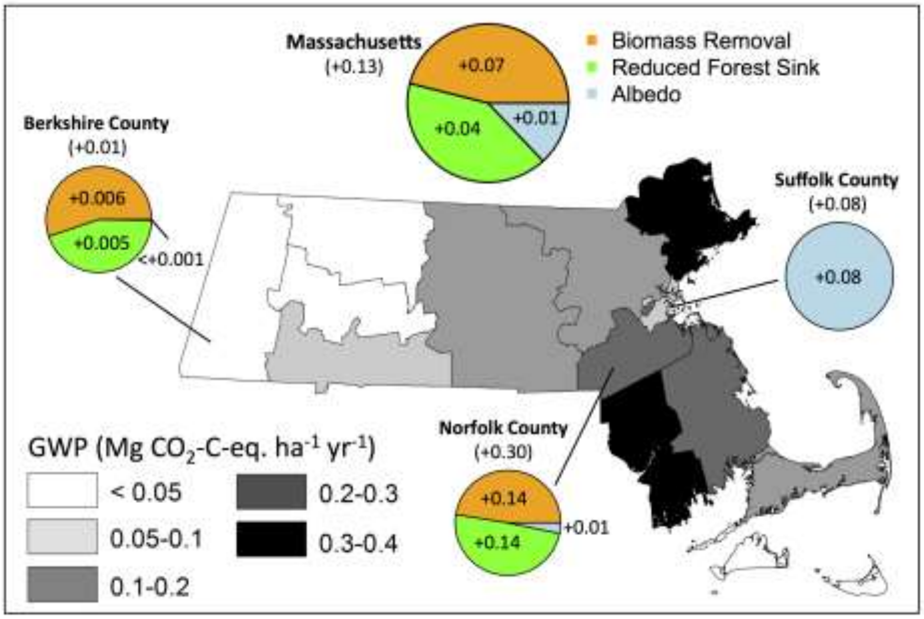


Figure 6: Global warming potential associated with expansion of human settlements between 2005 and 2050 in Massachusetts and each county for the BC (baseline) development trajectory. Values reflect the changes in GWP relative to a scenario without land use and land cover change between 2005 and 2050. Pie charts indicate the composition of increased global warming potential.

4. DISCUSSION:

Expansion of human settlements is a globally important driver of land cover change that occurs at twice the rate of population growth (Angel et al., 2010; Angel et al., 2011). During the 20th century these changes in land cover have reduced the strength of the terrestrial C sink (Imoff, 2004) and increased absorption of incoming solar radiation (Barnes & Roy, 2010). This study presents a novel approach to projecting the biophysical changes in the landscape associated with population growth and expansion of human settlements using nationally consistent and available data sets. Our results indicate strong geographic variability in the projected changes to the biophysical nature of the landscape in response to development, even within a small state like Massachusetts. While rates of land cover change are projected to be most rapid in tropical regions (Seto et al., 2012), we show that even in temperate regions with modest projected rates of population and urban growth, expansion of human settlements can significantly weaken the forest C sink. However, our results also suggest that vegetation within human settlements can be an important C sink in developed and rapidly developing landscapes and mitigate declines in the terrestrial C sink associated with forest loss. Further, we demonstrate that expansion of human settlements can make a significant contribution to changes in the total GWP (biophysical + fossil fuel) associated with population growth.

4.1. Historical and projected patterns of land cover change

During the latter half of the 20th century, the most important component of the terrestrial C sink in the conterminous United States was regrowth of eastern forests following agricultural abandonment (Birdsey & Heath, 1995; Goodale et al., 2002; Heath & Birdsey, 1993). Forest cover in Massachusetts has been declining since the 1970s and our results indicate that nearly all of the land that was developed between 1986 and 2010 displaced forest. Further, rates of development have been increasing over time, which is similar to patterns observed throughout the eastern U.S. (Drummond & Loveland, 2010). Our estimate of forest loss between 1986 and 2010 is about 40% lower than the FIA estimate, but this is likely due to differences in methodologies and definitions of “forest” (see discussion by Drummond & Loveland, 2010).

The most rapidly expanding housing densities in Massachusetts are projected to be suburban followed by urban categories between 2005 and 2050, which parallels projections for the entire U.S. (Bierwagen et al., 2010). Throughout Massachusetts, forest cover comprises nearly all of the land that is currently undeveloped. Following recent trends in Massachusetts (Nowack et al., 2005) and the eastern U.S. as a whole (Drummond & Loveland, 2010), our results indicate that forests will continue to be the land cover most impacted by expansion of human settlements between 2005 and 2050. However, similar to Thompson et al. (2011), our projections suggest large geographic variability in the rate and extent of forest loss that follow patterns of population growth. Rates of forest lost to development projected here for Massachusetts are 3 to 11 times lower than rates projected by Nowak & Walton (2005), but differences between these estimates are due to different definitions of 'forest loss'. We defined forest loss as forestland that is converted to a developed land cover, while Nowack & Walton (2005) broaden this definition to include forestland that becomes engulfed by an urban census block as defined by the US Census Bureau (i.e., >195 people km^{-2}).

