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Anticipating trade-offs between urban patterns and ecosystem service production: Scenario analyses of sprawl alternatives for a rapidly urbanizing region

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ABSTRACT

Expanding demand for low-density development has restructured the urban-rural frontier throughout North America, shifting the burden of ecosystem provisioning to increasingly fragmented green infrastructure remnants. Planners have responded with approaches to control low-density development ('sprawl') that dominates North American exurbia. However, the ability of sprawl alternatives to preserve ecosystem services have not been systematically evaluated. Using a novel integration of land change simulation and ecosystem services modeling, we used proxies to estimate changes in water quality, climate regulation and biodiversity, and returns to landowners associated with sprawl alternatives and business-as-usual trends for the rapidly urbanizing Charlotte (NC) region by 2030. We found no single growth scenario simultaneously reduced pollution, stored additional carbon, and retained sensitive habitat, underscoring trade-offs likely encountered when balancing development and environmental outcomes. Watersheds at the extremes of the urban-rural gradient exhibited significantly different and often opposing responses to policies aimed at reducing environmental impacts. Scenarios of increased land use density yielded stronger financial returns to landowners as concentrated economic activity drove up land rents while minimizing broader pollution costs. Our simulated landscape approach overcame limitations associated with scale and data, and projected regional environmental outcomes emerging from local development events.

1. Introduction

In many countries around the world, urbanization patterns over the last century have fundamentally reconfigured hydrological and ecological functions at multiple spatial scales (Gosnell & Abrams, 2011; Taylor, 2011). Evidence now points to substantial overburdening of urban "green infrastructure," the network of natural and semi-natural areas that generate beneficial ecosystem functions, now known as ecosystem services (Bolund & Hunhammar, 1999; MEA (Millennium Ecosystem Assessment), 2005). Urban green infrastructure provides ecosystem services by provisioning food, water, and raw materials, regulating climate and water flow, moderating extreme weather events, providing habitat that harbors both valued species and genetic diversity, and by generating opportunities for recreation, tourism, and aesthetic appreciation (Tzoulas et al., 2007).

However, degradation of this infrastructure has led to the formation of urban heat islands (Voogt & Oke, 2003), increases in storm damage, flash flooding and the waterway impairment (Pyke et al., 2011), invasions of exotic alien species (McKinney, 2006), fish kill and algal bloom events (Bowen & Valiela, 2001), and non-attainment of air quality standards (Cardelino & Chameides, 1990), among others. Despite the widespread recognition that urban land consumption impacts green infrastructure performance (Alberti & Marzluff, 2004; Lovell & Johnston, 2009; Tratalos, Fuller, Warren, Davies, & Gaston, 2007), the specific role of development pattern—unique from magnitude—in the generation of ecosystem services remains an active area of research (Fonseca, Estévez-Mauriz, Forgaci, & Björling, 2017; Marcus & Colding, 2014).

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Given that urban expansion is inevitable for the foreseeable future (United Nations, Department of Economic and Social Affairs, 2015), not understanding the most benign ways for cities to grow is a prime challenge to reaching sustainable development goals. Planning approaches have instead focused on the proximal dilemma of reducing sprawling forms of development that dominate North American exurbia (Preuss & Vemuri, 2004). In the United States, urbanization at the rural frontier produced a 500% increase in developed area between 1950 and 2000 (Brown, Johnson, Loveland, & Theobald, 2005), much of it taking the form of disjunct, low-density ("sprawl") development patterns (Ewing, Hamidi, Gallivan, Nelson, & Grace, 2014; Theobald, 2003). Sprawl development pattern has been shown to a driving factor that compromises key ecosystem services, notably the regulation of climate-changing CO_2 and the ability of landscapes to infiltrate and purify water (Barnes, Morgan III, Roberge, & Lowe, 2001; Beach, 2003; Mentens, Raes, & Hermy, 2006; Otto et al., 2002).

Sprawl-alternatives, such as the UK "Compact City" (Breheny, 1992) and the US "Smart Growth" programs (Duany, Speck, & Lydon, 2010), aim to limit these impacts with "infill" or "clustered" patterns that concentrate development around urban amenities and infrastructure to reduce land consumption and travel costs. Comparative studies have found smart growth programs effective in shaping neighborhood design (Song, 2005); however, we know less about the regional ecosystem service impacts of these alternative patterns (Gómez-Baggethun & Barton, 2013; Preuss & Vemuri, 2004). As a result, the sustainability of such policies remains unclear to planners and policy makers.

How will ecosystem services change as a result of planned and unplanned development? Further, what are key trade-offs (the balance between desirable, but possibly incompatible services) when we develop land? For example, urban densification efforts to that preserve greenfields[1] and retain networks of green infrastructure functions (e.g. flood storage; MEA 2005), may promote higher air pollution levels and exposure. To explore these questions, in this paper we use a novel coupling of urban growth and ecosystem services simulation models to conduct scenario analyses of ecosystem service indicators for the area around the City of Charlotte, North Carolina (USA), a rapidly urbanizing metropolitan region. Given the lack of empirical case study, and the impracticality of producing test cases at relevant scales, we employ simulation modeling as a practical approach (Pickard, Van Berkel, Petrasova, & Meentemeyer, 2017; Rounsevell et al., 2012) to evaluate the response of ecosystem service indicators to different urban growth scenarios. We create these scenarios using FUTURES, a patch-based, multi-level urban growth model (Meentemeyer et al., 2013), and feed them into a suite of ecosystem service assessment models from which we estimated change in accrued benefits over time.

Using these models, we evaluate the relative impact of urban design on the generation of ecosystem services by first estimating the effect of simulated sprawl and sprawl-alternatives scenarios on indicators of green infrastructure function: 1) water purification (i.e., non-point source nutrient pollution); 2) climate regulation (i.e., carbon storage); and 3) biodiversity, (i.e., habitat provision). To better understand economic consequences of development scenarios, we also estimated future land-cover based revenues to owners (e.g. projected land rents and working lands revenues from timber harvest, crop yields) for each scenario. To facilitate comparisons among indicators, revenues and scenarios, we converted indicators to $USD and estimated the value-over-time of water purification and climate regulation services using methods described in the next section. Finally, to identify trade-offs inherent in development decisions leading to sustainable outcomes (Biggs, Schluter, & Schoon, 2015), we aggregated indicator and revenue values system-wide and watershed scales and compared scenario outcomes against a business-as-usual trajectory.

[1] "Greenfields" refer to previously undeveloped sites. Greenfield development contrasts "redevelopment" projects, which alter the use or development intensity of previously developed sites.

2. Materials and methods

2.1. Study area

Our study area centers on the City of Charlotte, North Carolina, which is part of the Piedmont Atlantic Megaregion (Fig. 1A; Regional Planning Association, 2009), and falls within six counties (Cabarrus, Mecklenburg, Iredell, Rowan, Stanly, and Union) located in the Piedmont bio-region of the southeastern United States. This region is North Carolina's most urban and is projected to grow an additional 1.2 million people (50%) by 2030 (North Carolina Office of State Budget and Management [NC OSBM], 2012). In the 19th and early-20th centuries, the region was largely bereft of forest due to plantation agriculture, but the agricultural abandonment associated with the regions' textile industry decline lead to widespread reforestation (American Forests, 2010). Today, approximately 25% of the region's landscape is urban, 25% agricultural, and over 50% is forested with Oak/Hickory/Pine upland forests, connected by 2nd and 3rd order stream networks (Singh, Vogler, Shoemaker, & Meentemeyer, 2012; Fig. 1B). These land cover proportions both fuel the rapid expansion of development and provide a rich representation of conversion opportunities for urbanization studies. During the 1990's and 2000's, the region experienced extensive low-density development, converting 30 acres [~7 ha] of greenfield per day (Meentemeyer et al., 2013).

