Analyzing the impacts of land use policies on selected ecosystem services in the upper Chattahoochee Watershed, Georgia, United States

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Abstract
The rapid rate of urbanization within the Upper Chattahoochee Watershed (UCW) is threatening the provision of ecosystem services (ESs) for six million residents of the Atlanta Metropolitan Area. This study uses the land cover change model TerrSet to project future land cover from 2016 to 2040. The modular toolset InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) is used to assess the efficacy of four land use policies in maintaining the provision of three ESs (carbon storage, wildlife habitat, and water quality) within the UCW. The Baseline scenario represents past urbanization trends, whereas the Urbanization scenario accounts for a higher urban growth rate. The Plan 2040 scenario includes existing policy guidelines, and the Conservation scenario adds forested riparian buffer areas. Two integrated indexes and an economic valuation of ESs were used to combine all ESs and analyze the overall performance of each policy. The first index uses unequal weights for ESs based on the Analytical Hierarchical Process, whereas the second index uses equal weights. The values of both integrated indexes and economic values were highest in the Conservation scenario and lowest in the Urbanization scenario. No significant differences in the provision of ESs were found between the Baseline and the Plan 2040 scenarios. However, the integrated indexes and economic values for both land use policies declined over time. Our study will feed into the ongoing movement of sustainable watershed management for ensuring the provision of ESs, especially for rapidly urbanizing cities worldwide, in general, and in the United States, in particular.

1. Introduction

The world’s urban population is expected to increase from four billion in 2016 to almost six billion in 2040 (United Nations 2018). In the United States, about 82% of the population lives in urban centers, with projections suggesting that this percentage will reach 87% by 2040 (United Nations 2018, World Bank 2018). The Atlanta Metropolitan Area (AMA), located in Georgia, is among the ten most populous metropolitan areas in the Southern United States and is experiencing one of the fastest population growth rates in the country (Liu and Yang 2015). The AMA had a population of about 5.8 million people in 2016, and it is projected that its population will reach a total of 7.7 million by 2040 (Governor’s Office of Planning and Budget 2019).

Urbanization is characterized by land cover transitions from natural areas to developed lands. Urbanization typically results in the reduction of the provision of ecosystem services (ESs), i.e., benefits people gain from ecosystems (Millennium Ecosystem Assessment 2005). As a result, several adverse environmental impacts are associated with urbanization, such as lower air quality, increased greenhouse gas (GHG) emissions, and a higher likelihood of flooding (Wilson and Chakraborty 2013). Moreover, the deforestation and forest fragmentation caused by urban expansion negatively affect wildlife habitat integrity and quality (Miller 2012). Additionally,
high urban land cover and high population density are related to high concentrations of water pollutants at a watershed level, resulting in lower water quality (Tu et al. 2007).

A variety of strategies, such as regulations and the establishment of public or private protected areas, are used to address the impacts of urbanization on the environment. Public land use policies often include conservation components to maintain the provision of ESSs. For example, protection of riparian buffer areas (Garrastazú et al. 2015, Wallace et al. 2018) and zoning of environmentally sensitive areas (Munroe et al. 2005), are two common strategies often employed by public land management agencies for ensuring the provision of ESSs. It is important to characterize the impacts of public land management policies in light of urban expansion. As a result, several models have been developed for simulating the effects of urban expansion under different land management scenarios. For example, Envision is a GIS-based tool that couples human and natural systems for ascertaining the impacts of urban growth under different scenarios (Bole et al. 2007). Similarly, FUTURES (FUTURE Urban-Regional Environment Simulation) is another tool that accounts for site suitability, per capita demand, and spatial patterns of past land use transitions for projecting future land use (Meentemeyer et al. 2013). Additionally, several platforms exist which model changes in ESSs relative to changes in land cover change. For example, InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) is a popular suite of models used to quantify, map, and value the ESSs (Sharp et al. 2016). Similarly, ARIES (ARTificial Intelligence for Ecosystem Services) is another platform for modeling ESSs. It focuses on beneficiaries and flows of services, employing an algorithm to choose appropriate ESSs models from a modular library (Villa et al. 2014).

