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# Terrestrial carbon stocks across a gradient of urbanization: a study of the Seattle, WA region

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## Abstract

Most of our global population and its $CO_2$ emissions can be attributed to urban areas. The process of urbanization changes terrestrial carbon stocks and fluxes, which, in turn, impact ecosystem functions and atmospheric $CO_2$ concentrations. Using the Seattle, WA, region as a case study, this paper explores the relationships between aboveground carbon stocks and land cover within an urbanizing area. The major objectives were to estimate aboveground live and dead terrestrial carbon stocks across multiple land cover classes and quantify the relationships between urban cover and vegetation across a gradient of urbanization. We established 154 sample plots in the Seattle region to assess carbon stocks as a function of distance from the urban core and land cover [urban (heavy, medium, and low), mixed forest, and conifer forest land covers]. The mean (and 95% CI) aboveground live biomass for the region was $89 \pm 22\,\mathrm{Mg\,C\,ha^{-1}}$ with an additional $11.8 \pm 4\,\mathrm{Mg\,C\,ha^{-1}}$ of coarse woody debris biomass. The average live biomass stored within forested and urban land covers was $140 \pm 40$ and $18 \pm 14\,\mathrm{Mg\,C\,ha^{-1}}$, respectively, with a 57% mean vegetated canopy cover regionally. Both the total carbon stocks and mean vegetated canopy cover were surprisingly high, even within the heavily urbanized areas, well exceeding observations within other urbanizing areas and the average US forested carbon stocks. As urban land covers and populations continue to rapidly increase across the globe, these results highlight the importance of considering vegetation in urbanizing areas within the terrestrial carbon cycle.

*Keywords:* carbon cycle, climate change, development, mitigation, Pacific Northwest, urban

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## Introduction

Urbanization and growth of the human population has yielded cities of unprecedented size, extent, and form (Decker *et al*., 2000; McDonald, 2008; Schneider & Woodcock, 2008), that emit significant quantities of waste (e.g. $CO_2$, Folke *et al*., 1997), transform habitat (Vitousek *et al*., 1997), modify major biogeochemical cycles (Kaye *et al*., 2006), alter local climate (Oke, 1982), and impact human health (Patz *et al*., 2005). Around the globe, urbanization is expected to increase significantly in the coming decades as populations and economic activity continue to grow (Foley *et al*., 2005; Theobald, 2005). In 2008, half the world's population lived in urban areas and by 2050, 70% of the population is projected to become urbanites (UNFPA, 2007). Although humans have interacted with their biophysical environment since the beginning of human history, the magnitude, complexity, and implications of these interactions have increased dramatically in recent decades (Liu *et al*., 2007). If recent trends continue, the expansion of urban areas will markedly outpace the growth in urban populations, making urban land use change and carbon dynamics therein ever more important for the global carbon cycle (Brown *et al*., 2005; Churkina *et al*., 2010). Urban development choices play a central role in determining local, regional, and global carbon emissions (via factors such as land clearing, energy consumption, and transportation) and terrestrial carbon sinks (via vegetation carbon storage and uptake) (Alberti, 2008).

In urbanizing regions, organic carbon is stored within and cycled through the air, soils, waters, plants, and the built environment itself. Although it is abundantly clear that cities and urbanizing areas affect local and global sinks and sources of $CO_2$, the exact magnitude of and mechanisms for carbon exchange remain highly uncertain for urbanizing regions (Pataki *et al*., 2006). Some estimates suggest that in excess of 90% of anthropogenic carbon emissions are attributable (directly or indirectly) to cities (Grübler, 1994; Svirejeva-Hopkins *et al*., 2004; Churkina, 2008; IEA, 2009), but attribution of emissions to cities as a whole is very challenging (Kennedy *et al*., 2007) and prone to double counting and leakage given that most of the energy consumed in urban area is generated elsewhere (Cannell *et al*., 1999).

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The process of urbanization itself typically results in substantial emissions due to the land clearing and construction activities. Most developed lands typically have some level of vegetation returned after construction (from road median planting strips and greenbelts to residential landscaping and gardens), but urban expansion results in complex patterns of intermixed high- and low-density built-up areas and a fragmentation of the natural landscape. The complex interactions between urbanization and vegetation functions are influenced by both human and biophysical factors and competing positive and negative feedbacks among them (Gregg *et al*., 2003; Grimm *et al*., 2008). The aggregated effects of urbanization (including changing land cover characteristics, land use patterns, pervious surface fractions, urban heat islands, extended growing seasons, atmospheric pollution, management activities, etc.) on land–atmosphere exchange processes remains highly uncertain despite decades of study on components of the problem (Pouyat *et al*., 2006; Canadell *et al*., 2007; Trusilova & Churkina, 2008).

Simple characterizations of urbanization affects on vegetation are complicated by varying definitions of what is 'urban,' different regional patterns of urban development, and varying interactions with climate. We use the term 'urbanizing regions' to emphasize two aspects of 'urban' interactions within the carbon cycle: (1) interactions are not limited to the city boundaries and (2) urbanization is a process that involves several stages potentially including development, redevelopment, and renewal. Humans directly influence plant growing conditions by behaviors such as watering, fertilizing, pruning, removal of organic material (leaf litter and limbs), and the planting of exotic species. Vegetation change also affects local climate through changes in the urban energy balance (Oke, 1982) and can alter local heating and cooling requirements (and by extension carbon emissions). Significant regional differences can be expected in urban emissions and sinks due to local land use histories, plant species assemblages, human behaviors, and climatic differences.

An accurate characterization of urbanizing regions is critical to understand the mechanisms linking urban development to carbon stocks and fluxes (Alberti & Hutyra, 2009). The use of baselines in terrestrial carbon studies and climate negotiations is very important, but can be challenging in terrestrial ecosystems where land cover and vegetation are constantly changing with plant successional dynamics and disturbance (both natural and anthropogenic). Recovering and/or younger forests can often uptake more carbon due to more optimal growing conditions, but their carbon stocks are inherently not near their upper limits. Larger stature, less recently or severely disturbed forests, have been the focus of detailed and more integrated carbon assessments but these ecosystems are becoming increasingly rarer as urban and human influences increase in their spatial extent and intensity.

Forested ecosystems are capable of storing large quantities of carbon within their live and dead organic material. Smithwick *et al*. (2002) found that old forests in the Western Cascades of Washington could store near $450\,\mathrm{Mg\,C\,ha^{-1}}$ (only aboveground component reported here). Forest disturbances, natural or anthropogenic, have the potential to shift these carbon stocks quickly from the terrestrial biosphere into the atmosphere, increasing atmospheric $CO_2$ concentrations. However, growing and regrowing forests, where the photosynthetic uptake exceeds respiratory losses, can remove carbon from the atmosphere at rates of up to several $\mathrm{Mg\,C\,ha^{-1}\,yr^{-1}}$. Wood products, such as buildings or fences, can store (sequester) carbon for long periods of time, while forest management activities can either create additional plant growing space or create space for additional development. By accounting for changes in forested areas and characteristics (both forest carbon stocks and fluxes), it is possible to determine if an area is a net carbon source or sink. Biometric carbon accounting methods have been widely used within forested areas (e.g. Curtis *et al*., 2002), but comparatively few studies have attempted such efforts for urbanizing regions (with a few notable exceptions; e.g. Nowak & Crane, 2002; Churkina *et al*., 2010).

The potential of urban vegetation to provide a sink for $CO_2$ requires a full characterization of the urban carbon budget. Eddy covariance studies in suburban areas near Melbourne, Australia (Coutts *et al*., 2007), Chicago, USA (Grimmond *et al*., 2002), and Vancouver, Canada (Walsh *et al*., 2004) all found that carbon uptake by local suburban vegetation significantly reduced the local net emissions and reduced local background atmospheric $CO_2$ concentrations. Urban carbon emissions estimates (e.g. VULCAN or Hestia, Gurney *et al*., 2009) are a critical component of the urban carbon cycle, but they are not the entire budget and vegetation can hold a key role for both reducing local net emissions. These new emissions products represent a major advancement in our consideration of urban carbon budgets, but they remain observationally unvalidated for vast areas. Currently, global climate models do not take urban areas or their carbon dynamics into account, and thus urban emissions have been largely prescribed (Svirejeva-Hopkins *et al*., 2004; Churkina, 2008).