To explore the effects of urban and exurban growth in and around Charlotte, we selected thirty-seven contiguous watersheds (at the 12-digit scale as defined by the widely used Hydrologic Unit Code [HUC] system ["HUC-12"]; Seaber, Kapinos, & Knapp, 1987) along an urban-rural gradient that is representative of many fast growing and unconstrained urban regions throughout the US (Fig. 1A). To understand potential changes to the urban-rural gradient, and facilitate comparisons in subsequent analyses, we classified watersheds into a typology set representative of settlement characteristics in North American metropolitan areas. Using unsupervised k-means methods, we identified clusters of watersheds subsequently classified as "urban", "peri-urban" and "rural" based on the relative area of projected developed, forested and agricultural (i.e. aggregated cropland and pasture) land covers (Fig. 1C).

While we expand on the scale implications of our choice of watershed as unit of analysis in the discussion section, our choice of watersheds as the unit of analysis – rather than jurisdictional (i.e. municipal boundary) or demographic (i.e. census) units – reflects our goal of measuring ecosystem service indicators at process-relevant scales (Kremen, 2005). We excluded three watersheds directly adjacent to reservoirs along the impounded Catawba River as they included significant non-gradient flows, which enormously increased hydrological modeling complexity. To reduce boundary effects during analyses, we extended the study system's 346,000 ha extent using a 1000-m buffer.

2.2. Urban growth simulations

Urban growth simulations for the study area 2007–2030 were generated by the Future Urban-Regional Environment Simulation ([FUTURES], Meentemeyer et al., 2013), a dynamic, spatially explicit raster modeling framework for simulating scenarios of per-capita land consumption and settlement patterns. FUTURES is comprised of three interacting sub-models that represent key domains of land change processes: 1) a suitability analysis of where development is likely to occur, 2) population-based projections of much land area will convert during an interval of analysis, and 3) a patch-growing algorithm that renders the spatial configuration of landscape change given historical precedent and stochastic elements designed to mimic modern development patterns. To characterize stochastic effects, FUTURES is run iteratively and the results sampled or aggregated for statistical analyses.

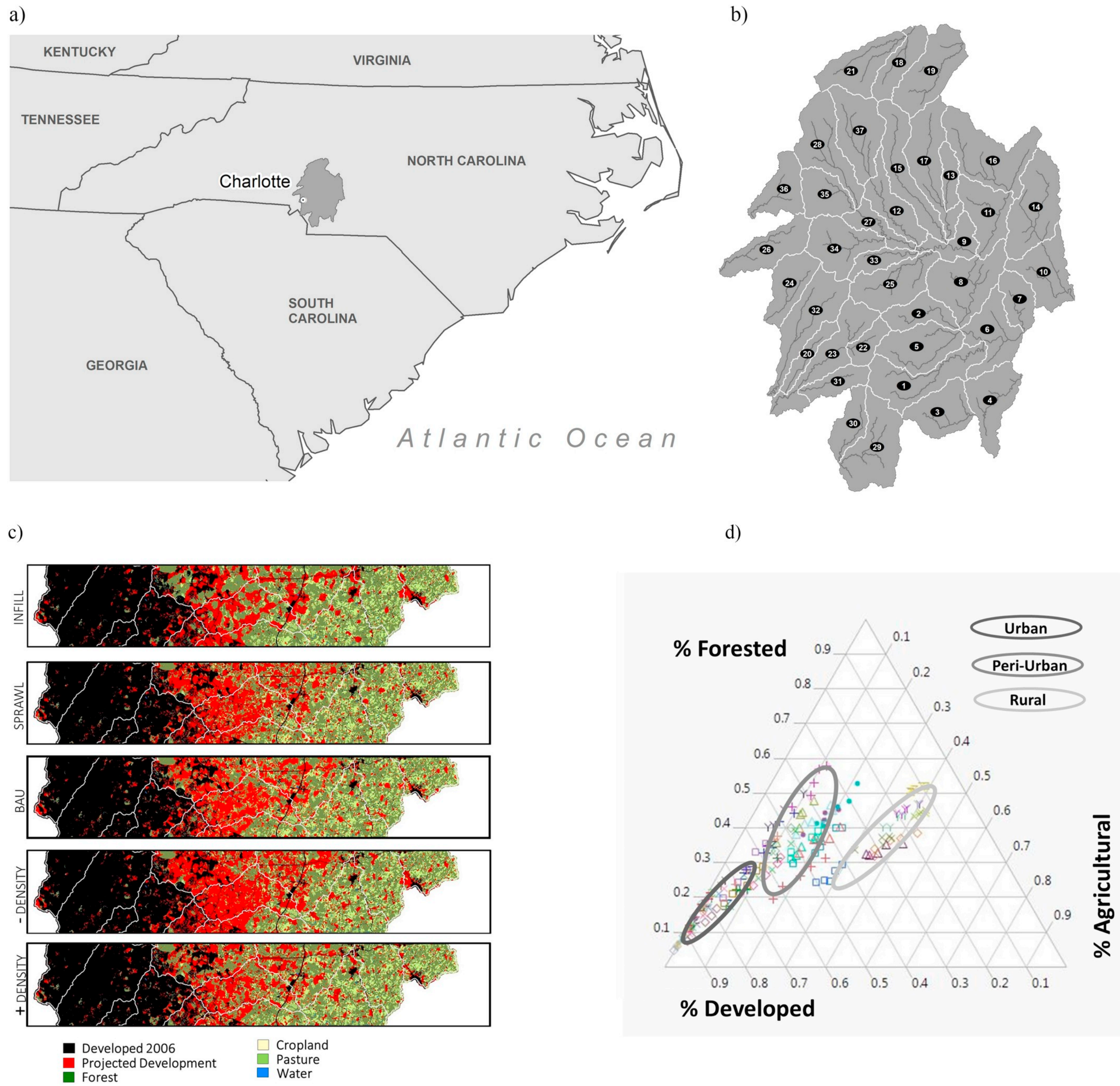

Fig. 1. Charlotte metropolitan study system. (A) Within central North Carolina, (B) 37 HUC 12-digit watersheds were mapped along an urban gradient extending West (urbanized) East (rural). (C) Detail of business-as-usual (BAU) and alternative futures projected for the region. (D) Cluster analysis identified three watershed typologies (urban, peri-urban and rural) based on relative forest, agricultural, and developed land cover.

Each iteration of FUTURES generates regional raster maps projecting developed land covers at a resolution of 30 m, for specified intervals. FUTURES has been shown to be comparatively realistic in projecting the spatial configuration of new development and can simulate relatively rare phenomena such as “leap frog” development (Pickard, Gray, & Meentemeyer, 2017). Using a back-casting method, Meentemeyer et al. (2013) estimated accuracy of the projections in the study area at 86.7%, with error distributed along a gradient of underestimation in more urban areas, and overestimations in rural areas.

Like many other dynamic urban growth models (Triantakonstantis & Mountrakis, 2012), only new development that converts greenfields (e.g. forest, agriculture) causes land cover change; previously existing development, and undeveloped forest and agricultural areas, remain static. We also assumed that undeveloped areas maintained the same land cover classes as used in the United States Department of Agriculture (USDA) National Agricultural Statistical Services (NASS) Cropland Data Layer (CDL) in 2011, which represents a conservative projection of the agricultural landscape. Like most urban growth models (e.g., Westervelt, Bendor, & Sexton, 2011), we did not endogenize road construction proxying for it instead using the patch growing algorithm described in Meentemeyer et al. (2013). Furthermore, based on historical analyses of the region 1976–2006 (Meentemeyer et al., 2013), we assumed that no forest would be converted into row-crop agriculture, that there was no agricultural abandonment leading to new forests, and that once developed, land cover classes could not revert to undeveloped types.