4.2. Biophysical implications of land cover change

4.2.1. Carbon

As the dominant land cover type, it was not surprising that forests comprised the largest pool of aboveground biomass in Massachusetts in 2005. Our estimate of 63.3 ± 4.3 Tg C for forests within the ICLUS domain is proportionally consistent with previously reported biomass estimates for the entire state (Thompson et al., 2011). While C stored in the aboveground biomass of human settlements is often omitted from estimates of aboveground C pools, recent studies have found that these developed areas can make significant contributions to aboveground C storage (e.g., Raciti et al., 2012). Similarly, we found that omitting biomass within human settlements would have underestimated the terrestrial aboveground C pool by 14%.

All of the development trajectories presented here are expected to reduce the total forest area of Massachusetts, but forest aboveground biomass is projected to increase between 2005 and 2050 (this study; Thompson et al., 2011), indicating that tree growth in the remaining forest will outpace reductions from land conversion. However, forest aboveground biomass accumulation is expected to be nearly 50% higher under the B1 (low growth) trajectory than the A2 (high growth) trajectory. Therefore, although the forests of Massachusetts may continue to be a net C sink through 2050, expansion of human settlements may reduce the strength of this sink by up to 12%, compared to the 18% reduction projected by Thompson et al. (2011). Carbon emissions associated with forest lost to development are projected to result in important reductions in the net terrestrial C sink of Massachusetts, particularly under the A2 (high growth) trajectory. Reductions in the forest C sink between 2005 and 2050 are projected to be larger than the increase in fossil fuel emissions under the B1 (low growth) trajectory and may be equivalent to nearly one-third of the increase in fossil fuel emissions under the A2 (high growth) trajectory.

Few studies have considered the potential of biomass in residential areas to contribute to the terrestrial C sink (e.g., Briber et al., 2015; Imhoff et al., 2004). While landowner management is undoubtedly an important driver of aboveground biomass accumulation, our 2x Urban Tree Growth scenario, which provides an upper limit of the

C sequestration potential of human settlements, suggests that these areas comprise an important component of the Massachusetts terrestrial C sink. Similarly, trees in urban areas of Greater Boston have been shown to grow faster than forest trees (Briber et al., 2015) and Imhoff et al. (2004) found that NPP of urban areas in the northeastern United States (405 g m^{-2}) is only 20% lower than non-urban areas (500 g m^{-2}). Human settlements become an increasingly important component of the terrestrial C sink as their proportion of the landscape increases and we found that by 2050, vegetation within human settlements may comprise 75% of the terrestrial C sink in rapidly developing counties of eastern Massachusetts. Further, C sequestration by vegetation within human settlements can more than offset C emissions associated with losses of forest biomass during development under the B1 (low growth) trajectory. These results suggest that while deforestation associated with expansion of human settlements could substantially reduce the strength of the forest C sink, vegetation within human settlements can play an important role in mitigating overall reductions of the total terrestrial C sink.

Soil C pools comprise about half of the C stored in the forests of Massachusetts (Van Deusen and Heath 2015). Perturbations to soil C pools from human settlement expansion can have large implications for the terrestrial C balance, but characterizing the response of these pools is inherently complicated because of the heterogeneous nature of land use and land cover in urban ecosystems. Previous research indicates that soil C storage beneath impervious surfaces is 66% lower than adjacent open areas (Raciti et al. 2012). However, physical soil disturbances, anthropogenic inputs of fill materials with varying C content, and land management practices (e.g., lawn mowing) also associated with human settlement expansion interact to create open area soils with highly variable C storage that can be higher or lower than in forest soils (Pouyat et al. 2002). Currently, there is limited data to develop reliable models that characterize the response of soil C pools to human settlement expansion. Developing empirically derived constants defining the rate of change in these C pools in response to human settlement expansion would greatly advance the sophistication of urban C cycling models and should be the focus of future research.

4.2.2. Global warming potential of expansion of human settlements

Urbanization and expansion of human settlements can alter climate by changing terrestrial C fluxes and land surface albedo. We show that the projected conversion of forested to developed land covers with a lower albedo could increase growing season radiative forcing in Massachusetts up to 0.23 ± 0.09 W m^{-2} by 2050, a forcing 1.5 times as large as that associated with global N_2O emissions (0.15 ± 0.10 W m^{-2}; IPCC, 2014). Similarly, land cover change between 1973 and 2000 in the ecoregion that includes most of Massachusetts has resulted in up to a 0.004 W m^{-2} yr^{-1} increase in snow-free radiative forcing (Barnes & Roy, 2008), which is in the range of the development trajectories explored in this analysis, BC (baseline; 0.003 W m^{-2} yr^{-1}) and A2 (high growth; 0.005 W m^{-2} $yr^{-}1$).