Our objectives in this paper are to (1) use direct field observations of aboveground live and dead biomass to assess terrestrial carbon stocks within the Seattle, WA urbanizing region; (2) assess how terrestrial carbon stocks and forest structure vary as a function of urban land covers across a gradient of urbanization; and (3) quantify baseline carbon stocks for future assessments of regional urban sequestration potentials. Although several studies have started to characterized the urban carbon cycle (Pataki *et al*., 2006; Churkina *et al*., 2010) and quantify the carbon stocks in urban land uses (Nowak, 1994; Nowak & Crane, 2002; Pouyat *et al*., 2006; Churkina *et al*., 2010), these studies have rarely taken into account the heterogeneity of urban land cover across a gradient of urbanization. In this study, we differentiate our observation of carbon stocks to capture land cover heterogeneity that typifies an urbanizing landscape. Direct field observations across an urban gradient are a critical step in characterizing the biological components of the urban carbon cycle across a range of spatial scales to understand the ecosystem responses to urbanization and land cover change.

## Methods

### *Site description*

The Seattle, WA area is bounded by the Puget Sound to the west and the Cascade Mountains to the east. The soils are dominated by loam and sandy loam, with over 30% of the area classified as 'Alderwood gravelly sandy loam' (NRCS, 2009). The vegetation is largely temperate, moist forest. The west slope of the Cascade Mountains (~50 km from Seattle) is dominated by coniferous trees: Douglas-fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), and western red cedar (*Thuja plicata*). Puget Sound lowland forests include a similar coniferous species composition and deciduous species such as bigleaf maple (*Acer macrophyllum*), black cottonwood (*Populus trichocarpa*), and red alder (*Alnus rubra*). The annual mean temperature in Seattle is 11.3 °C (mean monthly range from 4.8 to 18.7 °C, 1971–2000 mean climatology; NOAA, 2004) and the mean annual precipitation is 942 mm (increasing with elevation; 1971–2000 mean climatology; NOAA, 2004).

For the last several decades, the Seattle region has experienced sustained population growth in excess of 1% $yr^{-1}$, with most of the development occurring outside the Seattle urban core. The Seattle region is projected to grow another 32.4% between 2005 and 2030 (from 3.2 to 4.3 million people; WOFM, 2007). The Puget Sound region was once very heavily forested, but forest cover has been reduced by 40% in just over a century and is expected to be reduced by another 20% in the coming decades (Hepinstall *et al*., 2008). The patterns of urbanization and sprawl in the Seattle region are not atypical for Western US cities (Robinson *et al*., 2005).

### *Sampling design*

Sampling was conducted at 154 sites across the Central Puget Sound, WA, including both publicly and privately held lands. Sample locations were between 0 and 58 km from the Seattle central business district. Our sampling strategy was designed to (1) characterize aboveground terrestrial carbon stocks as a function of different land cover types; (2) assess how carbon stocks varied across a gradient of urbanization; and (3) generate the data to conduct a preliminary change analysis based on a 20-year time series of Landsat land cover data (L. R. Hutyra, B. Yoon & M. Alberti, unpublished results). We used a 2002, 30 m Landsat TM land cover classification to stratify our field samples (Alberti *et al*., 2004a, b). Parcel-level GIS data and road/trails network data were obtained from the local counties to assess land ownership and accessibility.

We established three sample transects radiating from the Seattle central urban core (UTM zone 10°N: 549518°E, 5273765°N) to cover a range of land development types and land use intensities (Fig. 1). The transects followed bearings of 43°, 110°, and 300° and extended for 50, 50, and 58 km, respectively. The three transects included patches of varying development types, site histories, and local income levels. Transect 1 (43° bearing) extended to Monroe, WA passing through Lake Washington, Kirkland, and Woodinville, WA. Transect two (110° bearing) roughly followed the Interstate-90 corridor extending from Seattle to North Bend, WA passing through Lake Washington, Mercer Island, Bellevue, and Issaquah, WA. The third transect (300° bearing) passed through Renton, Covington, and Enumclaw, WA.

The transects were divided into three sections (sections 1–3: 0–7.5, 7.5–30, and >30 km from the Seattle core, respectively). We determined the section breakpoints based on observed discontinuities in the fraction of urban impervious cover (data not shown). We delineated grid boxes of 450 × 450 m centered on the transect lines as the study sample areas (Fig. 1, inset). Only areas below 500 m in elevation were included in the potential pool of sample sites to remove some of the variance in forest structure associated with elevation changes (3.1% of the total transect area was excluded due to elevation).

We assessed aboveground carbon stocks and site characteristics within five different land cover classes: heavy urban, medium urban, low urban, mixed forest, and coniferous forest (Table 1). The urban classes were defined based on percentage impervious surface: heavy (>80% impervious surface), medium (50–80% impervious surface), and low (20–50% impervious surface) urban (Alberti *et al*., 2004a, b). Approximately, 30 samples were obtained for each of the five land cover classes of interest; 10 samples per class within each section (0–7.5, 7.5–30, and >30 km from the Seattle). Potential samples were randomly drawn across the three transects within each section, with a minimum requirement of one sample from each transect for each cover class and section. For each of the five land cover classes, 25 initial, random samples were drawn from the grid boxes within each section. The 25 initial random plots were assigned a random rank order of 1–25, such that we moved through the list in numeric order until we were able to gain access to 10 plots per land cover type within each section with each section. Ownership and addresses were determined from the parcel-GIS for all 25 potential sites and letters requesting access (including prepaid return postcards) were sent to all private landowners. As the potential sample plots often fell within multiple ownerships, letters were sent to all landowners within the plot area or private land holdings

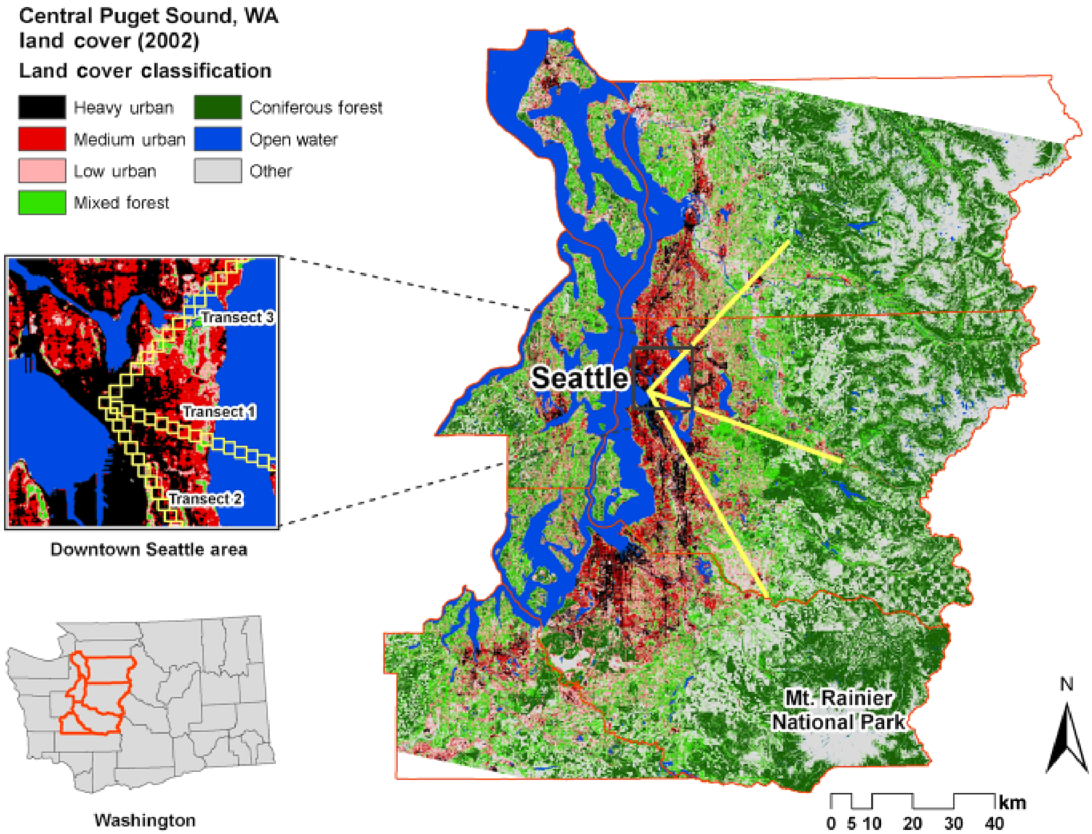


**Fig. 1** Sample transects overlaid on 2002 land cover map for the Central Puget Sound, WA region (including Snohomish, Island, King, Kitsap, Pierce, and Thurston county areas).