2.2.1. Alternative urban pattern scenarios

We developed a series of scenarios representing alternative futures for comparison with continuation of recent and current low-density development patterns (i.e., business-as-usual [BAU]). In translating generalizable urban growth narratives (e.g., those presented in Clifton, Ewing, Knaap, & Song, 2008) into formal modeling parameters, we made two key choices: 1) alternative growth patterns emerge as a result of changes in global parameters and not regional or local parameters indicative of policy or ordinance, and 2) we endeavor to disentangle the effects of configuration and composition by holding key parameters such as climate and population growth constant and altering variables of interest. The former choice recognizes that our study area is a strong property rights state, where there are no regional plans, agreements, or regulations sufficient to shift aggregate urban patterns in an organized fashion (Bendor, Shoemaker, Thill, Dorning, & Meentemeyer, 2014). The latter choice allows us to investigate the effects of the configuration of new development and abundance in the generation of ecosystem services.

With this rationale, we developed four pattern alternatives for testing in a factorial experiment (Montgomery, 2012; Fig. 1B). We designed the first factor pair of scenarios to isolate the effects of urban spatial configuration on a suite of ecosystem services indicators (below). Using FUTURES, we held population growth rate, per-capita land consumption, and patch size and shape distributions constant, and created *Sprawl* and *Infill* patterns by adjusting a development dispersion parameter (the "INCENTIVE" variable, as described by Meentemeyer et al., 2013). This parameter modifies the evenness of the development suitability surface, changing the likelihood spatially dispersed development events will emerge. After Meentemeyer et al. (2013), we applied five dispersion treatments to the projections and we report on widest range values, 0.25 for *Sprawl* and 4.0 for *Infill*. In the *Infill* case, parameterization making the suitability surface 'peakier' increases likelihood of development proximate to places that are already highly suitable, thus clustering development events closer to, for example, transportation infrastructure.

We designed the second factor pair to isolate the role of landscape composition on those same indicators. Holding population growth, development dispersion (i.e., INCENTIVE), and patch size and shape distributions constant, we adjusted a per-capita land consumption parameter, altering the relative proportion of developed-forest-farmland land covers and generating maps of *Increased Density* and *Decreased Density* scenarios. Reducing per-capita land consumption in the case of *Increased Density* places more individuals on less land, simultaneously "up-zoning" lands (i.e., increasing density) while reducing greenfield conversions (i.e., land use conversion from agriculture or forest to urban). Conversely, increasing per-capita land consumption in the *Decreased Density* scenario implied each individual in situ uses more land, thereby simulating more greenfield conversions than patterns referenced by the historical (1996–2006) BAU trend, an effect likened to increasingly widespread low-density zoning, or deregulation resulting in the relaxing of local zoning ordinances. Following Meentemeyer et al. (2013), we developed five density treatments and report on the two widest range values of 40% below (*Increased Density*) and above (Decreased Density) BAU per-capita land consumption.

2.3. Modeling ecosystem service indicators

Urban greenfields, many of which are remnants of older cultural landscapes (Barthel, Colding, Elmqvist, & Folke, 2005), comprise and encompass networks of green infrastructure that generate multiple ecosystem services (Ahern, 2007; Lovell & Taylor, 2013). Of these, we measure three—water purification, climate regulation, and biodiversity—using indicators and valuation methods described below. However, the capacity of green infrastructure to generate benefits is dependent both temporally and spatially on specific geographic contexts, connectivity, and biophysical characteristics (Ahern, 2007; Andersson et al., 2014; Andersson et al., 2015). As such, spatial planning strategies have been suggested as effective in retaining function (Bendor et al., 2014; Bendor and Doyle, 2010; Meerow & Newell, 2017). In this study, spatial dependency of ecosystem service generation creates opportunities to conduct analyses exploring cause (i.e. land cover change) and effect (changes in ecosystem service indicators) in a realistic simulation environment. Given that conversions necessarily change the spatial configuration of green infrastructure remnants, we were able to use ecosystem service modeling to expose the response of indicators to varying patterns and timing of the independent variable, land cover patterns generated by FUTURES.

2.3.1. Water purification

Land covers both contribute and retain on-site soluble nonpoint source pollutants (NPSP) loads, such as lawn chemicals or depositions of atmospheric pollution, up to a dynamic threshold bounded by the amount of impervious surface, the topography-influenced speed of water (gravity flow rates and flow accumulation paths following topographic low points; see Tallis et al., 2013 for more information), the porosity of soil, the metabolism of vegetation, and climate. When those dynamic thresholds are exceeded, exported pollutants follow hydrological gradients downhill to the next area where they may be retained (e.g. by green infrastructure), passed-on, or amplified. Owing in part to the complexity of biogeochemical dynamics, and the scarcity of monitored data at scale, modeling is frequently needed to quantify pollutant fluxes in specific topographies (see Petrucci, Gromaire, Shorshani, & Chebbo, 2014 for an example).

We modeled this process for the study area using FUTURES-produced land-cover maps and the Water Purification Nutrient Retention (WPNR) module, a spatially-explicit eco-hydrological package within the InVEST ecosystem services model (v3.1; Tallis et al., 2013). This model estimates per-cell annual nitrogen (N) and phosphorous (P) pollutant balances (loading, retention and export; [$\mathrm{kg\,yr^{-1}\ cell^{-1}}$]) based on projected land cover. We held annual rainfall and potential evapotranspiration components (annual mean temperature, annual mean relative humidity) constant, but allowed plant available water content and root restricting layer depth to change when projected to develop.

Both urban and agricultural land covers are significant sources of NPSPs, and the magnitude of loading varies by type or crop. To couple FUTURES projections with the INVEST models, we re-classified development simulated in 2030 into four intensity classes using additive analyses of three parameters determined to be significant predictors in the development of the FUTURES, including the proximity of development, road density and slope (Meentemeyer et al., 2013). Our simulations contained 50 cover types, including 42 agricultural types, four urban types (as specified above) and four forest types, to which we appended empirically determined (see Supplementary Material, Table S1) evapotranspiration coefficients, nutrient loading and vegetative filtering data added to each (Reckhow, Beaulac, & Simpson, 1980; Supplementary Material, Table S2). For each cell, we estimated land-cover supplied loads of N and P, as well as nutrient loading passed from upslope exports, retention, and exports ($\mathrm{kg\,yr^{-1}\ cell^{-1}}$). The WPNR module does not address biogeochemical interactions, and as such, soils were assumed to not saturate their ability to uptake nutrient loading. The module estimates terrestrial water and non-point source nutrient balances; no subsurface contributions or in-stream processes are considered. Following Polasky, Nelson, Pennington, and Johnson (2010), nutrient balances are not tracked in water bodies, and all exports are assumed to be delivered to the watershed outfall. Thus, the biogeochemical balances of all watersheds are considered independent. We estimated hydrologic gradients using biophysical factors and excluded engineered stormwater infrastructure due to a lack of data regarding the location and performance of such infrastructure.

Currently, the costs of non-point source pollution in the study area are unknown as they are not currently monitored; however, to attribute a value to N and P exports, we used a fee schedule from nearby watersheds as a proxy. North Carolina's Division of Mitigation Services administers a regulatory Nutrient Offset Program in the Neuse and Tar-Pamlico basins east of the study area (North Carolina Department of Environment and Resources [NCDENR], 2015). This program applies a fee schedule for offsetting N and P, which averages across these basins to \$43.85 $\text{kg yr}^{-1}\text{ ha}^{-1}$, and \$524.30 $\text{kg yr}^{-1}\text{ ha}^{-1}$, respectively. To estimate the accrued costs associated with changing land covers between 2006 and 2030, we assumed a linear rate of greenfield conversions over the period and estimated the present value of 24 years (length of model simulation) of aggregated pollution offset fees using a discounted (4%) cash flow model. These aggregated and discounted offset costs (\$USD 2015) were used to compare water purification outcomes at both system-wide and watershed scales. All estimates are developed on an annual basis, and significant intra-annual variations, such as nutrient pulses during the growing season, are not considered in our modeling.