Previous studies have treated radiative forcing due to changes in surface character as directly comparable to radiative forcing due to increased atmospheric greenhouse gas, with both expressed on the scale of C emissions (Akbari et al., 2009; Schwaab et al., 2015). However, climate modeling studies have shown that the climate impacts from surface-change radiative forcing can diverge unpredictably from similar sized forcing due to greenhouse gas emissions (Jones et al., 2013), potentially hampering their comparison. As such, the estimated radiative forcing and CO_2 emissions equivalence calculated for the projected decreases in albedo in our study can be used as an indicator of the sign and relative scale of climate disturbance contributed by expansion of human settlements in the region, but are subject to uncertainty in their global and regional effects on climate in comparison to CO_2 emissions.

Numerous studies have quantified shifts in albedo or terrestrial C storage associated with LULCC (e.g., Barnes & Roy, 2010, Houghton et al., 2012, Jones et al., 2015). However, to our knowledge no studies have compared the relative contributions of these effects of LULCC to the GWP of human settlement expansion. Our projections indicate that reductions in forest aboveground biomass, the forest C sink and albedo from expansion of human settlements result in net warming and can make a substantial contribution to the change in total GWP (biophysical + fossil fuel) associated with population growth in Massachusetts between 2005 and 2050. These findings compliment those of Georgescu et al. (2014) who found that reduced vegetation cover and evapotranspiration associated with human settlement expansion can also impart warming that is a significant fraction of anticipated warming from fossil fuel emissions. Interestingly, although changes in C fluxes associated with forest loss resulted in the largest increase in GWP associated with expansion of human settlements, we found strong spatial variability in the relative contribution made by the underlying drivers of GWP. Reductions in albedo are expected to be an important driver of GWP in densely developed regions such as Suffolk County (i.e., Boston) where shifts towards high housing densities are projected to be the dominant biophysical change to the landscape. Further, while we use GWP as a means to compare the climate perturbations associated with shifts in C fluxes and albedo, the effects of changes in albedo on local climate are likely to be much more profound than their GWP in CO_2-eq. might indicate. These results highlight the importance of accounting for the multiple facets of climate disturbance associated with expansion of human settlements, particularly when considering spatial scales of relevance to municipal policymakers.

5. Conclusions

Using Massachusetts as a case study, we provide a framework for integrating nationally consistent datasets to quantify spatially explicit biophysical implications of land cover change associated with projected expansion of human settlements. Our results indicate that expansion of human settlements can be an important driver of land cover change even in states with only moderate rates of population growth and result in positive GWP from significant changes in radiative forcing and the forest C sink. Further, by incorporating vegetation within human settlements into the modeling framework, this study imparts new insight into the role of urban vegetation in the terrestrial C cycle. In particular, our findings highlight the potential of vegetation within human settlements to mitigate declines in aboveground C storage and uptake associated with forest lost to development.

Developed land is the most rapidly expanding biome in the United States (Sleeter et al., 2013; USDA, 2013) and is projected to continue to displace large areas of forestland and other land covers throughout the 21st century (Bierwagen et al., 2010). By 2025, the extent of developed land in the contiguous U.S. is projected to comprise an area equivalent to nearly half of the countries forestland (Alig et al., 2004). As such, our results indicate that not only will expansion of human settlements likely reduce the strength of the forest C sink, but vegetation within developed areas will become an increasingly important component of the terrestrial C sink in the U.S. Further, our findings suggest that managing vegetation within human settlements as well as declines in the forest C sink and land surface albedo associated with expansion of human settlements can play an important role in the climate change mitigation strategies of states and municipalities experiencing even moderate rates of population growth.

Appendices:

Table A.1. County-level rates of forest loss obtained from NLCD (2001 to 2006) and the unadjusted ICLUS BC (baseline) development trajectory projection (2005 to 2010). Scalars applied to county-level ICLUS estimates of forest lost to development were calculated by dividing the NLCD estimate by the ICLUS estimate. Note, the largest adjustments imposed by the scalars occur in counties with low rates of forest loss (< 25 ha yr^{-1}).

	Forest Loss (ha yr^{-1})		
County	NLCD	ICLUS	Scalar
Barnstable	84.3	20.9	4.0
Berkshire	23.9	0.1	224.3
Bristol	447.1	128.7	3.5
Dukes	0.1	0.4	0.1
Essex	276.0	106.2	2.6
Franklin	5.7	0.2	34.3
Hampden	81.8	157.7	0.5
Hampshire	19.3	63.3	0.3
Middlesex	480.5	125.2	3.8
Nantucket	0.1	6.4	0.0
Norfolk	351.1	36.4	9.6
Plymouth	525.5	81.7	6.4
Suffolk	3.7	4.1	0.9
Worcester	485.6	161.9	3.0
Massachusetts	2784.7	889.2	3.1

Table A.2. Forest biomass growth rates and 95% confidence intervals used to model changes in forest biomass between 2005 and 2050 for the Mainland and Cape and Island regions of Massachusetts. Note, growth rates used for each county comprising the Cape and Islands were derived by adjusting the Cape and Islands growth rate up or down based on the site index of each county relative to the area-weighted mean of the region.