**Table 1** Distribution of land cover classes (2002) for the sample transects and the Central Puget Sound region (only including land areas below 500 m elevation)

| | Sample transects | | | | |
|---|---|---|---|---|---|
| | Section 1 | Section 2 | Section 3 | Overall transects | Central Puget Sound |
| Heavy urban | 335 ha (49%) | 425 ha (18%) | 86 ha (4%) | 846 ha (16%) | 63 753 ha (6%) |
| Medium urban | 217 ha (32%) | 836 ha (36%) | 241 ha (10%) | 1293 ha (24%) | 120 936 ha (11%) |
| Low urban | 80 ha (12%) | 425 ha (18%) | 432 ha (18%) | 936 ha (17%) | 179 483 ha (17%) |
| Mixed forest | 33 ha (5%) | 292 ha (13%) | 698 ha (29.%) | 1023 ha (19%) | 246 735 ha (23%) |
| Conifer forest | 0.5 ha (0.1%) | 194 ha (8%) | 433 ha (18%) | 628 ha (12%) | 230 966 ha (21%) |
| Other land covers* | 20 ha (3%) | 167 ha (7%) | 502 ha (21%) | 689 ha (13%) | 241 352 ha (22%) |

The spatial extent of the Central Puget Sound area is shown in Fig. 1.
*Other land cover classes include regenerating forest, clearcut, cleared for development, wetland, shoreline, agriculture, grassland, and snow/bare rock.

through which we had to pass. Table 2 summarizes the total mailings, response rates, and degree of parcelization of the transects. In section 3, we required 175 potential sites in order to obtain access to 10 sites per cover class due to a lower response rate and higher fraction of private land ownership. Each of the sample sites was surveyed, unless it was deemed too dangerous for student access due to a combination of factors such as dogs, areas being used for various illicit activities, homeless encampments, or extremely steep slopes ($>\sim45°$) covered in Himalayan blackberries (*Rubus armeniacus* or *Rubus discolor*). All of the final surveyed plot locations are shown in Fig. 2.

In the most heavily urbanized sample segment (section 1), we were unable to sample 10 coniferous sites within the transect area because there were only four pixels classified as coniferous forest within the over 9000 pixels covered by the transect grid boxes in that section. The initial four sample points were classified as isolated, single conifer pixels; upon

**Table 2** Field access permission information for private property

| | Sample transects | | | |
|---|---|---|---|---|
| | Section 1 | Section 2 | Section 3 | Overall transects |
| Plots requiring private land owner permission | 10 of 54 plots | 11 of 50 plots | 27 of 50 plots | 48 of 154 |
| Total access requests sent | 114 | 191 | 204 | 509 |
| Permission granted | 33 | 37 | 27 | 97 |
| Permission not granted | 7 | 11 | 10 | 28 |
| No response | 74 | 143 | 167 | 384 |
| Mean number of parcels per plot | 2.3 parcels overall (1.8 parcels for the 54 surveyed plots) | 2.2 parcels overall (1.6 parcels for the 50 surveyed plots) | 1.7 parcels overall (1.2 parcels for the 50 plots surveyed) | 2.0 parcels overall (1.6 parcels for the 154 surveyed plots) |

For each mailing sent, a short letter describing the project and prepaid response card was included.

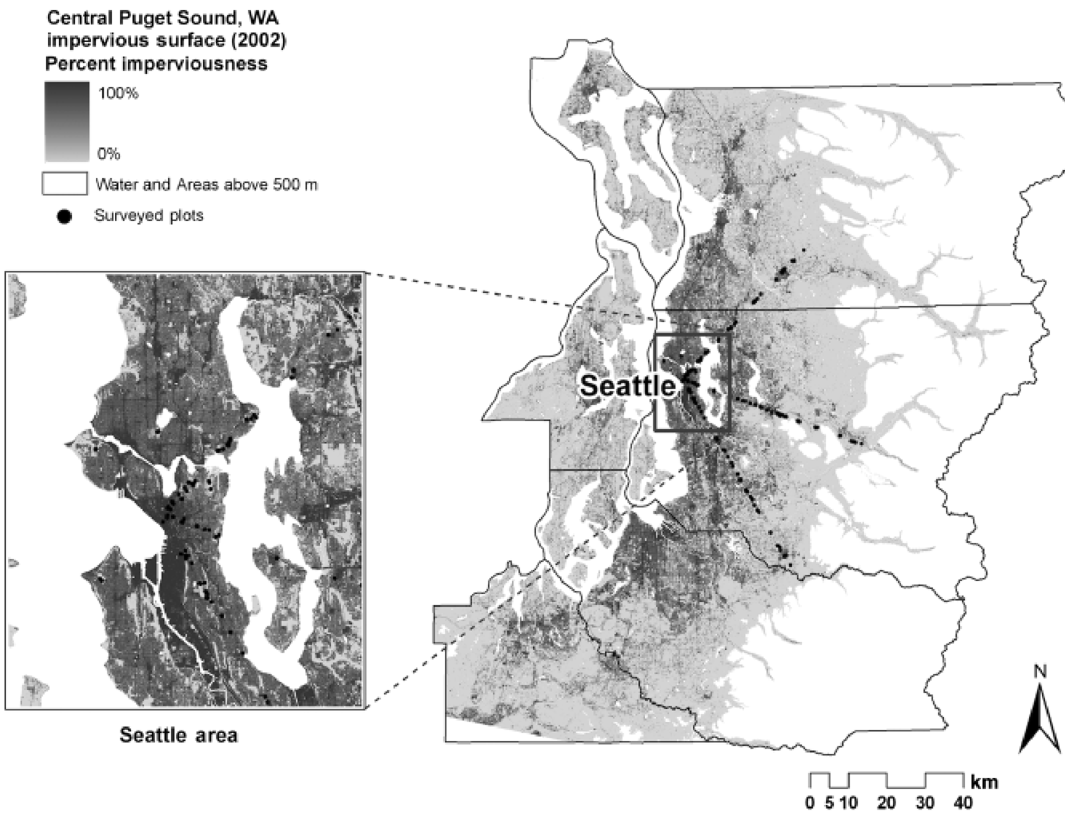


**Fig. 2** The 154 sample plot locations overlaid on 2002 percent impervious surface cover (Alberti *et al.*, 2004a) for the Central Puget Sound, WA region. Only areas below 500 m elevation are shown here.

visiting the sites, none of these four pixels were actually conifer forests. The overall accuracy of the 2002 Puget Sound land cover classification was 83% and the conifer forest user's accuracy was 75% (Alberti *et al.*, 2004b). To address this section 1 conifer sampling problem, we randomly drew additional plots from the full area (within a radius of 7.5 km from the urban core) in order to obtain an additional 10 sample plots. We required that all of the supplementary conifer plots not be single conifer pixels to minimize misclassification errors and obtain a robust conifer forest estimate. All of the 14 conifer forest plots from section 1 are included in the analysis.

All field sampling was conducted between April and October 2009. Sample plots were circular with a 15 m radius (706 $m^2$). We used the Garmin 60CXs GPS (Olathe, KS, USA) to locate plot centers. Typical GPS errors for plot center locations were 5–10 m, depending on canopy cover thickness. We attempted to match the center of the land cover pixels (30 m resolution) with the on-the-ground plot centers (30 m plot diameter), but errors in the GPS locations meant that exact colocation was not possible. Depending on the local topographic and vegetation circumstances, we used a combination of a Nikon Forestry 550 laser range finder (Tokyo, Japan), meter tapes, and a Brunton CM360LA clinometer (Riverton, WY, USA) to determine exact plot boundaries. All slope distances were corrected to horizontal distance. Canopy cover area was manually delineated for each sample plot using the Washington State, National Agriculture Imagery Program digital orthophotos from 2006 (0.46 m resolution).

### *Aboveground live biomass*

All live trees larger than 5 cm in diameter at breast height (DBH) were surveyed. DBH was measured at 1.37 m unless slope or tree form abnormalities (particularly buttresses) required adjustments; measurements followed the protocols outlined in Fahey & Knapp (2007). Tree diameters were measured with DBH tapes to the nearest 0.1 cm. Where possible, trees were identified to species or genus (if species could not be determined), but due to the large number of exotic species present within urban area, 2.7% of stems were identified as miscellaneous hardwood species.

Biomass of live trees was estimated using published allometric equations relating plant diameter to dry mass. Species-specific equations were used where possible, including bigleaf maple (Gholz *et al*., 1979), black cottonwood (Singh, 1984), Douglas-fir (Gholz *et al*., 1979), red alder (Binkley, 1983), western hemlock (Gholz *et al*., 1979), and western red cedar (Gholz *et al*., 1979). Genus-level or more general equations were applied in other cases: *Acer/Betula*, *Cedrus/Larix*, mixed hardwood, *Pinus*, *Picea*, and *Populus/Alnus/Salix* (Jenkins *et al*., 2003). Biomass for *Camellia* spp. and *Rhododendron* spp. was estimated using a *Rhododendron macrophyllum* biomass equation (Gholz *et al*., 1979). In the case of bigleaf maple, if the diameter of a sample tree exceeded the 25.3 cm (the maximum tree size sampled to create the allometry), the Jenkins *et al*. (2003) miscellaneous hardwood equation was applied to avoid over estimation from the exponential form of the equation. There were seven individual trees (of the total 3261 surveyed) which exceed the diameter used to define the miscellaneous hardwood equation (max. diameter = 56 cm), the estimated biomass for those trees was reduced by 15% to partially correct for the over-estimation error. The most specific equation possible was applied in all cases; where species or genus level equations were unavailable, we applied the Jenkins *et al*., (2003) miscellaneous hardwood equation. One half of live plant biomass was assumed to be carbon. All of the plant biomass is reported in units of dry weight carbon, $\mathrm{Mg\,C\,ha^{-1}}$.