2.3.2. Carbon storage and sequestration

Forest and other vegetative covers use photosynthesis to actively remove atmospheric carbon and sequester it in tissues and soil-building organic debris, a regulation service that in part mitigates the accumulation of climate-changing CO_2 in the atmosphere. Accumulated carbon stores are typically lost when greenfields are developed. To estimate the contribution of our study system to climate regulation we modeled changes in carbon over time using the InVEST Carbon Storage and Sequestration (CSS) module. Carbon stores (Mg ha^{-1}) for above- and below-ground biomass, soil, and dead wood stocks were estimated for all scenarios in 2006 and 2030 using an empirically derived land cover class associations (Supplementary Material, Table S3). From the 2006 start, CSS estimates a linear accumulation of carbon biomass based on land cover class. To facilitate comparison, we assumed no addition (e.g. new tree growth) or losses of carbon (e.g. timber harvests) in undeveloped land covers, there holding carbon stored in soils and biomass constant at 2011 values.

Calculated as the difference between starting and final projected stocks, we set the social value of carbon sequestration at \$60.00 USD per Mg, a value used in a recent ecosystem service assessment of Wake County, North Carolina (Schmidt, 2012). The social cost of carbon represents the marginal damage incurred for each additional ton of carbon released into the atmosphere (Ackerman & Stanton, 2012). To estimate the accrued social costs of carbon associated with land cover change over the period of analysis, we assumed a linear rate of change in carbon storage between 2006 and 2030, and report net present value of 2030 carbon sequestration using discounted (4%) cash flow modeling. These aggregated and discounted offset costs (\$USD 2015) were used to compare carbon sequestration outcomes at both system-wide and watershed scales.

2.3.3. Vertebrate habitat

Changes in land cover have direct implications for biodiversity (Seto, Guneralp, & Hutyra, 2012), with urban land covers typically favoring habitat generalists, colonizers, and other species, while excluding habitat specialists. As a proxy for biodiversity, which can be alternately considered a regulator or service itself (Mace, Norris, & Fitter, 2012), we projected changes in the amount and location of vertebrate habitat. In the study area, "urban adopters" (McKinney, 2002) include invaders such as coyote (*Canis latrans*; Wine, Gagné, & Meentemeyer, 2014), as well as many culturally and economically significant game animals, such as white-tailed deer (*Odocoileus virginianus*). "Urban avoiders" include forest interior birds and woodland salamanders, such as the spotted salamander (*Ambystoma maculatum*). The degree of intolerance for human-associated disturbances is a synoptic habitat indicator developed by the Southeast GAP Analysis Project (U. S. Geological Survey, 2011), as part of their effort to predict species distributions for 602 native vertebrates. The "Human Disturbance Index" (high, medium and low tolerance, and intolerant) is based on road density and development intensity covers, and while the GAP program uses the indicator to limit spatial distributions relative to specific species, it is used here to interactively facilitate post-hoc assessment and map change introduced by projected development patterns. Estimating the intrinsic, extrinsic and cultural values of biodiversity and its proxies, in this case habitat, is beyond the scope of this study. Instead, we reported estimates of change in habitat area and configuration for use in comparative analyses.

2.4. Returns to land owners

Land-covers generate income to land owners in the form of timber and agricultural production for managed greenfields, and rents in urban areas. Forests, croplands and pastures can, depending on management, deliver both private revenues and generate common good ecosystem services. In contrast, development typically exchanges non-rival service goods provided by the green infrastructure for exclusive rents (e.g. Bendor and Doyle, 2010; Organisation for Economic Co-operation and Development [OECD], 1992). To investigate the relationship between land-derived revenues and service provisioning by green infrastructure, we compared returns to landowners to assess ecosystem service valuations using discounted cash flow models.

We estimated annual average returns to landowners for 2006 and projected landscapes using data from Lubowski, Plantinga, and Stavins (2008) that matches land-use/land cover to revenues per unit area. Lubowski et al. (2008) estimated 1) cropland values from averaged, county-level net market returns for 21 crops; 2) pasture returns from soil productivity data (USDA NRI) and prices from NASS; 3) timber revenues for aggregated forest types based on state and regional prices, yields, costs and area; and 4) the annualized net present value of returns from urban land assuming a 5% discount rate. Estimates of per-area urban returns for higher or lower intensities in the *Increase Density* and *Decrease Density* scenarios are unknown: in order to compare plausible outcomes, urban land covers were weighted by a factor of 1.4 and 0.6, respectively, based on the ± 40% variation population density reflected in per-capita land consumption. We converted Lebowski et al.'s (2008) estimates, based on 1988–1992 data (the latest period of analysis), to 2015 dollars by averaging a range of indicators (Williamson, 2015) and applied them to both 2006 and projected landscapes, thereby facilitating comparison. In all cases, cell values were summed and aggregated by watershed, and a discounted (4% annually) cash flow model generated net present values for 2030 land cover patterns.

2.5. Analysis comparison

Following Meentemeyer et al. (2013), we generated 50 runs for each of the five scenarios (i.e. BAU, *Sprawl, Infill, Increase Density, Decrease Density)* using the FUTURES framework, and then randomly selected 10 iterations per scenario for integration with ecosystem service models. Integrated modeling outputs (*n = 50)* consisted of raster data (maps) of ecosystem service indicators, and we used spatial tools within a geographic information system (GIS) to aggregate results by both study and watershed extents, reporting mean values for each. A one-way ANOVA was conducted to examine whether there were statistically significant differences among indicators in different scenarios. Post-hoc Scheffe tests were used to evaluate pairwise comparisons between sprawl and sprawl-alternative scenarios and BAU. All statistical analyses were performed using the SPSS 22 statistical software package (IBM Corp, Armonk, NY).

3. Results

3.1. Business-as-usual land-cover projections

At the initiation of modeling, VIS analyses of 2006 remote sensing

Table 1
Ecosystem services associated with regional urban growth scenarios (2006–2030). All values for scenarios are taken from the mean values of 10 stochastic model runs.