A need exists to merge land cover change models and models that quantify ESSs to estimate the impacts of urbanization on the provision of ESSs in a dynamic setting (Daily et al. 2009, Olmedo et al. 2015, Sharps et al. 2017). Additionally, it is important to integrate the economic values of ESSs using various non-market valuation techniques into an integrated model for facilitating decision-making processes (see Champ et al. 2017 for a review). Finally, a need exists to develop an integrated index that addresses a suite of ESSs in a holistic manner, so a single indicator can inform decision-makers of the overall outcome of alternative public land use policies (Bodini et al. 2012, Sun et al. 2018). This index could integrate biophysical indicators or economic values of ESSs, which are both explored in this study.

A substantial portion of the urban growth that is happening in the AMA takes place within the Upper Chattahoochee Watershed (UCW), which supplies most of the drinking water consumed by the citizens of this metropolitan area (Atlanta Regional Commission 2010, Metropolitan North Georgia Water Planning District 2017). In addition to water, the UCW provides other essential ESSs to the region, such as habitat for wildlife and carbon storage (Sun et al. 2018, Benez-Secanho and Dwivedi 2020). Therefore, with the goal of analyzing the future environmental implications of alternative public land use policies within the UCW, this study sets three objectives. The first objective simulates the effects of selected land use policies on the watershed’s land cover. The second objective determines how a suite of ESSs (water quality, carbon storage, and wildlife habitat), selected based on their local importance and data availability (Lin et al. 2018), would be affected by the urban growth within the UCW under selected land use policy guidelines. The third objective assesses the overall provision of selected ESSs under these policy guidelines using three different metrics (biophysical indicators using equal weights and different weights derived using the Analytical Hierarchical Process (AHP) methodology, and economic values of ESSs). A better understanding of current and projected urbanization within the UCW and its effects on the provision of ESSs will assist decision-makers in improving and developing newer sustainable public land use policies for integrated development.

2. Study Area

The UCW is located in the northeast of Georgia, United States, and has a total area of 413,936 ha (figure 1). Its northern borders lie within the Appalachian Mountains, and the AMA is located in its southern portion. The average elevation of the watershed is 408 m above sea level (United States Geological Survey 2019), with an average annual precipitation of 1,459 mm and a mean annual temperature of 15 °C (PRISM Climate Group 2019). In 2016, 34.2% (141,756 ha) of the UCW was covered by urban lands, while forests and pastures represented 50.6% (209,662 ha) and 8.7% (36,032 ha) of the watershed, respectively. From 2001 to 2016, the UCW lost 11,304 ha of natural lands and 4,504 ha of pasture and cultivated crops, gaining 16,657 ha of new urban developments (Yang et al. 2018).

3. Public land use policies in effect within the UCW

Currently, various public policies partially or to the full extent aim to frame future land use and promote the conservation of ecosystems within the UCW. First, the Atlanta Regional Commission (ARC) Plan 2040, through its Regional Development Guide (Atlanta Regional Commission 2011), classifies the AMA in different areas (e.g., Region Core and Rural Areas) and places (e.g., Regional Centers and Town Centers), and set guidelines to
provide directions for their future growth. The Water Resource Management Plan, created by the Metropolitan North Georgia Water Planning District (Metropolitan North Georgia Water Planning District 2017), presents plans for watershed management, water conservation, and wastewater management of the region, which includes most of the UCW. Finally, the Urban Redevelopment Plan addresses the economic development of Gainesville and Hall County, with an emphasis on reducing the adverse effects of urbanization on the environment (City of Gainesville and Hall County 2012). In addition, several laws focus on establishing stream riparian buffer protection in Georgia (buffer areas), with their width ranging from 8 m to 46 m depending on the characteristics of the water bodies (appendix A7). Buffer areas are only required for state waters (defined as water bodies that are not entirely contained within a property (parcel) owned by a company, individual, or organization) with wrested vegetation (Georgia Environmental Protection Division 2017). Finally, there are multiple types of conservation lands within the UCW. According to the Protected Areas Database of the United States (PADUS), 83,100 ha were under some level of protection within the UCW in 2016, which represents 20.2% of the total area of the watershed (United States Geological Survey 2020).