Region	Growth Rate ($MgC\ ha^{-1}\ yr^{-1}$)	r^2
Mainland	0.9 ± 0.03	0.96
Cape and Islands	0.6 ± 0.12	0.82
Barnstable County	0.7 ± 0.14	-
Dukes County	0.4 ± 0.08	-
Nantucket County	0.2 ± 0.03	-

Table A.3. Albedo and absorption of incoming solar radiation for each housing density category. Area-weighted values from 2005 for Massachusetts, Norfolk County and Suffolk County are also included. Values are means ± 95% confidence intervals.

Housing Density or Spatial Extent	Albedo	Absorption of Incoming Solar Radiation (W m^{-2})
1	0.1471 ± 0.0008	176.0 ± 0.9
2	0.1455 ± 0.0010	176.3 ± 1.2
3	0.1474 ± 0.0006	175.9 ± 0.7
4	0.1444 ± 0.0004	176.5 ± 0.5
5	0.1443 ± 0.0004	176.5 ± 0.5
6	0.1421 ± 0.0002	177.0 ± 0.2
7	0.1402 ± 0.0002	177.4 ± 0.2
8	0.1356 ± 0.0004	178.3 ± 0.5
9	0.1315 ± 0.0014	179.2 ± 1.9
10	0.1272 ± 0.0027	180.1 ± 3.9
11	0.1236 ± 0.0025	180.8 ± 3.7
12	0.1206 ± 0.0018	181.4 ± 2.7
13	0.1001 ± 0.0018	185.6 ± 3.3
Massachusetts	0.1389 ± 0.0008	177.6 ± 1.0
Norfolk County	0.1302 ± 0.0003	179.4 ± 0.4
Berkshire County	0.1428 ± 0.0004	176.8 ± 0.5

Figure A.1. Distribution of land covers within Massachusetts in 2006 (NLCD 2006).

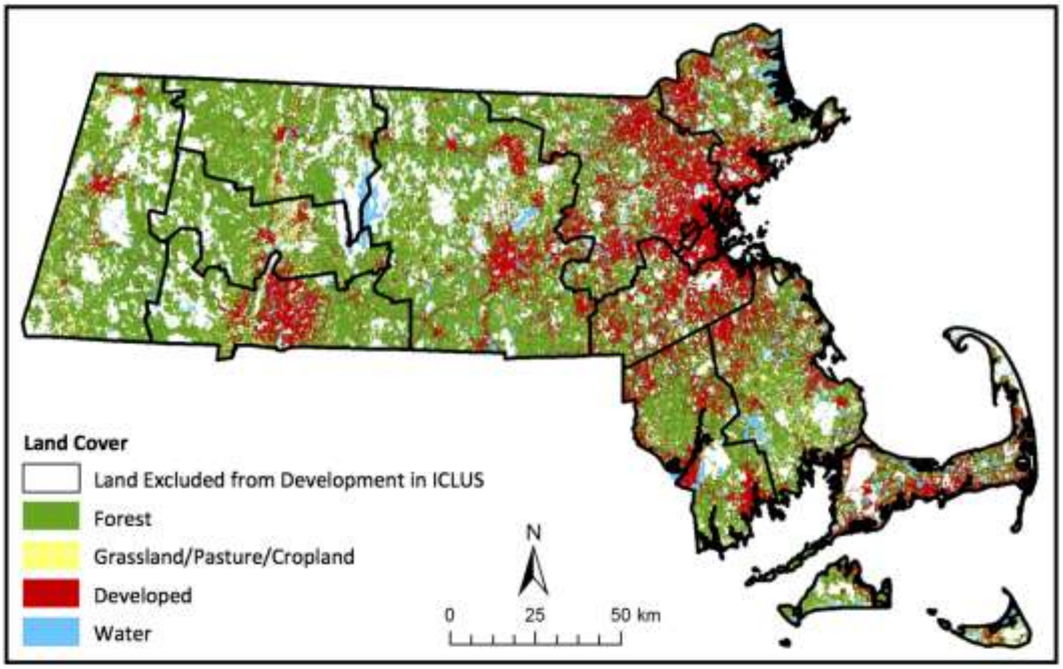
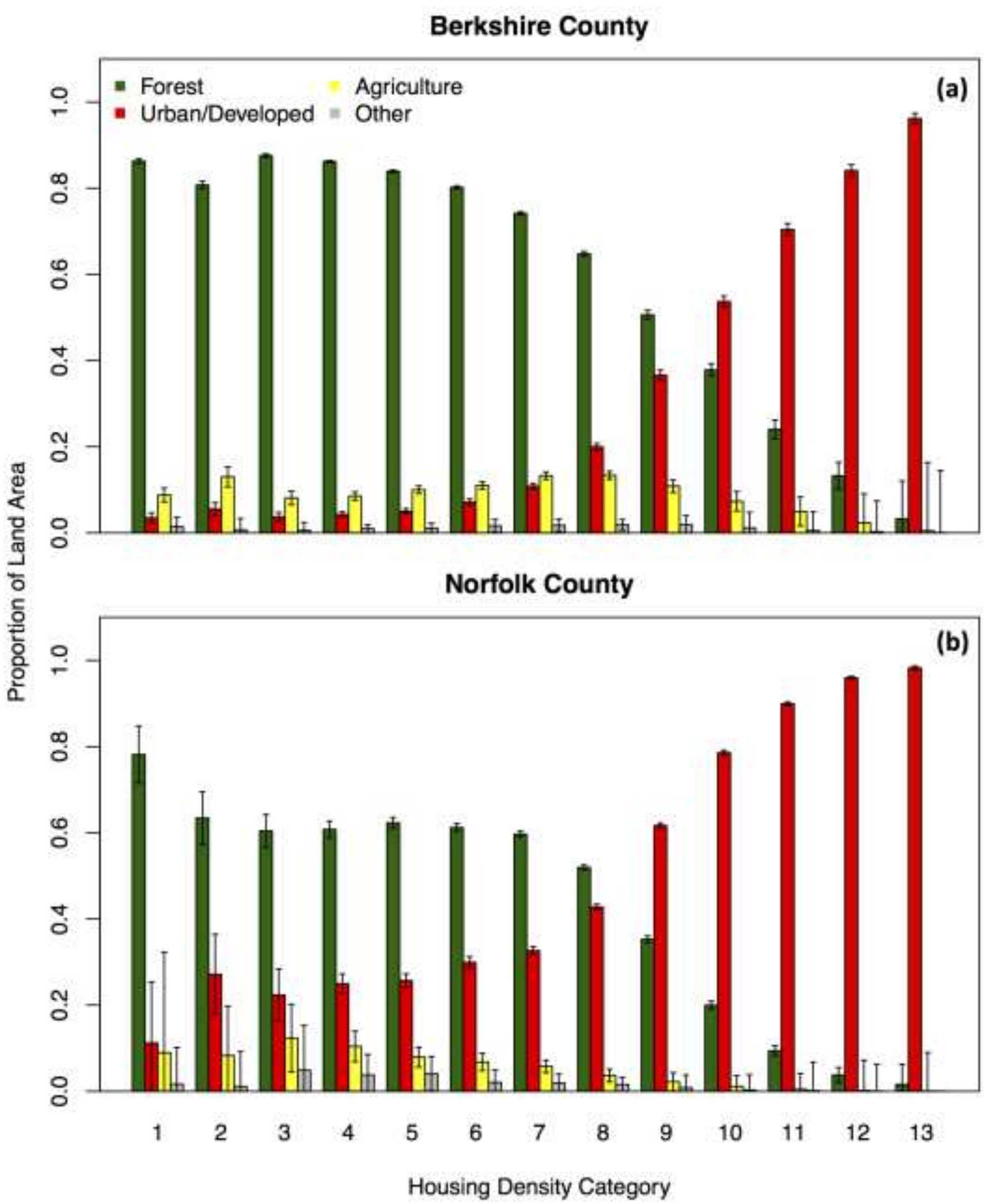