Nowak (1994) reported that open-grown, maintained, urban trees tended to have a lower aboveground biomass than their forest counterparts from which allometries are almost exclusive derived. For field plots with fewer than seven stems per plot (each with a minimum diameter of 5 cm, $\sim$70 trees $\mathrm{ha^{-1}}$), the biomass estimate was multiplied by 0.8 to correct for urban allometric overestimation ($n = 26$ plots of $N = 154$, median reduction in biomass was $1.3\,\mathrm{Mg\,C\,ha^{-1}}$). Note that recent results from McHale *et al*. (2009a) suggest that broader problems with the application of forest-derived allometries to urban trees likely exist. A more complete allometric analysis was not possible within the scope of this study due to the lack of availability of urban tree allometries for the regional species assemblage. The use of forest-derived allometric equations also afforded the analysis a methodological consistency across the urban-to-rural gradient which included a range from heavily urbanized street trees to rural forest trees. It is unclear if the use of urban-specific allometric equations would have increased or decreased the vegetation carbon stock estimates in the most urban plots, this is an important area for additional research.

Within the more urbanized areas, some trees had the top portion of their crowns removed to improve residential views. In such cases, additional height and/or diameter measurements were made to improve biomass estimates. For example, if a tree consisted of only a central stem with a few leaves remaining (most branches removed), the biomass was estimated by multiplying the calculated volume of the remaining stem portion as a tapered cylinder by the mean hardwood or softwood density to estimate biomass. For all plots, understory vegetation conditions and actual land cover were recorded. To explore possible methods for spatial extrapolation of aboveground biomass, we carefully digitized the canopy cover area for all the surveyed plots using 0.46 m resolution digital orthophotos from 2006 (the most recently available photos).

### *Coarse woody debris (CWD)*

Downed and standing CWD with a diameter $>10$ cm and a minimum length of 1 m were surveyed within the full sample plot areas. Logs were identified as hardwood or softwood (where possible) and assigned decay class values. Following the conventions defined by Harmon & Sexton (1996) and Barford *et al*. (2001), decay classes used for both standing and fallen CWD were: (1) decay class 1 – solid wood, recently fallen, bark and twigs present; (2) decay class 2 – solid wood, significant weathering, branches present; (3) decay class 3 – wood not solid, may be sloughing but nail still must be pounded into the log; (4) decay class 4 – wood sloughing and/or friable, nails may be forcibly pushed into log; and (5) decay class 5 – wood friable, barely holding shape, nails may be easily pushed into log. Dimensional measurements were converted to volumes, using Newton's formula for a cylinder (Harmon and Sexton, 1996). CWD wood density values were applied to calculated volumes to estimate biomass; CWD hardwood density values from Gough *et al*. (2007) and softwood density values from Harmon & Sexton (1996) were utilized. Where log wood type could not be confidently identified, each stand type was classified as hardwood, softwood, or mixed and density values were thus applied. For mixed stands, a mean of the hard- and softwood densities was used to estimate biomass. One half of the CWD biomass was assumed to be carbon. All of the CWD biomass is reported here is in units of dry weight carbon, $\mathrm{Mg\,C\,ha^{-1}}$.

For recently dead and standing trees, where branches and twigs were still present, we estimated biomass by reducing the allometrically estimated biomass for a live tree by 1/3 to account biomass losses associated with mortality (Liu *et al*., 2006). For more decomposed standing dead trees (no branches), volume was estimated by measuring tree height, base diameter, and decay class. Top diameter was measured (where possible) or visually estimated. Biomass for standing logs was estimated using the same 5-class decay method.

Large CWD piles were found in many of the managed and/or restored urban forests making direct measurement not possible without significantly disrupting forest conditions. In such cases, the volume of the pile and percent solid volume (meeting requisite size requirements) was estimated and a mean decay class was used to estimate biomass. The volume of some highly decomposed and/or noncylindrical CWD pieces was estimated as blocks or pyramids based on height, length, and width measurements.

Although not typical for biometric studies, the human built woody biomass (BWB), such as utility poles, pilings, and fences, were also surveyed as part of this study because within the most intensely urban areas such carbon pools can be significant. Utility poles biomass volume was estimated based on the measured length, base diameter, and a visual estimate of the top diameter. Fences and other substantial wooden structures (excluding building) within sample plots were also included in the BWB estimates if they met minimum size criteria (1 m in length and a minimum area of 75 $cm^2$). The BWB is not included in the CWD estimates.

### *Error estimates and spatial extrapolation*

Sample plot results were scaled to the full segment, transect, or Central Puget Sound area using the distribution of observed land cover within the given area (Table 1). When scaling to the full Central Puget Sound region, we applied the proportion of the land cover distribution within the full section radius (e.g. 7.5, 7.5–30, or $>30$ km radius from Seattle urban core), rather than limiting the distribution weighting to the sample transect or using the overall transect mean values. The mean biomass by land cover class and sample segment was used to estimate the biomass for the corresponding radial segment.

Unless noted otherwise, all parenthetically reported errors are 95% confidence intervals. Sampling uncertainties around these biomass quantities were calculated by bootstrap analyses (Efron & Tibshirani, 1993) due to heteroscedasticity within the data distributions. Bootstrap samples were drawn 1000 times with replacement to estimate 95% confidence intervals around live and dead carbon stocks. Error estimates include field sampling, but do not include allometric or spatial scaling errors. Satellite land cover classification errors were not directly included, but field sampling and subsequent scaling did include misclassified pixels within the mean and confidence interval calculations such that it was partially accounted for within the overall error estimates.

## Results

### *Forest structure and land characteristics*

We surveyed a total of 3261 individual live tree stems and 1480 pieces of CWD across 154 sample plots between April and October 2009; 28 sample plots had no live trees present and 55 sample plots had no CWD present. The overall average stem density was $297 \pm 51.3$ stems $ha^{-1}$. The mean stems density did not directly correspond to tree density because many of the flowering trees (particularly *Acer* spp., *Rhododendron* spp., and *Camellia* spp.) had multiple stems associated with the same individual plant. Larger trees ($\geq 20$ cm DBH) accounted for the vast majority of the total biomass (95%), though smaller trees ($\geq 5$ cm and $<20$ cm DBH) were much more common (2011 trees, 62% of stems). The distribution of stem density vs. size was log linear with slightly steeper slopes for more urban land covers (Fig. 3). The total stem density per area increased with decreased urbanization. The mean observed DBH was 21.5 cm (median = 13.5 cm), with a maximum observed tree diameter of 125.4 cm found for a Douglas-fir tree (section 3, conifer forest land cover).

The average canopy coverage by cover class is reported in Table 3. As expected, the fraction canopy cover was negatively correlated with urban intensity and positively correlated with the distance from the urban core. The fraction canopy cover explained approximately 61% of the observed variance in live aboveground biomass (Fig. 4). The pixels classified as heavy urban showed the strongest correlation between canopy cover and biomass ($R^2 = 0.87$), likely due to the narrow range in canopy cover (0–48%) and the influence of plots with no canopy or biomass present ($n = 15$ of $N = 30$ heavy urban plots). Across the full sample, plots with $\geq$95% canopy cover showed a range of over 350 Mg C $ha^{-1}$ in their aboveground live biomass. Modifying the plot sample area to exclude the paved portions further weakened the correlation between canopy cover and biomass per unit area (data not shown). The poor overall relationship between canopy cover and biomass was largely a function of the wide range of observed biomass (likely broader than what could be expected in other ecoregions) and the temporal mismatch between the date the orthophotos were taken and the time of measurement. Temporal mismatch between photos and measurements would likely to be present in virtually all analyses of this variety due to constraints in data availability. While simple multiplicative factors to relate canopy cover and biomass have commonly been used in other studies (e.g. Myeong *et al.*, 2006), the Seattle results indicate that the fraction canopy cover alone was a poor indicator of the total live biomass in these forested and urban (medium or low urban) ecosystem types.