4. Methods

This section is divided into a total of six sections. The first section details projection of land cover changes in the study area. The second section provides details of four land cover change scenarios selected for the study. The third section describes steps taken for modeling three ESs (Water quality, carbon storage, and wildlife habitat quality) in the study area. The fourth section focuses on the steps taken for determining the economic value of selected ESs. The fifth section covers details of two integrated biophysical indexes used to compare the provision of ESs across scenarios. Finally, the last section details the statistical analyses used for ascertaining any differences in the provision of ESs relative to the baseline scenario. The novel approach taken in this study is unique as it combines three indexes (one economic and two biophysical) to provide decision-makers with alternative tools to analyze the provision of ESs within the UCW relative to potential land cover change scenarios.

4.1. Projection of land cover change

We used the module Land Change Modeler (LCM) embedded in the software TerrSet 18.31 (Eastman 2016) to project future land cover changes under different scenarios. This approach is consistent with the literature and
Table 1. Driving factors used in the five multi-layer perceptron (MLP) sub-models. Evidence likelihood spatially displays how likely a pixel would change from one class to another (an evidence likelihood layer is generated for each unique land cover transition). Urban disturbance represents areas that experienced a transition from any natural land cover class to an urban class. The transitions modeled in each sub-model are shown in parenthesis (modified classification).

| Variable                                      | Low² to High (2 to 3) | Forest to Low (5/6 to 2)³ | Forest to High (5/6 to 3)³ | Pasture/Shrub to Low (7/8 to 2)³ | Pasture/Shrub to High (7/8 to 3)³ |
|-----------------------------------------------|-----------------------|---------------------------|---------------------------|----------------------------------|----------------------------------|
| Distance to urban disturbance                 | X                     | X                         | X                         | X                                | X                                |
| Distance to retailers                         | X                     | X                         |                           | X                                | X                                |
| Evidence likelihood                           | X                     | X                         | X                         | X                                | X                                |
| Distance to highway nodes                     | X                     | X                         |                           | X                                | X                                |
| Distance to roads                              | X                     |                           |                           | X                                | X                                |
| Distance to highways                           | X                     |                           |                           |                                  |                                  |
| Distance to regional urban centers            | X                     |                           |                           |                                  |                                  |
| Household median income (tracts)              |                       |                           |                           |                                  |                                  |
| Accuracy rate (%)                             | 80.15                 | 83.36                     | 84.45                     | 83.14                            | 82.02                            |

² Low: Developed (open space); High: Developed (Low to high-intensity classes).

can be readily adapted at the watershed level (Nelson et al 2009, 2010, Polasky et al 2011). The LCM requires two land cover maps (described in Section 4.2 and Appendix A8) from distinct periods as input data to analyze transitions among land cover classes and create a square matrix of land cover transition probabilities (Markov matrix). Each entry of the matrix represents the probability of a given land cover class (rows) converting to another (columns) at a pre-selected year. To analyze the land cover changes within the UCW, we modified 2001, 2006, 2011, and 2016 National Land Cover Datasets (NLCD) for the watershed (Yang et al 2018), narrowing down the number of classes from 15 to 9 (table A1 of appendix A), thereby reducing the potential combination of transitions when projecting future land cover using the LCM (Eastman 2016).

Predictor variables (e.g., distance to roads and distance to urban disturbance) were used to calculate the suitability of a pixel to be converted from one land cover class to another (table 1). The LCM can apply a layer of constraints and incentives (CI layer) to each of the transitions being modeled. We used this layer to simulate the effects of four different public land use policies (scenarios) on the allocation of land cover transitions. The sub-models and their respective driving forces used across all scenarios were the same. Therefore, the differences among scenarios are associated with the Markov matrix (quantity of land use transitions) and the CI layers (allocation of land use transitions). Finally, we identified the laws pertaining to the establishment of buffer areas within the study area and used them to restrict new urban development in our land cover change projections (Appendix A7). All scenarios considered the same population growth over time, but with different land use allocations. More details about projection of land cover change can be found in Appendix A8. All scenarios were projected to the years of 2020, 2025, 2030, 2035, and 2040.