Figure A.2. Land cover composition of each ICLUS housing density category. Values are means ± 95% confidence intervals for Berkshire County (a) in rural western Massachusetts and rapidly developing Norfolk County (b) in eastern Massachusetts.



Acknowledgements:

We would like to thank Mark A. Friedl, Curtis E. Woodcock and Jackie M. Getson for providing feedback on this manuscript and Christopher Holden for assistance with data analysis. This research was funded by the National Oceanic and Atmospheric Administration (NOAA grant # NA14OAR4310179) and the National Aeronautics and Space Administration Interdisciplinary Science Program (NASA IDS grant # NNX12AM82G).

References:

Akbari H, Menon S, Rosenfeld A (2009) Global cooling: Increasing world-wide urban albedos to offset CO_2. Climatic Change, **94**, 275–286.

Alig R, Kline JD and Lichtenstein M (2004) Urbanization on the US landscape: looking ahead in the 21st century. Landscape and Urban Planning, **69**, 219-34.

Angel S, Parent J, Civco DL, Blei AM (2010) The persistent decline in urban densities: Global and historical evidence of 'sprawl'. Lincoln Institute of Land Policy, 151 p.

Angel S, Parent J, Civco DL, Blei A, Potere D (2011) The dimensions of global urban expansion: Estimates and projections for all countries, 2000-2050. Progress in Planning, **75**, 53-107.

Barnes CA, Roy DP (2008) Radiative forcing over the conterminous United States due to contemporary land cover and albedo change. Geophysical Research Letters, **35**, L09706.

Barnes CA, Roy DP (2010) Radiative forcing over the conterminous United States due to contemporary land cover land use change and sensitivity to snow and Interannual albedo variability. Journal of Geophysical Research, **115**, G04033.

Bierwagen BG, Theobald DM, Pyke CR, Choate A, Groth P, Thomas JV, Morefield P, Band LE (2010) National housing and impervious surface scenarios for integrated climate impact assessments. Proceedings of the National Academy of Sciences, **107**, 20887-20892.