### *Aboveground live biomass*

The aboveground live carbon stocks within vegetation were found to increase with decreasing intensity of

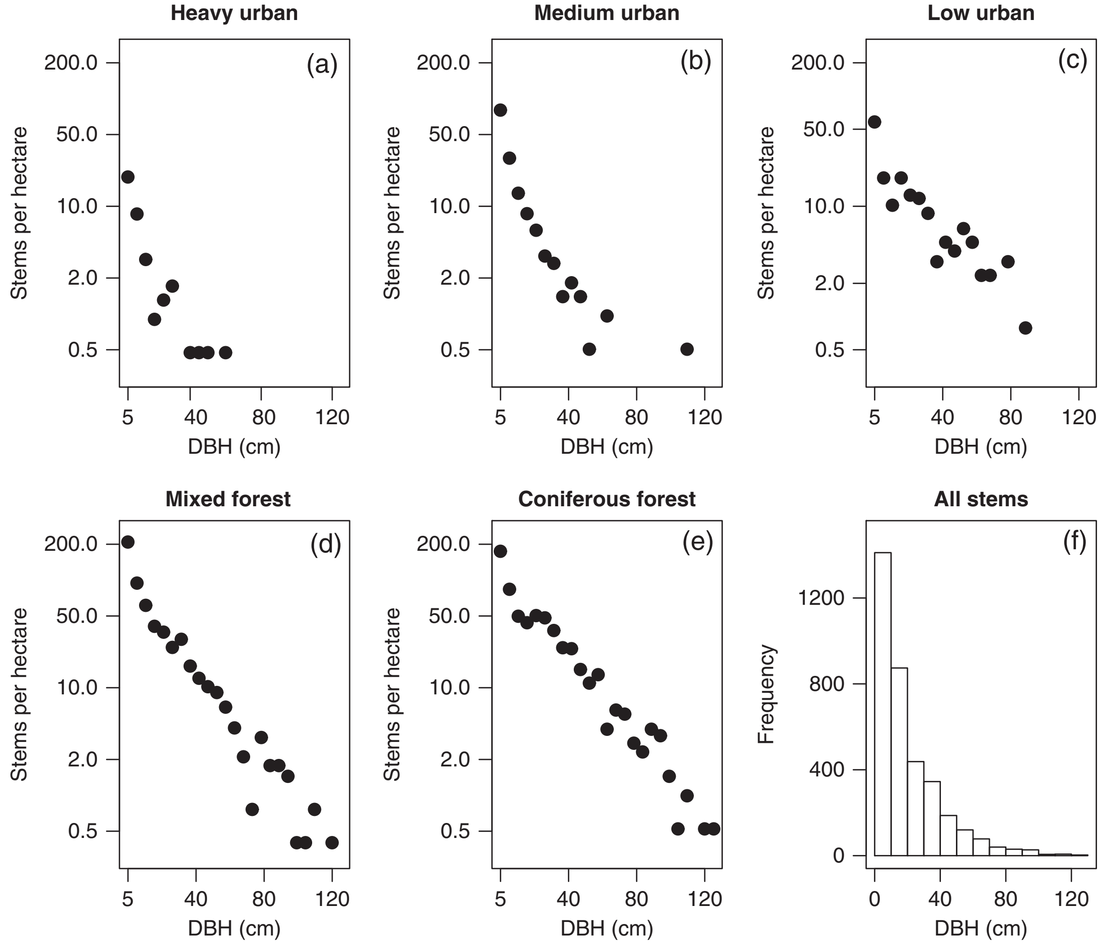


**Fig. 3** (a–e) Stem density (log scale) vs. DBH (binned by 5 cm size classes) as a function of observed land cover type. (f) Histogram of overall stem diameter distribution.

**Table 3** Estimated aboveground live carbon stocks across the Central Puget Sound area as a function of the 2002 regional land cover classification

| | Sample transects | | | Overall Sample transects | Central Puget Sound Region |
|---|---|---|---|---|---|
| | Section 1 | Section 2 | Section 3 | | |
| Heavy urban | 1.5 ± 2 (4% CC) | 5 ± 5 (12% CC) | 0.4 ± 0.5 (1% CC) | 2 ± 2 (6% CC) | 2 ± 1.9 |
| Medium urban | 18 ± 13 (26% CC) | 12 ± 12 (19% CC) | 14 ± 18 (19% CC) | 15 ± 8 (21% CC) | 13 ± 12 |
| Low urban | 26 ± 19 (26% CC) | 44 ± 50 (30% CC) | 36 ± 42 (38% CC) | 36 ± 23 (31% CC) | 38 ± 38 |
| Mixed forest | 102 ± 51 (76% CC) | 114 ± 47 (76% CC) | 96 ± 48 (77% CC) | 104 ± 27 (76% CC) | 98 ± 41 |
| Conifer forest | 159 ± 42 (90% CC) | 151 ± 62 (91% CC) | 186 ± 67 (87% CC) | 166 ± 32 (89% CC) | 182 ± 60 |
| Weighted mean | 14 ± 5.2 (17% CC) | 43 ± 10 (34% CC) | 87.9 ± 56 (59% CC) | 56 ± 32 (41% CC) | 89 ± 22 (57% CC) |

Mean values and 95% confidence intervals are reported for each section of the sample transects, the overall transects, and for the full Central Puget Sound area. Weighted mean biomass estimates are based on scaling using a 2002 Landsat land cover classification and the observed biomass distribution within a cover class in the specified spatial area. All estimates are based on only lowland areas (<500 m elevation) and the five surveyed sample classes. All biomass units are Mg C $ha^{-1}$ and the percent canopy cover (% CC) is reported parenthetically.

urban development (conifer forest > mixed forest > low urban > medium urban > heavy urban land cover), but there was little statistically distinguishable difference within observed land cover types as a function of distance from the Seattle urban core (Fig. 5). We report the mean biomass as a function of the 2002 classified

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land cover (Table 3) and as a function of the land cover visually observed in the field (Fig. 5). Note that Table 3 (Landsat-based land cover classification) and Fig. 5 (visually observed 2009 land cover) show somewhat different results. The overall biomass error estimates in Fig. 5 were smaller than those reported in Table 3 because the land cover was directly observed at the time of measurement and did not contain any pixel misclassification errors (although both still contained sampling errors). Both sets of results are reported to (1) show the patterns between observed land cover and biomass (Fig. 5) and (2) to extrapolate the plot-based estimates to the region using the Landsat classification (Table 3, Fig. 6). The inclusion of misclassified land cover pixels within the Table 3 results was important because misclassification is inherently present within any land cover classification of satellite data and needed to be included in the overall extrapolations. Across the Seattle urbanizing area, the mean aboveground live biomass within forested land covers was estimated to be $140 \pm 40$ and $18 \pm 14\,\mathrm{Mg\,C\,ha^{-1}}$ within urban land covers, based on the 2002 land cover data (Fig. 6).

A significant increase in the overall aboveground live biomass (per unit area) was observed as distance from the urban core increased due to changes in the distribution of land cover types across the area (Table 3). On a per area basis, the aboveground biomass increased from $14 \pm 5.2$ to $88 \pm 56\,\mathrm{Mg\,C\,ha^{-1}}$ between sections 1 and 3, with a commensurate increase in the percent canopy cover (from 17% to 59%, respectively) (Table 3).

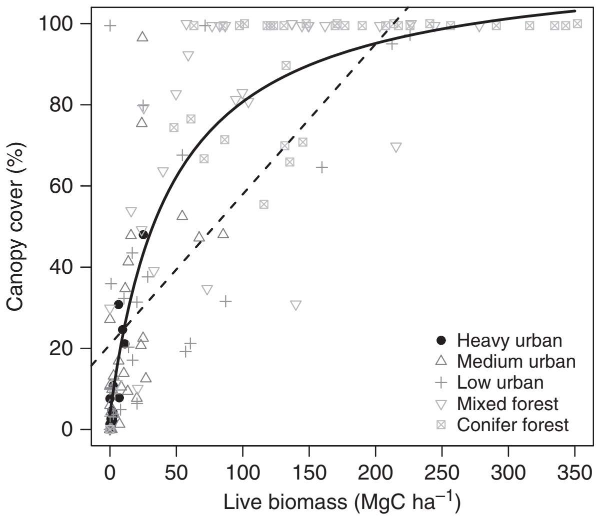


**Fig. 4** Aboveground live biomass as a function of canopy cover observed in 2006 digital orthophotos and Landsat classified land cover. The solid line shows a nonlinear least squares fit [$y = A + (B \times x)/(C + x)$, $A = 3.7^*$, $B = 112.4^{**}$, $C = 46.1^{**}$] and the dashed line shows linear least square fit ($R^2 = 0.61$). *$P$-value $\leq 0.05$, **$P$-value $\leq 0.001$.