4.2. Future land planning scenarios

4.2.1. Baseline
This scenario represents the future land cover of the UCW if the characteristics of development remain the same as observed between 2001 and 2016. The land cover layers from 2001 and 2011 were used in the LCM (see Appendix A8 for more details). The CI layer of the Baseline is based on the median household income of census tracts (considered constant over time for simplicity, as discounting it would be a linear transformation with no effect on the model, see Appendix A8), and new urban developments are not allowed in protected areas or legal buffer areas.

4.2.2. Urbanization
A higher urban growth rate than the Baseline was simulated to analyze its impact on the future land cover of the UCW. Transitions from natural areas to urban lands were more likely to occur under this assumption. A Markov matrix was created for each pair of land cover layers used in the LCM. We tested all pairs of land cover layers available from NLCD and identified the highest urban growth rate that the UCW experienced within these years, which happened between 2001 and 2006. The Markov matrix (table A8 of appendix A) from this period was used in the Urbanization scenario. This scenario used the same CI layer as the Baseline.
4.2.3. Plan 2040
The ARC Plan 2040 recommends the ideal gross densities (development units per unit area) that areas and places described in the plan should have by 2040 in the AMA (Atlanta Regional Commission 2011). Using a map with all these locations mentioned in the plan, we assigned them with their respective densities to create a scenario representing the region if those guidelines were applied. Finally, we used those densities as the base for the CI layer and added the protected areas from PADUS and the same buffer areas as the Baseline to create the final CI layer. This scenario used the same Markov matrix as the Baseline.

4.2.4. Conservation
This scenario represents a conservation-based land use policy that increases the width of buffer areas and reforests them over time. We assumed that the buffer areas of main rivers would be increased to 100 m and secondary rivers to 30 m in the very first simulated year (2020) (Dosskey et al 2010, Pärn et al 2012). The rationale for this increase in buffer areas is that they protect water bodies from nonpoint pollutants and improve the water quality of streams once restored. As in other scenarios, the development of new urban areas within buffer areas is not allowed. In addition, all pastures, crops, and barren lands within buffer areas in the Conservation scenario were gradually converted to mixed forest starting in 2025, representing a reforestation effort to protect water bodies (table 2). This scenario used the same Markov matrix as the Baseline and a similar CI layer, except for the larger buffer areas.

4.3. Modeling ecosystem services
We used four InVEST models (Nutrient Delivery Ratio (NDR), Sediment Delivery Ratio (SDR), Carbon Storage, and Habitat Quality) to quantify and map the provision of three ESs (water quality, carbon storage, and wildlife habitat quality) (Sharp et al 2016). Some input data are common across models. The NLCD layers were retrieved at a spatial resolution of 30 m (Fry et al 2011), and modified versions with nine land cover classes with the same spatial resolution were used in all models (table A1 of appendix A). Watershed boundaries data were downloaded from the USGS and used in the SDR and NDR models (United States Geological Survey 2018). These two models also require a digital elevation model (DEM) dataset, which was retrieved at a 10 m spatial resolution from the USGS as well (United States Geological Survey 2019). Finally, precipitation data at a 4 km spatial resolution from the PRISM Climate Group were used in the NDR model (PRISM Climate Group 2019). In addition to these common data, each InVEST model requires some specific input data, which can be found in table 3 with their respective sources. Particularities of InVEST models used in this study are discussed in appendix A9.