		Start period (2006)	BAU[a]	Sprawl[b]	Infill[b]	Increase Density[b]	Decrease Density[b]
Land cover (ha)	Development (change)	75,208	179,874 (139.2%)	−4247 (−2.4)	3384 (1.9)	−19,007* (−10.6)	18,816* (10.5)
	Forest (change)	182,633	109,048 (−40.3%)	1065 (1.0)	−3989 (−3.7)	13,545* (12.4)	−13,443* (−12.3)
	Cropland (change)	25,815	17,855 (−30.8%)	453 (2.5)	730* (4.1)	1342* (7.5)	−1446* (−8.1)
	Pasture (change)	60,348	37,751 (−37.4%)	2685* (7.1)	−68 (−0.2)	4027* (10.7)	−3847* (−10.2)
Social metrics	Estimated population (change)	1,177,507	1,483,291 (26.0%)	−34 (0.0)	−182 (0.0)	−466 (0.0)	−465 (0.0)
	Per-capita land consumption (ha/person)	0.0639	0.1213	0.1184	0.1236	0.1085	0.1340
Carbon	Carbon loss, sequestration 2006–2030 (Mg)	48,873,449	46,102,001 (−5.67%)	199,617* (0.43%)	−791,436* (−1.7%)	492,908* (1.1%)	−470,460* (−1.0%)
	Present value SCC 2006–2030 (thousands $USD)	na	$-105.605	$7.612*	-$30.156*	$18.780*	-$17.925*
Nitrogen (kg)	Annual N loading	1,173,997	1,436,385 (22.3%)	−11,108 (−0.8%)	23,585 (1.6%)	−46,154 (−3.2%)	44,137 (3.1%)
	Annual N retention	806,358	820,105 (1.7%)	2854 (0.3%)	−12,172 (−1.5%)	1336 (0.2%)	−4209 (−0.5%)
	Annual N export	367,638	616,279 (67.6%)	−13,962 (−2.3%)	35,758 (5.8%)	−47,490* (−7.7%)	48,346* (7.8%)
	N accumulated exports 2006–2030	na	11,807,003	−167,548 (−1.4%)	429,091 (3.6%)	−569,883* (−4.8%)	580,157* (4.9%)
	Present value of offset costs N 2006–2030 (millions $USD)	na	$328.883	-$4.667	$11.952	-$15.874*	$16.160*
Phosphorus (kg)	Annual P loading	190,264	248,491 (30.6%)	−2688 (−1.1%)	7217 (2.9%)	−10,187 (−4.1%)	9637 (3.9%)
	Annual P retention	128,496	139,382 (8.5%)	376 (0.27%)	−685 (−0.49%)	−1252 (−0.90%)	616 (0.44%)
	Annual P export	61,768	109,109 (76.6%)	−3064 (−2.8%)	7903 (7.2%)	−8935 (−8.2%)	9022 (8.2%)
	P accumulated exports 2006–2030	na	2,050,513	−36,768 (−1.8%)	94,833* (4.6%)	−107,216* (−5.2%)	108,258* (5.3%)
	Present value offset cost 2006–2030 (millions $USD)	na	$682.995	-$12.247	$31.587	-$35.712*	$36.059*
Habitat (Ha)	High tolerance habitat	32,026	194,482 (507%)	−7858 (−4%)	−8385 (−4%)	−18,888.13* (−10%)	15,612.51 8%)
	Medium tolerance habitat	30,550	11,412 (−62.6%)	2322* (20%)	−2584* (−23%)	1554.67* (14%)	−1209.65 (−11%)
	Low tolerance habitat	160,273	83,792 (−47.7%)	19,584* (23%)	−14,033* (−17%)	8386.37* (10%)	−6948.44 (−8%)
	Intolerant habitat	123,508	56,670 (−54.2%)	−14,048* (−25%)	25,002* (44%)	8947* (16%)	−7454.42 (−13%)
Returns	Present value revenues 2006–2030 (Millions $USD)	$970.34	$23,937.55	-$389.14 (−1.6%)	$311.94 (1.3%)	$4204.27* (17.6%)	-$5619.96* (−23.5%)

na = Not applicable.
[a] Change from 2006 (%).
[b] Change from BAU (%).
* Significantly different than BAU at $p < 0.05$.

data revealed 21.7% of the study area was in developed covers, 52.7% forests, 7.5% cropland, and 17.4% pasture lands in 2006 (Fig. 1A; Table 1). Overall classification accuracy, assessed using high-resolution aerial imagery, was 86%. Study area population in 2006 was 1,177,507 (North Carolina Office of State Budget and Management [NC OSBM], 2012), yielding a per-capita consumption of 0.064 developed hectares per person (Table 1). Population was projected to grow 26% to 1.48 million, by 2030 (North Carolina Office of State Budget and Management [NC OSBM], 2012). Based on population trends, the DEMAND sub-model projected per-capita land consumption to roughly double to 0.12 ha per person by 2030. With this parameterization, we applied a calibrated FUTURES model to the region and simulated BAU growth between 2006 and 2030.

FUTURES projected that, by 2030, the landscape would be composed of 51.9% developed land, 31.5% forests, 5.2% cropland and 11.0% pasture lands given BAU trends (middle, Fig. 1B; Table 1). The anticipated 139% increase in development, and over 30% reductions in greenfields, represents a substantial restructuring of the largely-forested 2006 landscape to one dominated by low density development and high tolerance habitat. Associated with these changes were increases in average nutrient pollution loading, 22% and 30% P and N, respectively, along with a loss of an average of ~2.7 million metric tons of carbon storage, representing costs to society of over $105 million (Table 1).

3.2. Study system response to alternative growth scenarios

Table 1 aggregates the regional response of our study system to simulated urban pattern alternatives. All scenarios of urban growth increased exports of nutrient pollution, reduced terrestrial carbon stores, and restructured habitat regimes as compared to starting conditions in 2006. While we found specified designs and development rates differentially influenced projected ecosystem services, no single scenario simultaneously reduced non-point source water pollution, stored carbon, and retained sensitive habitat.

Comparisons of *Sprawl* and *Infill,* scenarios that isolated the effects of landscape configuration (i.e. the location of new development rather