Briber BM, Hutyra LR, Reinmann AB, Raciti SM, Dearborn VK, Holden CE, Dunn AL (2015) Tree productivity enhanced with conversion from forest to urban land covers. PLoS ONE, **10**, e0136237.

Brown D, Johnson KM, Loveland TR, Theobald DM (2005) Rural land-use trends in the conterminous United States 1950-2000. Ecological Applications, **15**, 1851-63.

ESRI (Environmental Systems Research Institute) (2012) ArcGIS Version 10.1. ESRI, Redlands, California, USA.

Fitzjarrald D, Sakai R (1999) Radiation Measurements at Harvard Forest EMS Tower since 1991. Harvard Forest Data Archive: HF102.

Foster DR, Aber JD (2004) Forests in Time: The Environmental Consequences of 1,000 Years of Change in New England 2004 Forests in Time. Yale University Press, New Haven.

Gately CK, Hutyra LR, Sue Wing I (2015) Cities, traffic, and CO_2: A multidecadal assessment of trends, drivers, and scaling relationships. Proceedings of the National Academy of Sciences, **112**, 4999-5004.

Georgescu M, Morefield PE, Bierwagen B, Weaver CP (2014) Urban adaptation can roll back warming of emerging megapolitan regions. Proceedings of the National Academy of Sciences, **111**, 2909-2914.

Golubiewski NE (2006) Urbanization increases grassland carbon pools: effects of landscaping in Colorado’s Front Range. Ecological Applications, **16**, 555-71.

Gregg JW, Jones CG, Dawson TE (2003) Urbanization effects on tree growth in the vicinity of New York City. Nature, **424**, 183-7.

Grimm NB, Faeth SH, Golubiewski NE, Redman CL, Wu J, Bai X, Briggs JM (2008) Global change and the ecology of cities. Science, **319**, 756-60.

Gurney KR, Mendoza DL, Zhou Y, Fischer ML, Miller CC, Geethakumar S, De La Rue Du Can S (2009) High resolution fossil fuel combustion CO_2 emissions fluxes for the United States. Environmental Science and Technology, **14**, 5535-41.

Handy S (2005) Smart growth and the transportation-land use connection: What does the research really tell us? International Regional Science Review, **28**, 146-67.

Heath LS (2012) Using FIA data to inform United States forest carbon national-level accounting needs: 1990-2010. In: *Long-term Silvicultural & Ecological Studies: Results for Science and Management, Volume 2* (eds Camp AE, Irland LC and Carroll CJW), GISF Research Paper 013. Yale University School of Forestry & Environmental Studies, Global Institute of Sustainable Forestry. p. 149-160

Hoover C, Birdsey R, Goines B, et al. (2014) Chapter 6: Quantifying Greenhouse Gas Sources and Sinks in Managed Forest Systems. In: *Quantifying Greenhouse Gas Fluxes in Agriculture and Forestry: Methods for Entity-Scale Inventory* (eds Eve M, Pape D, Flugge M, Steel R, Man D, Riley-Gilbert M, and Biggar S), Technical Bulletin Number 1939. Office of the Chief Economist, U.S. Department of Agriculture, Washing DC. 606 p.

Houghton RA, Hackler JL, Lawrence KT (1999) The U.S. carbon budget: Contributions from land-use change. Science, **285**, 574-578.

Houghton RA, Hackler JL (2001) Carbon flux to the atmosphere from land-use changes: 1850 to 1990. Oak Ridge National Laboratory/Carbon Dioxide Information Analysis Center. U.S. Department of Energy, Oak Ridge National Laboratory, Oak Ridge, Tennessee, U.S.A. 86 p.

Houghton RA, House JI, Pongratz J, van der Werf GR, DeFries RS, Hansen MC, Le Quéré. C. Ramankutty N (2012) Carbon emissions from land use and land-cover change. Biogeosciences, **9**, 5125-5142.

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, **103**, 83-93.

Idso CD, Idso SB, Bailing RC Jr (1998) The urban CO_2 dome of Phoenix, Arizona. Physical Geography, **19**, 95-108.

Imhoff ML, Bounoua L, DeFries R, Lawrence WT, Stutzer D, Tucker CJ, Ricketts T (2004) The consequences of urban land transformations on net primary productivity in the United States. Remote Sensing of the Envrionment, **89**, 434-443.

Jeon SB, Olofsson P and Woodcock CE (2014) Land use change in New England: a reversal of the forest transition. Journal of Land Use Science, **9**, 105–30.

Jin S, Yang L, Danielson P, Homer CG, Fry J and Xian, G (2013) A comprehensive change detection method for updating the National Land Cover Database to circa 2011. Remote Sensing of Environment, **132**, 159-75.

Jones AD, Collins WD, Torn MS (2013) On the additivity of radiative forcing between land use change and greenhouse gases. Geophysical Research Letters, **40**, 4036–4041

Jones AD, Calvin KC, Collins WD, Edmonds J (2015) Accounting for radiative forcing from albedo change in future global land-use scenarios. Climatic Change, **131**, 691-703.