### Table 2. Width (in meters) of the buffer area on each side of the rivers that were converted to mixed forest over time.

| River class | 2020 | 2025 | 2030 | 2035 | 2040 |
|-------------|------|------|------|------|------|
| Main rivers | 0    | 25   | 50   | 75   | 100  |
| Secondary rivers | 0    | 0    | 15   | 15   | 30   |

### Table 3. Input data used in each InVEST model alongside with their source.

| Model                  | Data                                                                 | Source                                                                 |
|------------------------|----------------------------------------------------------------------|------------------------------------------------------------------------|
| Sediment Delivery Ratio| Rainfall erosivity index (R)                                         | (PRISM Climate Group 2018)                                             |
|                        | Soil erodibility (K)                                                 | SSURGO (Soil Survey Staff 2018)                                       |
|                        | Biophysical parameters                                               | (Hancock et al 2014, Cooperative Extension/University of Georgia 2015, 2017, Lee et al 2017) |
| Nutrient Delivery Ratio| Carbon pool for each land cover class                               | (Brown et al 1999, The Intergovernmental Panel on Climate Change (IPCC 2006, Olson et al 2006, Heath et al 2011, Pan et al 2011, Timilsina et al 2013, Jerath et al 2016) |
| Carbon Storage         | Conservation lands (protection against threats)                      | Protected Areas Database of the United States (PADUS) (United States Geological Survey 2020) |
| Habitat Quality        | Threats to wildlife                                                  | (He et al 2017, Li et al 2018, Sun et al 2018).                         |
|                        | Wildlife species                                                     | (National Park Service 2018)                                           |
4.4. Economic values of ecosystem services

We used different techniques described below to assign economic values to ESs. Since we projected the land cover of the years 2020, 2025, 2030, 2035, and 2040, we considered that the years in between had a linear change from one projected year to another for valuation purposes. We calculated the annual and aggregated valuations, and therefore, urban pixels were assigned with zero economic value (Moore et al. 2011).

4.4.1. Carbon storage

The social cost of carbon (SCC), which is the economic damage caused by carbon emissions (Greenstone et al. 2013), was used to estimate the economic value of carbon stored within the UCW. We used an SCC value of $7.26/ton C/year (2020$), which is estimated based on national (USA) climate change damages (United States Government Accountability Office 2020). We did not incorporate population density into this calculation because damages caused by carbon emissions impact a much larger area than the UCW (Groot et al. 2010), and the public nature of storing carbon is already accounted for in the SCC calculation (United States Government Accountability Office 2020).

4.4.2. Wildlife habitat

The value transfer technique was used to estimate economic values associated with wildlife habitat. Natural ecosystems provide support to various species and are crucial to sustaining biodiversity worldwide, which is another large-scale public good (Groot et al. 2010). Therefore, we did not incorporate population density into these estimates. Moore et al. (2011) estimated the economic values of wildlife habitat in Georgia, and we adopted their values to our study. We used a maximum value of $737.66/ha/year associated with wildlife habitat. We multiplied this value by the output of the Habitat Quality model. Since this output ranges from 0 to 1, the highest economic values were assigned to areas with the highest wildlife habitat quality.

4.4.3. Water quality

First, we combined all InVEST outputs associated with water quality (SDR and NDR models) to create one layer representing the contribution of each pixel of the watershed to water quality (see details of water quality valuation in Appendix A10). Then, we calculated the population density and the distance to surface water intakes and to water bodies within the UCW. We combined all these layers and normalized the output to range from 0 to 1. We identified economic values associated with water quality in Georgia using the value transfer technique and estimated it in $7,755.00/ha/year (2020$) (Liu et al. 2010, Moore et al. 2011, 2013). Finally, we multiplied this value to the normalized combined layers, where the highest economic values associated with water quality are found in areas with the highest contribution to water quality, nearest to surface water intakes with the highest water withdrawal, nearest to water bodies, and located in areas with highest population density.