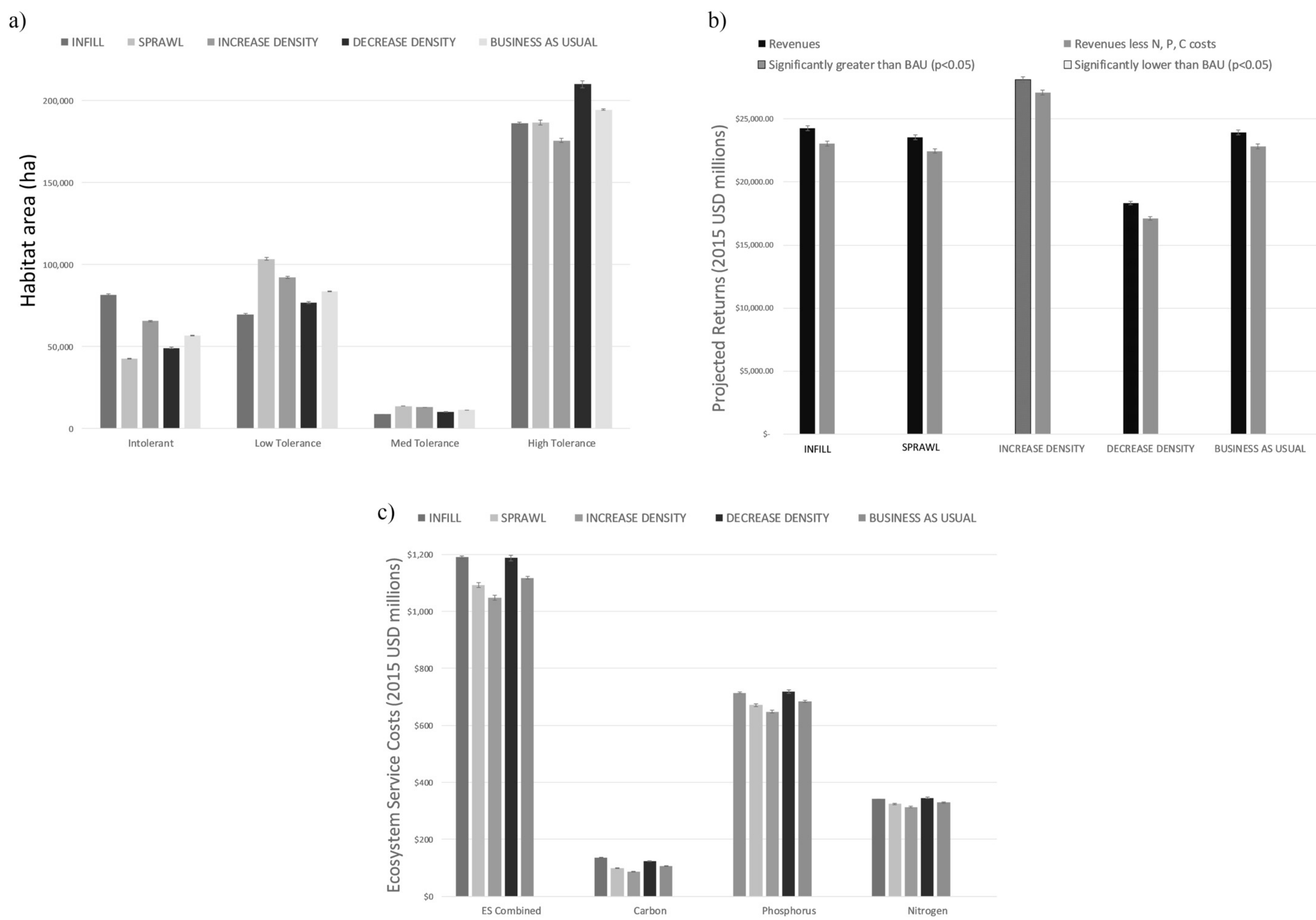

Fig. 2. (A) Scenario analyses of habitat area projections for 2030. Habitat classified using the human disturbance index (U. S. Geological Survey, 2011). For each scenario, mean area of 10 iterations is reported. (B) Scenario analyses of ecosystem services costs and projected returns to landowners attributed to land cover revenues (2006–2030). Revenues (2015 USD$) are aggregated annual returns of rents, timber sales, and agricultural yields, projected using data from Lubowski et al. (2008). For each scenario, mean net present value for 10 iterations was estimated using a discounted (4%) cash flow model. Environmentally adjusted revenues are the net present value of projected land cover revenues minus the net present value of aggregated pollution offset fees and terrestrial carbon losses (4% discount rate). (C) Scenario analyses of ecosystem service costs (2006–2030). Costs are the present value (2015 USD$) of aggregated pollution offset fees and terrestrial carbon losses estimated using a discounted (4%) cash flow model. Hatched lines indicate greater than BAU ($p < 0.05$), while dots indicate lower than BAU ($p < 0.05$).

than amount or form), revealed two counter-intuitive results. First, while all alternatives were expected to reduce stored carbon, *Sprawl* retained significantly more C than BAU and *Infill* (Table 1). *Infill* lost the most carbon storage of all scenarios considered, as it converted relatively more dense carbon stores in forested land covers, which in the study area are found proximal to extant development due to decades of urban-driven agricultural abandonment.

Second, nutrient pollution from both *Sprawl* and *Infill* landscapes did not differ significantly from BAU (Table 1), calling into question the effectiveness of infill approaches to reducing water pollution. However, on further analyses (below) we found landscape context was a key modulator of relatively better, or worse, scenario performance. *Infill's* clustered growth did retain the largest habitat area for human-intolerant vertebrates of all scenarios (Table 1).

In scenarios where landscape configuration was held constant while varying per-capita land consumption, the *Increased Density* scenario lead to significantly lower nutrient pollution than *Decreased Density* for the region as whole (Table 1). Compared with all other scenarios, *Increased Density* stored the most carbon, projected second highest level of habitat conservation behind the *Infill* scenario (Fig. 2A), and generated the most revenues (Fig. 2B). Landcover-based revenues for *Increased Density* increased 17.6% over BAU, reflecting projected intensification of human activity within constrained expansion, whereas other scenarios (*Infill, Sprawl)* held new development constant, and *Decrease Density* diluted the effect. Overall, *Increased Density* yielded stronger financial returns to landowners as concentrated economic activity drove up land rents while minimizing broader pollution costs. Decreasing per-capita land consumption retained more greenfields than other scenarios, and on average avoided over $70 million of estimated costs associated with offsetting nutrient pollution and carbon emissions, as compared to BAU (Fig. 2C).

Increasing per-capita land consumption in the *Decreased Density* scenario had the effect of converting greenfields (and therefore green infrastructure) without concurrent increases in revenue intensity. As a result, the *Decreased Density* alternative generated the highest environmental costs, second highest loss of species habitat, and the lowest returns to landowners for the revenue streams examined (see Fig. 2 A-B). The decrease in returns to land owners (primarily in land rents) followed the effect of spatially diluting urban activity.

3.3. Watershed responses to alternative urban growth scenarios

In contrast to the similarity of ecosystem service response to BAU, *Infill* and *Sprawl* scenarios demonstrated system-wide differences (Table 1), same-site comparisons of watersheds also revealed significant differences, and in many cases, we discovered that watersheds "flipped"

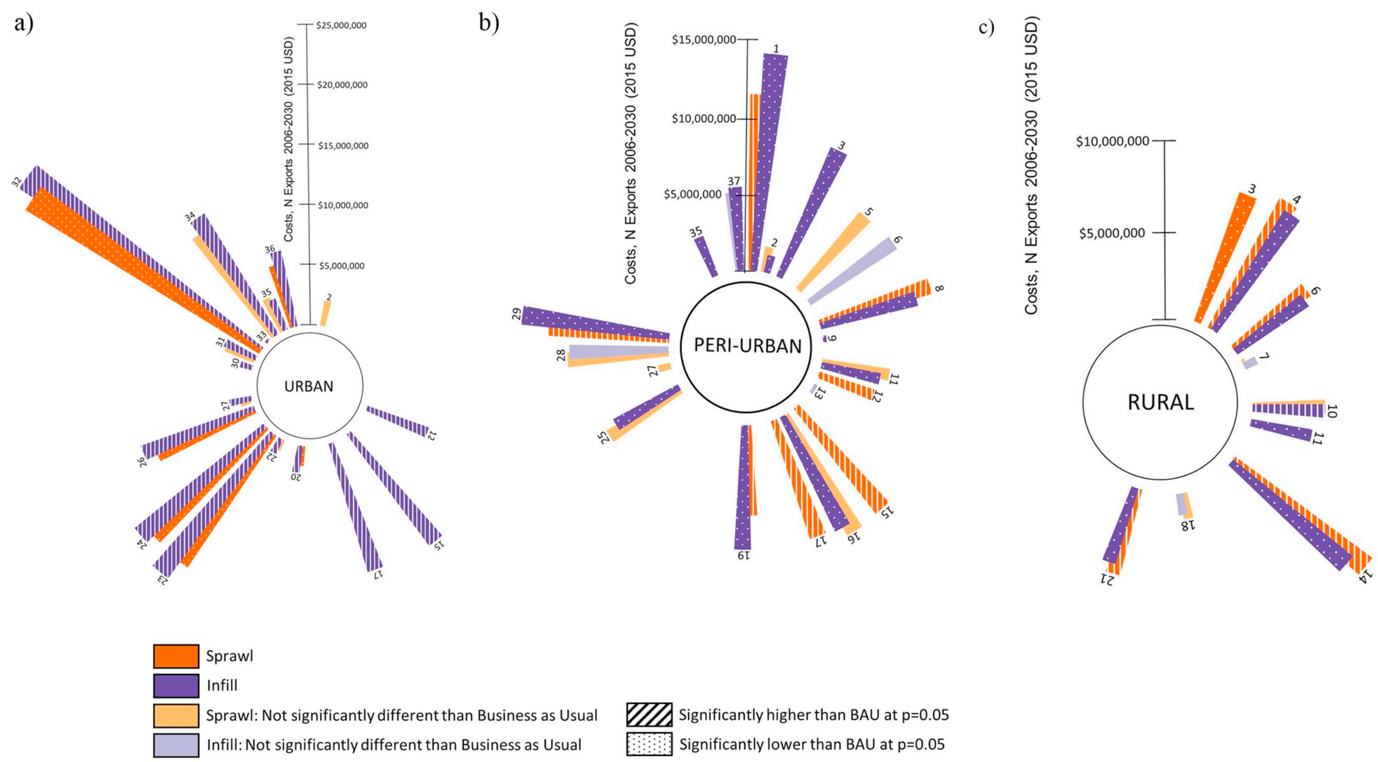

Fig. 3. Scenario analysis of projected treatment cost for non-point nitrogen pollution (2006–2030). Rose diagrams are used to illustrate relative differences in scenario effects between BAU, *Sprawl* and *Infill* scenarios for each watershed, as well as contrary responses observed between typologies. Each petal in the rose diagram designates a numbered watershed (see Fig. 1B), and the length of the petal represents aggregated cost ($USD 2015) accrued over the period of analyses. The number of watersheds varied in each typology, in response to scenario treatments and stochastic effects, and we include non-significant results to portray cluster membership. While the measured effects of *Sprawl* and *Infill* varied significantly in comparison to the reference BAU, costs were positive as all growth scenarios exported more nitrogen than the 2006 landscape. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