Kong FH, Yin HW, James P, Hutyra LR, He HS (2014) Effects of spatial pattern of greenspace on urban cooling in a large metropolitan area of eastern China. Landscape and Urban Planning, **128**, 35-47.

Lawler JJ, Lewis DJ, Nelson E, et al. (2014) Projected land-use change impacts on ecosystem services in the United States. Proceedings of the National Academy of Sciences, **111**, 7492-7497.

Liu Z, He C, Zhou Y, Wu J (2014) How much of the world's land has been urbanized, really? A hierarchical framework for avoiding confusion. Landscape Ecology, **29**, 763-771.

McDonald RI, Motzkin G, Bank MS, Kittredge DB, Burk J, Foster DR (2006) Forest harvesting and land-use conversion over two decades in Massachusetts. Forest Ecology and Management, **227**, 31-41.

Menon S, Akbari H, Mahanama S, Sednev I, Levinson R (2010) Radiative forcing and temperature response to changes in urban albedos and associated CO2 offsets. Environmental Research Letters **5**, 014005.

Mini C. Hogue TS, Pincetl S (2014) Estimation of residential outdoor water use in Los Angeles, California. Landscape and Urban Planning, **127**, 124-135.

Muñoz I, Campra P, Fernández-Alba AR (2010) Including CO2-emission equivalence of changes in land surface albedo in life cycle assessment. Methodology and case study on greenhouse agriculture. The International Journal of Life Cycle Assessment, **15**, 672–681.

Nakane K, Kohno T, Horikoshi T (1996) Root respiration rate before and just after clear-felling in a mature, deciduous, broadleaved forest. Ecological Research, **11**, 111-19.

Nakicenovic N, Swart R (2000) Special Report on Emissions Scenarios: Summary for Policymakers. Cambridge University Press, Cambridge.

Nowack DJ, Walton JT, Dwyer JF, Kaya LG, Myeong S (2005) The increasing influence of urban environments on US forest management. Journal of Forestry, **103**, 377-82.

Nowack DJ, Walton JT (2005) Projected urban growth (2000-2050) and its estimated impact on the US forest resource. Journal of Forestry, **103**, 383-389.

USDA (U.S. Department of Agriculture) (2010) Summary Report: 2010 National Resources Inventory. Natural Resources Conservation Service, Washington, DC, and Center for Survey Statistics and Methodology, Iowa State University, Ames, Iowa. 166p.

Oke TR (1973) City size and the urban heat island. Atmospheric Environment Pergamon Press, **7**, 769-779.

Oliver JGJ, Janssens-Maenhout G, Muntean M, Peters JAHW (2013) Trends in global CO2 emissions; 2013 Report, The Hague: PBL Netherlands Environmental Assessment Agency, Ispra: Joint Research Centre.

Olofsson, P, Foody, GM, Stehman, SV, Woodcock, CE (2013) Making better use of accuracy data in land change studies: Estimating accuracy and area and quantifying uncertainty using stratified estimation. Remote Sensing of Environment, **129**, 122–131.

Olofsson P, Foody GM, Herold M, Stehman SV, Woodcock CE, Wulder MA (2014) Good practices for estimating area and assessing accuracy of land change. Remote Sensing of Environment, **148**, 42–57.

Olofsson P, Holden CE, Bullock EL, Ma Y, Woodcock CE. Time series analysis of satellite data reveals continuous deforestation of New England since the 1980s. Environmental Research Letters, (submitted for review).

Pan Y, Birdsey RA, Fang J, et al. (2011) A large and persistent carbon sink in the world's forests. Science, **333**, 988-993.

Proctor P, Heath, LS, Van Deusen PC, Grove JH, Smith JE (2005) COLE: A web-based tool for interfacing with forest inventory data. In: *Proceedings of the fourth annual forest inventory and analysis symposium* (eds McRoberts RE et al.). General Technical Report NC-252. USDA, Forest Service, North Central Research Station, St. Paul, MN. p. 167-172.

Raciti SM, Hutyra LR, Newell JD (2014) Mapping carbon storage in urban trees with multi-source remote sensing data: Relationships between biomass, land use, and demographics in Boston neighborhoods. Science of the Total Environment, **500-01**, 72-83.

Russell MB, Woodall CW, Fraver S, D'Amato AW, Domke GM, Skog KE (2014) Residence times and decay rates of downed woody debris biomass/carbon in eastern US forests. Ecosystems, **17**, 765-777.

Sailor D (1996) Simulated urban climate response to modifications in surface albedo and vegetative cover. Journal of applied meteorology, **34**, 1694-1704.

Schwaab J, Bavay M, Davin E, Hagedorn F, Hüsler F, Lehning M, Schneebeli M, Thürig E, Bebi P (2015) Carbon storage versus albedo change: radiative forcing of forest expansion in temperate mountainous regions of Switzerland. Biogeosciences, **12,** 467–487.

Seto KC, Güneralp B, Hutyra LR (2012) Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. Proceedings of the National Academy of Sciences of the United States of America, **109**, 16083-16088.