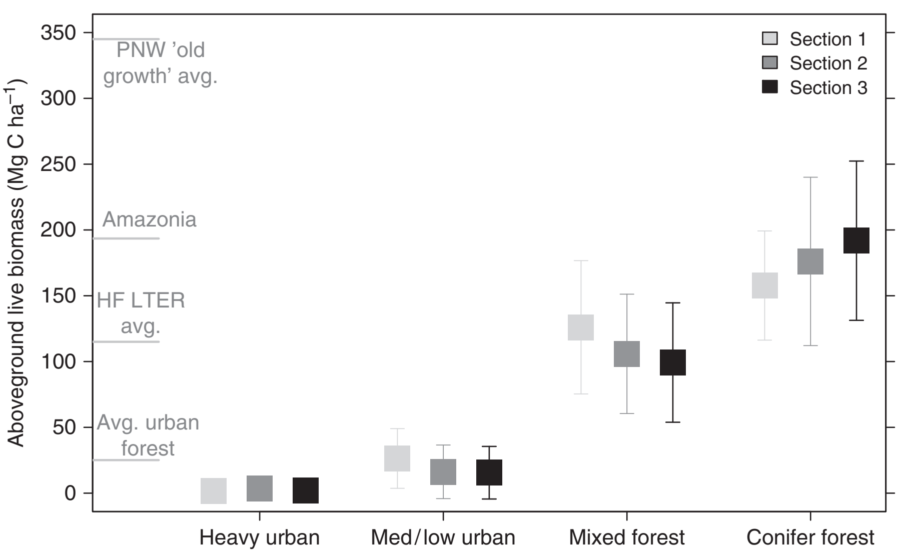


**Fig. 5** Aboveground live biomass across the Seattle Metropolitan Area as a function of the observed land cover. Section 1 extends from 0 to 7.5 km from the Seattle urban core, section 2 extends from 7.5 to 30 km from Seattle, and section 3 extends from 30 to 50 km from Seattle. For comparison, the US average aboveground urban forest carbon stocks were estimated to be 25.1 (Nowak and Crane, 2002), the average overall US forests have been estimated to hold 53.5 (urban and rural; Birdsey and Heath, 1995), the Harvard Forest (HF) LTER currently holds 115 (Urbanski *et al*., 2007), 'old growth' Amazonian forests have been found to hold 197 (Pyle *et al*., 2008), and 'old growth' Pacific Northwest forests in the Cascade region were estimated to hold 345 (Smithwick *et al*., 2002). All biomass units are $\mathrm{Mg\,C\,ha^{-1}}$.

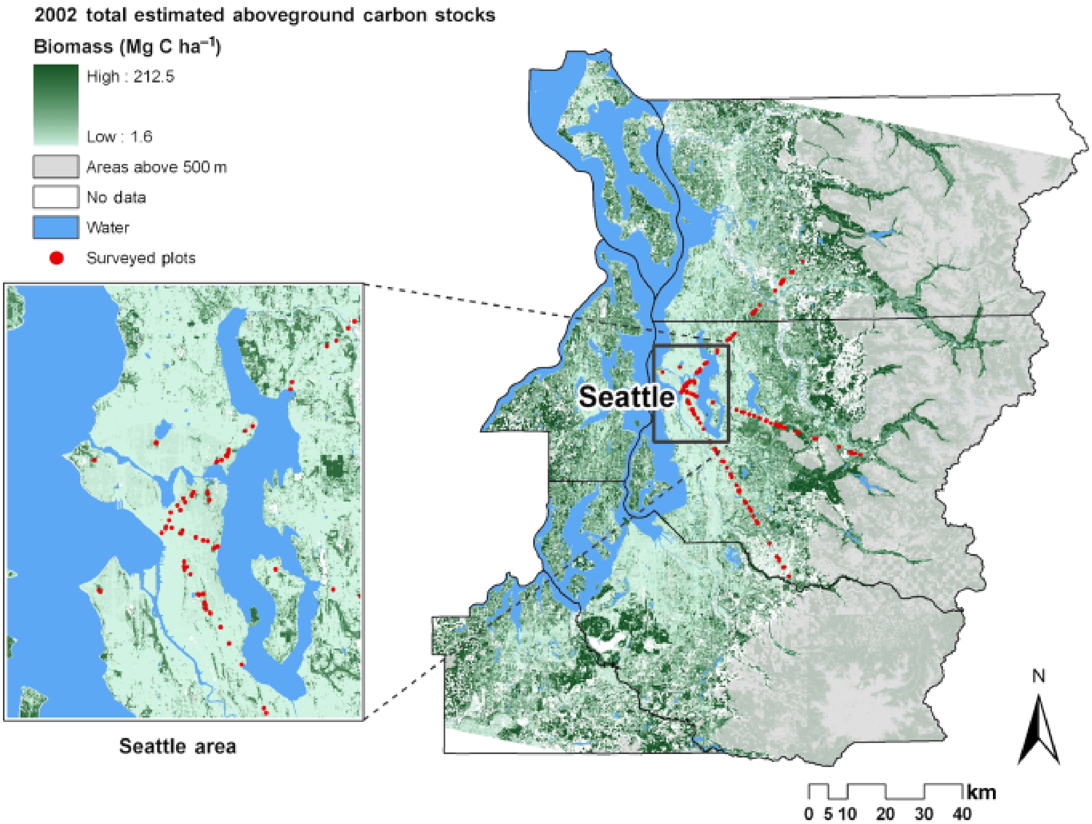


**Fig. 6** Estimated total aboveground carbon stocks for the Seattle region based on the mean values reported in Table 3 and the classified 2002 regional land cover. White areas denote areas with no estimate due to high elevation or a land cover class not assess in this study.

The overall variance in live biomass as a function of classified land cover was high due to the wide range of land uses that can occur within a given cover type; it is unlikely that an increase in sample size would significantly reduce the estimated errors without additional sample stratification based on land use.

The Central Puget Sound region is comprised of many different land covers from wetlands and agricultural lands to urban areas and forests (Fig. 1, Table 1). The five land cover classes surveyed as part of this study constituted 70% of the land area and were believed to hold the preponderance of the aboveground live biomass. Across the three sample transects there was an average of $56 \pm 32\,\mathrm{Mg\,C\,ha^{-1}}$ stored within live vegetation, based on the five surveyed land cover classes and weighted by their relative abundance. Extrapolating to the full Central Puget Sound area (Table 3, Fig. 6), the mean aboveground live biomass estimate increased to $89 \pm 22\,\mathrm{Mg\,C\,ha^{-1}}$ due to an increased proportion of forest cover in the more rural portions of the area. Regionally, 89% of the aboveground live carbon stocks were held within forested land cover, with 56% stored within coniferous forests alone.

### *CWD*

Across the Central Puget Sound region, an average of $11.8 \pm 4\,\mathrm{Mg\,C\,ha^{-1}}$ (~12% of the total aboveground biomass) was stored within dead CWD (Table 4). Within a given land cover class, the fraction of dead biomass increased slightly with distance from the Seattle urban core (Table 4), but changes in the ratio of dead to total biomass were significantly influenced by different functional uses for woody debris within the different land cover classes. For example, we found large ratios of dead to total biomass at some of the more urban sites due to low quantities of vegetation and a frequent presence of BWB (e.g. wooden fences and utility poles) rather than CWD (dead logs lying across the ground). Further, in the mixed forest land cover in section 1, a number of our sample plots fell within past forest restoration projects where the CWD was piled. CWD piles created by humans skewed the spatial distribution of CWD biomass and created very different CWD habitat conditions and subsequent decomposition rates (C turnover times).