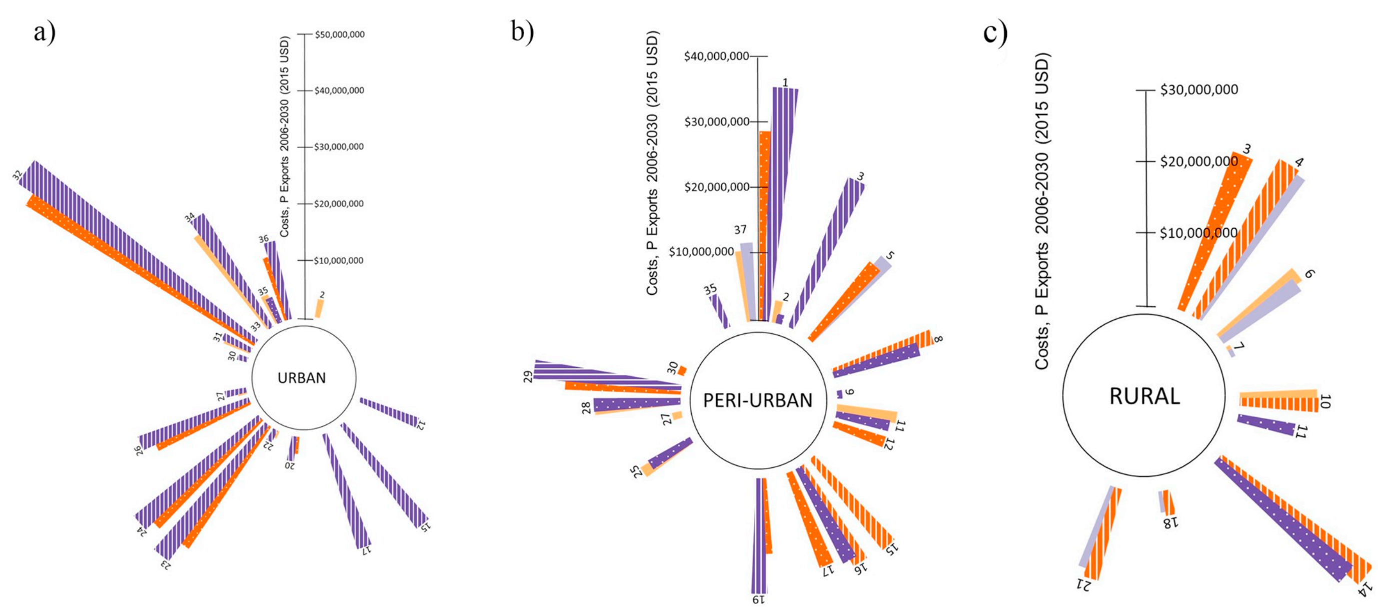

Fig. 4. Projected treatment cost for non-point phosphorus pollution (2006–2030) based on watershed response to *Sprawl* and *Infill* scenarios. While the measured effects of *Sprawl* and *Infill* varied significantly in comparison to the reference BAU, costs were positive as all growth scenarios exported more phosphorus than the 2006 landscape.

the directionality of effect, with the same watershed producing more or less pollution than BAU based solely on configuration associated with *Infill* or *Sprawl* (Figs. 3–5). We found that 56% of the watersheds changed the directionality of effect for one of the three ecosystem service values (N, P, or Carbon), 48% to two and 27% to all three, when *Infill* was compared with *Sprawl*. This behavior was not found in comparisons of *Increased Density* with *Decreased Density*.

Given that the amount of new development in these scenarios was

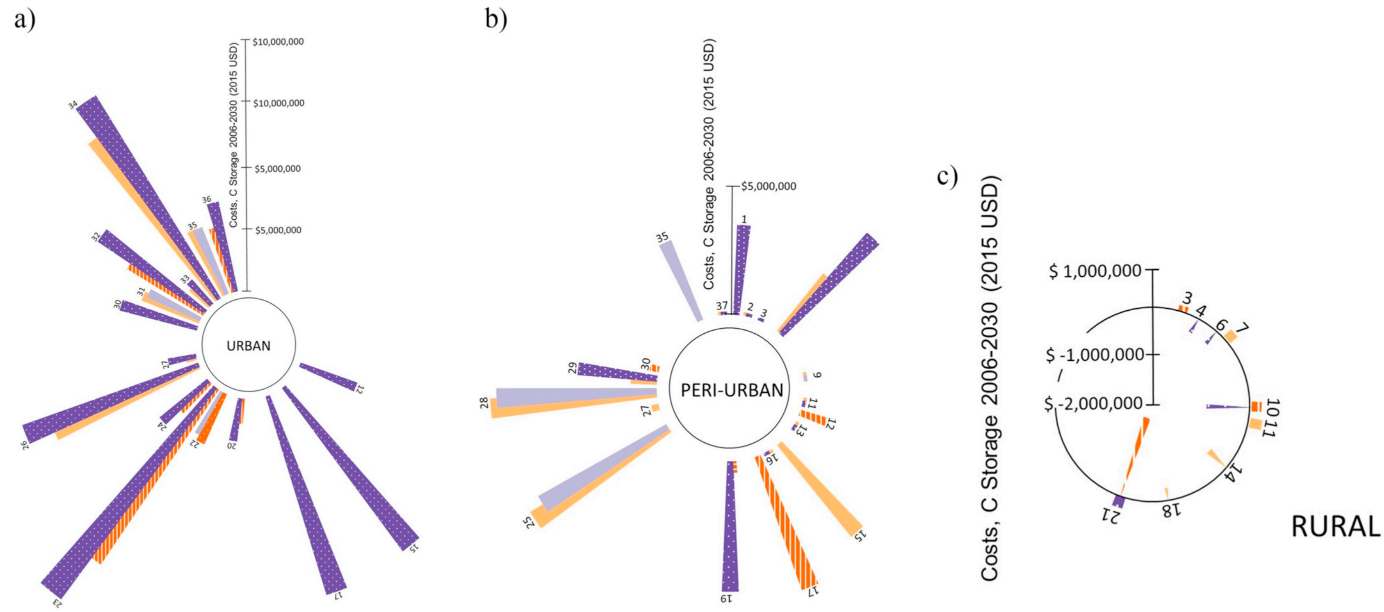

Fig. 5. Projected costs associated with changes in carbon storage 2006–2030 based on watershed response to scenario-based changes in land cover configuration. Avoided costs, associated with carbon sequestration, are represented as inward radiating petals.

controlled to be equal, we looked for significant correlation between watershed characteristics (e.g. topography, soil type) and sensitivity to configuration and found none. However, variable screening techniques indicated the best predictor of this behavior was urban typology. Our application of unsupervised k-means methods had identified three clusters (cubic clustering criteria [CCC] = 4.132; see Milligan & Cooper, 1985) of watersheds based on proportionality of land covers (Fig. 1C). Overall, 35.5% of watersheds were classified as "Urban", 20% as "Rural" and 44.5% "Peri-urban". Rural watersheds within the study systems were relatively balanced between urban, forest, and farmland covers, while peri-urban and urban had less farmland and were increasingly skewed toward urban types. For a third of the watersheds, cluster membership varied with stochastic iteration of landscape simulation, where ten watersheds (27%) alternated between peri-urban and urban classification clusters based on scenario, and two between rural and peri-urban. When used in stepwise multinomial logistic regression, "urban" and "rural" and "peri-urban" typologies achieved a 58.0% accuracy predicting which watershed exhibited a "flipped" response to BAU urban growth scenarios.

For N and P pollution, we found *Sprawl* scenario simulations were less costly than *Infill* scenarios in Urban watersheds, but costlier in Rural (Figs. 3 and 4). Results were mixed in Peri-Urban watersheds. This suggests that in areas dominated by urban land covers, *Infill* was costlier as the scenario increased the contiguity of impervious surfaces, reducing opportunities for the absorption or uptake of nutrient pollution.