Sleeter BM, Sohl TL, Loveland TR, Auch RF, Acevedo W, Drummond MA, Sayler KL, Stehman SV (2013) Land-cover change in the conterminous United States from 1973-2000. Global Environmental Change, **23**, 733-48.

Sohl TL, Sleeter BM, Zhu Z et al. (2012) land-use and land-cover modeling strategy to support a national assessment of carbon stocks and fluxes. Applied Geography, **34**, 111-124.

Sohl TL, Sayler KL, Bouchard MA et al. (2014) Spatially explicit modeling of 1992 to 2100 land cover and forest stand age for the conterminous United States. Ecological Applications, **24**, 1015-1036.

Thompson JR, Foster DR, Scheller R, Kittredge D (2011) The influence of land use and climate change on forest biomass and composition in Massachusetts, USA. Ecological Applications, **21**, 2425-2444.

Thompson JR, Fallon-Lambert K, Foster DR, Blumstein M, Broadbent EN, Almeyada Zambrano AM (2014) Changes to the Land: Four Scenarios for the Future of the Massachusetts Landscape. Harvard Forest, Harvard University.

U.S. Census Bureau. Population Division of the Population Estimates Program, Washington, D.C., U.S.A.

U.S. EPA (Environmental Protection Agency) (2010) ICLUS v1.3 User's Manual: ArcGIS Tools and Datasets for Modeling US Housing Density Growth. Global Change Research Program, National Center for Environmental Assessment, Washington, DC; EPA/600/R- 09/143F. Available from the National Technical Information Service, Springfield, VA, and online at http://www.epa.gov/ncea/global.

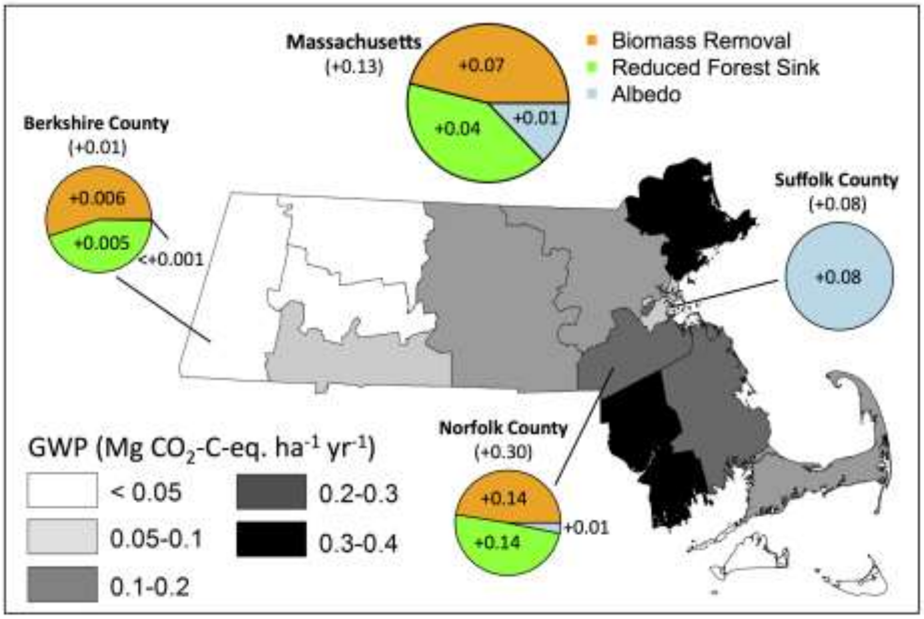
U.S. EPA (Environmental Protection Agency) (2013) The 2011 National Emissions Inventory version 1. Available online at http://www.epa.gov/ttnchie1/net/2011inventory.html

Van Deusen P, Heath LS (2015) COLE web application suite. NCASI and USDA Forest Service, Northern Research Station. Available only on internet: http://www.ncasi2.org/COLE/ (Accessed June 2015 and December 2015).

Yang X, Tian Z, Chen B (2013) Thermal growing season trends in each China, with emphasis on urbanization effects. International Journal of Climatology, **33**, 2402-2412.

Zhao T, Brown DG, Fang H, Theobald DM, Liu T, Zhang T (2012) Vegetation productivity consequences of human settlement growth in the eastern United States. Landscape Ecology, **27**, 1149-1165.

Zhu Z, Woodcock CE (2014) Continuous change detection and classification of land cover using all available Landsat data. Remote Sensing of Environment, **144**, 152–171.


Massachusetts
(+0.13)
+0.07
+0.04
+0.01
Biomass Removal
Reduced Forest Sink
Albedo
Berkshire County
(+0.01)
+0.006
+0.005
<+0.001
Suffolk County
(+0.08)
+0.08
Norfolk County
(+0.30)
+0.14
+0.14
+0.01
GWP (Mg CO_2-C-eq. ha^{-1} yr^{-1})
< 0.05
0.05-0.1
0.1-0.2
0.2-0.3
0.3-0.4