The regional mean CWD within observed urban land covers was $1.3 \pm 2.4$ and $20 \pm 7.9\,\mathrm{Mg\,C\,ha^{-1}}$ within

**Table 4** Estimated coarse woody debris (CWD) carbon stocks across the Central Puget Sound area as a function of the 2002 regional land cover classification

| | Sample transects | | | Overall Sample transects | Central Puget Sound Region |
|---|---|---|---|---|---|
| | Section 1 | Section 2 | Section 3 | | |
| Heavy urban | 0.0 (0.1 ± 0.3 BWB) | 0.1 ± 0.2 (0.0 BWB) | 1.0 ± 1.4 (0.4 ± 0.7 BWB) | 0.3 ± 0.6 (0.2 ± 0.3 BWB) | 0.6 ± 0.9 |
| Medium urban | 0.5 ± 0.9 (0.4 ± 0.4 BWB) | 0.2 ± 0.2 (0.0 BWB) | 0.1 ± 0.2 (0.2 ± 0.4 BWB) | 0.2 ± 0.4 (0.2 ± 0.2 BWB) | 0.2 ± 0.2 |
| Low urban | 0.5 ± 0.7 (0.2 ± 0.4 BWB) | 1.1 ± 2.3 (0.5 ± 0.7 BWB) | 3.2 ± 1.8 (0.0 BWB) | 1.5 ± 1.2 (0.2 ± 0.3 BWB) | 2.6 ± 1.6 |
| Mixed forest | 6.8 ± 4.8 (0.0 BWB) | 9.1 ± 9.2 (0.3 ± 0.6 BWB) | 13.1 ± 8.7 (0.0 BWB) | 9.8 ± 4.6 (0.1 ± 0.2 BWB) | 12.7 ± 7.4 |
| Conifer forest | 11.4 ± 16.9 (0.3 ± 0.5 BWB) | 23.4 ± 14.8 (0.0 ± 0.1 BWB) | 27.7 ± 12.3 (0.0 ± 0.1 BWB) | 20.0 ± 9.1 (0.1 ± 0.2 BWB) | 27.1 ± 12.1 |
| Weighted mean | 0.5 ± 0.4 (0.2 ± 0.2 BWB) | 3.6 ± 1.7 (0.1 ± 0.2 BWB) | 12 ± 6.3 (0.1 ± 0.2 BWB) | 5.2 ± 3.4 (0.2 ± 0.2 BWB) | 11.8 ± 4 |

Mean values and 95% confidence intervals are reported for each section of the sample transects, the overall transects, and for the full Central Puget Sound area. Weighted mean biomass estimates are based on scaling using a 2002 Landsat land cover classification and the observed biomass distribution within a cover class in the specified spatial area. All estimates are based on only lowland areas (<500 m elevation). The human built woody biomass (BWB; e.g. fences, utility poles) estimates are reported parenthetically. All biomass units are Mg C ha$^{-1}$.

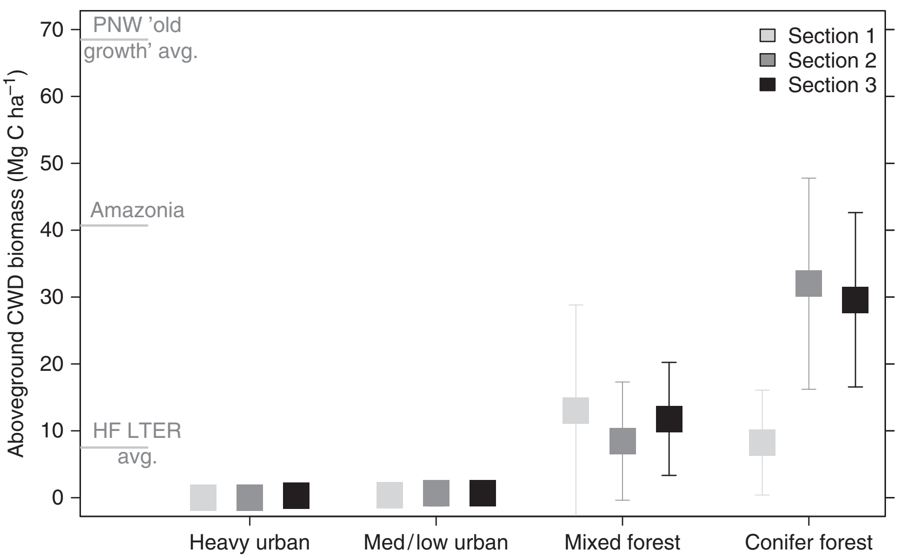


**Fig. 7** Aboveground coarse woody debris (CWD) biomass across the Seattle Metropolitan Area as a function of the observed land cover. For comparison, the Harvard Forest (HF) LTER was found to hold about 7.5 of CWD biomass (Barford *et al.*, 2001), an 'old growth' Amazonian forest was found to hold 40.7 (Pyle *et al.*, 2008), and 'old growth' Pacific Northwest forests in the Cascade region were estimated to hold 68.5 in CWD biomass (Smithwick *et al.*, 2002). All biomass units are Mg C ha$^{-1}$.

forested covers (Fig. 7). Urban land management activities often include removal of tree falls and limbs (the major sources of CWD) and the 'tidying' of urban vegetated areas (e.g. piling of CWD and clearing trails), which both reduces the total stocks of CWD and skews the spatial distribution of debris across the landscape.

The frequency distributions of the live and dead biomass differed significantly by land cover class (Fig. 8). For live biomass, the mixed and conifer forests showed approximately normal distributions, while all of the urban cover classes had skewed, long tailed distributions. The distribution of CWD was long tailed

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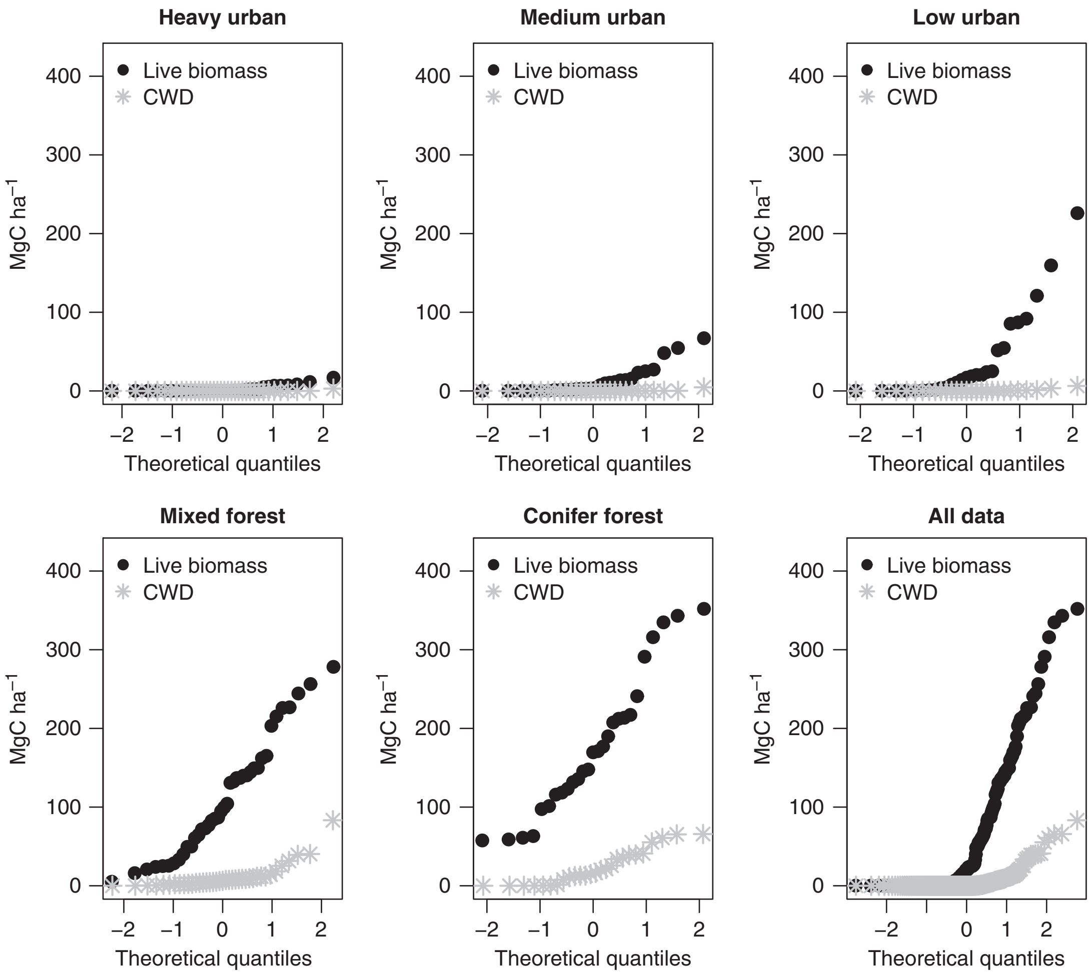


**Fig. 8** Quantile distribution plots showing live biomass and coarse woody debris (CWD) as a function of different observed land cover classes.

for all cover classes, reflecting a combination of human land management choices and the episodic nature of disturbance.

## Discussion

The carbon balance of terrestrial ecosystems is controlled by complex interactions between land cover, climate, and hydrology, and results from disturbance and recovery dynamics over timescales of years and decades. Superimposed upon these core drivers, weather anomalies influence the carbon balance on seasonal and annual timescales. Approximately, 30% of North American fossil-fuel emissions are currently offset by terrestrial sinks for carbon caused by factors such as recovery from disturbance, fire suppression, agricultural soil conservation, and woody encroachment (Pacala *et al*., 2007). Within most carbon studies, urban and urbanizing areas have been only considered as a source for emissions (associated with economic activities and pulse emissions due to land clearing). The vegetation within urban areas has been largely ignored or assumed to be negligible within the carbon cycle (Churkina, 2008; Churkina *et al*., 2010). In this study, we have found that the Seattle urbanizing region (the metropolitan statistical area) has very significant carbon stores within its terrestrial vegetation, which do play an important role in the terrestrial carbon cycle through a combination of carbon storage, carbon exchange, and urban land development choices.