Pattern observed for carbon differed in that rural watersheds were projected to store carbon over the period of analysis, registering as an avoided cost (Fig. 3A). However, urban areas lost carbon, and as in the case of N, P, *Sprawl* was less costly than *Infill*. Results were mixed in Peri-Urban watersheds. The relative advantage of *Sprawl* may be due to the character of disjunct development, which by definition is discontinuous, leaving in its wake forest remnants that continue to sequester and store carbon.

4. Discussion

In this study we used an integrative modeling strategy to anticipate future trade-offs encountered between urban growth and the environment in a rapidly urbanizing landscape. Using scenarios comprised of land cover simulations as the manipulated variable, and ecosystem services indicators as a response, we found that the spatial configuration of new development and green infrastructure, as well as the magnitude of change, had significant consequences for water quality, the retention of climate-changing carbon, and habitat for vertebrates with varying tolerances to humans. The most prominent response was observed when density was manipulated, resulting in either the concentration of revenue-generating activity while minimizing environmental costs, or the dilution of activity with expanded costs. We also detected a sensitivity to the location of growth along the urban-rural gradient. There, watersheds exhibited differential and often opposing environmental response to sprawl alternatives, an effect statistically correlated with urban, rural or peri-urban typologies, our proxy for geographic context. Despite our conclusion that no scenario (i.e. *BAU, Sprawl, Infill, Increase Density, Decrease Density*) simultaneously reduced pollution, stored carbon, and retained sensitive habitat, we found that in some cases planned alternatives to sprawl resulted in more benign environmental outcomes than taking no action at all.

Given the likelihood and perhaps necessity of future urban expansion, our results provide some guidance as to the relative impacts of regulatory decisions ostensibly in control of regional planners: density and configuration. A finding cutting across all iterations was that *Increase Density* yielded stronger financial returns to landowners as concentrated economic activity drove up land rents while reduced land consumption minimized broader pollution costs. Decreases in per-capita land consumption associated with the *Increase Density* scenario resulted in fewer greenfield conversions (as compared to BAU), which avoided over $70 million of estimated costs associated with offsetting nutrient pollution and carbon emissions (Fig. 2C), because population growth was accommodated while minimizing spatial accretion of impervious surfaces. These findings support suggestions that the promotion of urban density can foster sustainable development (Rees & Wackernagel, 2008; Brabec & Lewis, 2002; Næss, 2009), especially when geographic context is considered.

Configuration was important too, but not how we expected. *Infill*, conceptualized as an environmentally friendly growth alternative, was statistically no different than *Sprawl* and BAU for NPSP study-wide, and in fact reduced landscape carbon as forests were converted at relatively higher rates (Table 1). However, this finding was contextualized by pairwise comparisons of watersheds, where we found *Infill* and *Sprawl* had a significant and opposed response to scenarios, depending on location along the urban-rural gradient (Figs. 3B, 4B and 5B). In this way, offsets masked the significant and counter-intuitive finding that urban watersheds exhibited lower environmental costs with *Sprawl* scenarios than with *Infill*. Conversely, *Infill* patterns exhibited lower environmental costs in rural watersheds than *Sprawl*. The effect was mixed in peri-urban watersheds possibly reflecting a transitionary location on an urban-rural trajectory.

Further research is needed, but it appears that greenfield remnants associated with disjunct patterns of sprawl provided important opportunities for green infrastructure to regulate services through infiltration and metabolic uptake of NPSP. *Infill*, as generalized in our simulations, converted relatively more remnant greenfields and regrowth forests, removing many opportunities for uptake and increasing the contiguity of impervious surfaces. In rural watersheds, the comparatively small-scale *Infill* clustering appears to stay within broad uptake thresholds provided by dominant greenfield land covers. While infill development represents a planning paradigm that underpins many modern development regulations, these results indicate that a blanket focus on infill as a green alternative to sprawl could be problematic (Pickard, Gray, & Meentemeyer, 2017). Instead, our findings point to the benefits of retaining greenfields behind the development frontier in rapidly urbanizing areas.

Infill did have the effect of retaining more habitat suitable for human intolerant vertebrates than any other scenario (Fig. 2). Declines of amphibians, forest-interior birds and other urban avoiders in North Carolina have been well documented (Wear & Greis, 2002). Despite ongoing environmental presses such as climate variability and pathogen introduction, strategic planning is first among recommended conservation actions (Scheffers & Paszkowski, 2012). Overall, the *Increase Density* scenario retained the most vertebrate habitat, illustrating the potential for trade-offs between intrinsic and extrinsically valued resources (Dorning, Koch, Shoemaker, & Meentemeyer, 2015).

To be clear, all scenarios of alternative futures tested resulted in more pollution, losses of carbon, and irreparable changes to habitat as compared with the 2006 starting conditions. We estimated that by 2030 nutrient pollution exports to surface waters across the region would increase over 65%; that the Charlotte region will lose over 2 million metric tons of stored carbon from forests and soils, and that the composition of landscape will transition from one dominated by low human disturbance habitat to one of overwhelmingly high disturbance. In contrast to this environmentally dreary prognosis is the likelihood that landowners will enjoy increased revenue, the region generating > 24 times current returns as conversion replaces low-yielding forests and pastures for high-yielding urban covers.

Significant extensions of this research could evaluate the accuracy of ecosystem service assessments at smaller spatial scales or longer temporal scales (e.g., 2050 or 2100, although urban growth projections likely have limited meaning at this time horizon). Further exploration of spatial scale is particularly important to test the applicability of this modeling strategy to other regions and contexts; for example, regions with regional planning mechanisms or poly-centric urban centers that have strong land use planning regulations (e.g., zoning, urban growth/service boundaries) may have different abilities to steer ecosystem service provisioning in the future. Other regions, particularly those in the developing world, may have significantly sparser data available, preventing accurate calibrations of the underlying FUTURES and InVEST models. Additionally, future work should endeavor to normalize ecosystem service cost projections by population, thereby informing narratives around the relative ecological "footprints" of urban and rural residents.

That said, our use of scenario-based, land change simulations to inform ecosystem service modeling has constituted a safe-to-fail experimental strategy, which is capable of anticipating some of the long-term socio-ecological consequences of urban policy. This strategy has also sought to address some of the issues associated with spatio-temporal scale (McGarigal & Cushman, 2002) and the limited availability of relevant environmental data, such as those for nonpoint source pollution (Rissman & Carpenter, 2015).

5. Conclusions

Anticipating the aggregated effect emerging from regulatory choices has long constituted a challenge to planners looking for sustainable development outcomes (Nassauer & Opdam, 2008; Termorshuizen & Opdam, 2009; Williams, Jenks, & Burton, 2000; Wu, 2010). In this study, we demonstrated the feasibility of projecting targeted, long-term environmental effects of development using simulations generated through integrated modeling. Though complex, the approach has estimated non-linear biophysical interactions between topography, land composition, and time at fine spatial grain, revealing emergent environmental responses at functional scales (e.g. watersheds). The use of ecosystem service indicators as a response revealed complex, and often non-linear outcomes associated with regional planning decisions, and may facilitate bridging planning, management and governance practices when transitioning to more sustainable cities (McPhearson, Andersson, Elmqvist, & Frantzeskaki, 2015). Integrating urban growth and ecosystem services modeling represents an approach that advances the planning for ecosystem services paradigm (Bendor, Spurlock, Woodruff, & Olander, 2017; Cabral, Feger, Levrel, Chambolle, & Basque, 2015; Goldstein et al., 2012; McPhearson et al., 2015; Woodruff & Bendor, 2016) by providing sufficient detail and complexity to preemptively inform response to future environmental challenges.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.compenvurbsys.2018.10.003.

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