The Seattle region is an excellent example of a coupled human–natural system where human and natural functions coexist. We estimated that the mean aboveground live biomass across the Seattle urbanizing region was $89 \pm 22\,\mathrm{Mg\,C\,ha^{-1}}$ in 2002 (including both urban and forest areas, Fig. 6), with an average of $140 \pm 40\,\mathrm{Mg\,C\,ha^{-1}}$ stored within urbanizing area forests and $18 \pm 13.7\,\mathrm{Mg\,C\,ha^{-1}}$ stored within urban land covers. This same actively urbanizing area is also home to over 3.8 million people (as of the 2000 US Census).

These new Seattle regional results are substantially larger than the 25.1 Mg C ha$^{-1}$ (aboveground biomass within urban forest land only) reported by Nowak & Crane (2002) for 10 US cities, and than the average of 53.5 Mg C ha$^{-1}$ for all US forests (urban and rural) reported by Birdsey & Heath (1995). The high Pacific Northwest (PNW) vegetation carbon stock values are the result of the extremely productive land characteristics within the greater Seattle region and the high overall vegetation coverage, 57% overall mean canopy cover vs. 27% mean urban tree cover reported by Nowak & Crane (2002). While the observed carbon stocks are very high, plots surveyed as part of this study did not show any signs of being 'old growth forests,' with an abundance of stumps and other signs of active vegetation management present across the plots. In most cases, 'old growth' PNW forest plots would be expected to store larger quantities of carbon than those recently disturbed given the stand structure of these conifer-dominated ecosystems suggesting that this area has additional storage potential. Large-scale forest harvests in the Seattle region began in the 1880s; nearly all of the regional forests have been managed and harvested at least once. These results highlight the importance of secondary forests within both the urban and the broader terrestrial carbon cycle.

The remarkable magnitude of observed carbon stocks in the rapidly urbanizing Seattle region is particularly clear when compared with the biomass stored in other forested ecosystems. The regional conifer forests stored an average of $182 \pm 60$ Mg C ha$^{-1}$, comparable to the $197 \pm 11.6$ Mg C ha$^{-1}$ aboveground live carbon stocks reported for a well studied, primary Amazonian rainforest (Pyle *et al*., 2008). The high Seattle conifer carbon stocks are due to a combination of large tree diameter (mean DBH was 25.7 cm) and the overall height (Douglas-fir forests average $\sim$60 m, whereas the Amazonian site had only a 40–45 m canopy height). It is also possible that the products of urbanization (atmospheric emission, runoff, amendments, etc.) itself may have fertilized these trees through enhanced urban $CO_2$ concentrations and nitrogen inputs (Lovett *et al*., 2000; Gregg *et al*., 2003), but those factors were not directly assessed in this initial study.

For further comparison, the Harvard Forest LTER in MA, USA (forest age $\sim$100 years, 115 Mg C ha$^{-1}$ aboveground biomass), a well-researched forest area, contains less aboveground carbon than the forested land covers within the Seattle urbanizing region. The Harvard Forest LTER has been observed to have a terrestrial carbon sink averaging 2.5 Mg C ha$^{-1}$ yr$^{-1}$ between 1993 and 2005, with an associated aboveground live biomass change of 14 Mg C ha$^{-1}$ (increasing from 101 to 115 Mg C ha$^{-1}$, aboveground biomass only) (Urbanski *et al*., 2007). Smithwick *et al*., (2002) estimated an upper limit for terrestrial carbon storage in Western Washington conifer forests to be $345 \pm 77$ Mg C ha$^{-1}$ in live tree biomass (aboveground biomass only). While the age and sink/source status of the Seattle area forests cannot be determined through this initial survey, much of the vegetation is certainly small in size and well below its upper storage limits. If the rate of the forestland cover conversion to intense urban uses could be slowed (whether though urban densification or increasing green space requirements when building or even modifying landscaping requirements), it is conceivable that urban $CO_2$ emissions could be partially offset by local land use choices in the rapidly growing Seattle region. But, the carbon consequences of the management activities themselves (e.g. frequent plant turnover and replanting of street vegetation or fertilizing) must also be considered when evaluating the net potential benefits. Both natural landscape disturbances and human land management practices, from forest harvests to disposal of CWD to choices in plant species assemblages, have the potential to very significantly affect the sink/source status for any given forest patch. Similar to the Harvard Forest, any potential Seattle regional carbon sink would almost certainly be in part a recovery legacy of past disturbance. While one might argue that the high live carbon stocks observed in the Seattle region are a 'unique regional anomaly' because of the species assemblage (i.e. Douglas-fir dominance), large mean height (Douglas-fir being one of the tallest known conifers), and hospitable climate, nearly every region in the United States is likely to have a unique live aboveground carbon signature due to variations in the species assemblages, sizes, and climate. Early results emerging from the CAP LTER in Phoenix, AZ, USA (arid ecosystem) suggest an opposite pattern between urbanization and carbon storage with larger vegetative stores observed within the urban areas than outside, likely due to human plantings and amendments to the system (McHale *et al*., 2009b). It is important for future local, regional, and national land regulations to capture the local ecosystem signatures in developing carbon sensitive policies.

The stock of dead, aboveground biomass within forested land covers was found to be $20.3 \pm 7.8$ Mg C ha$^{-1}$ with the dead biomass fraction composing approximately 14.5% of the total observed aboveground biomass. The relative fraction of dead organic material within the Seattle urbanizing area (both urban and forest) was higher than the CWD stocks observed at the Harvard Forest [$\sim$7% of total biomass, 7.5 Mg C ha$^{-1}$ (Barford *et al*., 2001)], but significantly less than the dead fraction observed within primary PNW forest [$\sim$17% of total biomass, 68.5 Mg C ha$^{-1}$ (Smithwick *et al*., 2002)]. The stocks of dead, aboveground carbon

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are often excluded from both field estimates and models of the terrestrial carbon cycle because they are assumed to be inconsequential due to their comparatively short turnover times and what is assumed to be a small fraction of biomass relative to the total. However, within PNW forests, the stocks of CWD are significant and, given the large size stature of coniferous trees, their turnover times are longer than in some other bioregions. Further, disturbance recovery dynamics are known to be a key mechanism influencing the strength and duration of terrestrial carbon sinks. CWD generation (rapid) and decomposition (slow) are central components within these disturbance-recovery processes and should therefore be included in both biometric carbon studies and ecosystem models for human managed and natural ecosystems alike.

## Implications and conclusion

The observations from this study provide new insights to better characterize the effects of urban development on the carbon cycle, and to design measurements that are able to capture properties along a gradient of urbanization. Ultimately, such characterization will almost certainly vary with biomes and the socio-economic context of different settlements; cross-comparative studies will be needed to create a robust set of metrics across multiple urban regions. It is increasingly evident that to answer questions, such as whether land regulations and/or tree planting strategies can offset the emissions of urban dwellers in the long term, requires accurate representations and field-based measurements that account for the spatial and temporal interactions between the built environment and natural vegetation.

Urbanization, and all of the ecosystem and emissions changes associated therein, is a fundamental driver of current and future global change. The process of urban development is multifaceted and does not have a predefined trajectory or end point. While land clearing activities do typically result in significant modification to the vegetated canopy cover (typically reduction within forest dominated areas and potentially enhancement within arid areas), both the built and vegetated structures of urban environments change and evolve over time. Urban ecosystems are inherently coupled human–natural systems (Liu *et al*., 2007), and their dynamic interactions include nonlinearities, thresholds, and ill-defined boundary conditions. As we move forward in our consideration of urban carbon dynamics, we must build from our current understanding of the terrestrial carbon cycle while proceeding with caution because we know that urban ecosystems have different biophysical structures and feedbacks.

Empirical data that accurately take into account the diverse sources and sinks of carbon in urban regions are critical to gain a mechanistic understanding of the urban carbon cycle, and to guide policy makers and planners in developing carbon-sensitive land use and transportation strategies. In the longer term, such data will provide the baseline for assessing the effectiveness of policies and define best development practices.

Many cities and regional governments are taking significant steps to reduce and offset their carbon emission [e.g. US Mayors Climate Protection Agreement, Regional Greenhouse Gas Initiative (RGGI), etc.]. King County, the most populous portion of the Seattle region, has pledged to reduce its carbon emissions to 80% below 2007 levels by the year 2050 (KCCP 2007), but the attainment of such ambitious goals will require both reduction in emissions and changes in land use, transportation, and environmental management strategies. The assessment of terrestrial carbon stocks is a key first step in defining and validating land-based components of any climate action plan.

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