4.5. Integrated indexes of ecosystem services

In addition to the economic values, we used two integrated indexes to analyze the overall impact of different land use policies on the provision of selected ESs. First, we used equation (1) to normalize all biophysical units of ESs to a range from 0 to 1, making their units comparable. In equation (1), ES represents the normalized provision of each ES $i \in I$, $x_i$ is the provision of this ES in its original unit, $\min(x)$ and $\max(x)$ are the overall minima and maxima provision of ES in its original unit. Second, we used unequal weights for each ES (Carbon storage: 0.263, wildlife habitat quality: 0.312, and water quality: 0.425) adapted from Sun et al. (2018). The weights were derived using the software yaahp, based on the AHP, which is a methodology for multi-criteria decision making that accounts for people’s preferences through pairwise comparisons (Durbach et al. 2014). We combined all the weighted and normalized provision of ESs to create a final integrated index ranging from 0 to 1, where higher values indicate a greater overall provision of selected ESs (equation (2)). Where $ESI_x$ denotes the ES integrated index of each pixel $x \in X$, $N_a$ and $P_x$, denote the normalized levels of sediments, nitrogen, and phosphorus exports of each pixel $x \in X$ (the values of these three components were inverted, so smaller exports levels are assigned with higher values in equation (2)), and $C_x$ and $H_x$ denote normalized levels of carbon storage and wildlife habitat of each pixel $x \in X$. Finally, we also calculated the ESs integrated index using equal weights and considered exports of sediments and nutrients as one ES (water quality) (equation (3)).

\[
ES{i} = \frac{x_i - \min(x)}{\max(x) - \min(x)} \tag{1}
\]

\[
ESI_x = 0.263 \times C_x + 0.312 \times H_x + 0.139 \times N_a + 0.124 \times P_x + 0.161 \times S_x \tag{2}
\]
rejecting a null hypothesis with an addition of 35,565 ha of urban areas compared to 2016 that projected higher urban growth rates, resulting in total urban coverage of 42.9% of the watershed by 2040, the Baseline, Plan 2040, and Conservation scenarios had the same urban growth.

5. Results

5.1. Future land cover
The Baseline, Plan 2040, and Conservation scenarios had the same urban growth (16.1%) from 2016 to 2040, reaching an urban coverage of 39.7% of the UCW. The Urbanization scenario used a different Markov matrix that projected higher urban growth rates, resulting in total urban coverage of 42.9% of the watershed by 2040, with an addition of 35,565 ha of urban areas compared to 2016 (table 4). These new urban areas were mainly located in the southwest corner of Lake Lanier (figure 2).

5.2. Provision of ecosystem services and their economic values under different scenarios
In terms of the significance of the difference in the provision of ESs, similar results were found on sediment and nutrients exports. There were no significant differences among scenarios in sediment and nutrients exports in the first year of our projection, 2020. However, once the simulated reforestation program started in 2025, the Conservation scenario exported significantly less sediments (15.87 t/ha) and nutrients (nitrogen: 1.69 kg ha⁻¹, phosphorus: 0.287 kg ha⁻¹) compared to the Baseline (Sediments: 16.54 t/ha; nitrogen: 1.98 kg ha⁻¹, phosphorus: 0.324 kg ha⁻¹; figures 3(a)-(c)). This same pattern of the statistic was observed in all other years, including 2040. The Urbanization and Plan 2040 scenarios were not significantly different than the Baseline at any year. However, in terms of sediment exports, these three scenarios experienced significantly higher exports in 2040 than in 2016, while the Conservation scenario projected a significant decrease in sediment exports in the same period.

The results for carbon storage (figure 3(d)) and wildlife habitat quality (figure 3(e)) were similar. The Urbanization scenario projected significantly lower carbon storage (106.86 t/ha) and habitat quality (0.552) in the very first year of the analysis (2020) compared to the Baseline (108.76 t/ha of carbon and 0.567 in the habitat quality index), and in all other years until 2040. The Conservation scenario projected significantly higher carbon storage. Therefore, with respect to the provision of ESs, the Baseline scenario at each year, while the second test compared the outcome of each scenario at the end of the study. However, in terms of sediment exports, these three scenarios experienced significantly different than the Baseline at each year with the mean of each of the three other scenarios in that same year. In addition, we compared the mean provision of each ES in 2016 with the mean in 2040 for all four scenarios. We used two-sample weighted Student t-tests to conduct pairwise comparisons of the mean provision of each ES by the Baseline scenario at each year with the mean of each of the three other scenarios in that same period.

\[ ESI_x = 0.333 \times C_x + 0.333 \times H_x + 0.111 \times N_x + 0.111 \times P_x + 0.111 \times S_x \] 

(3)

4.6. Statistical analysis
We used two-sample weighted Student t-tests to conduct pairwise comparisons of the mean provision of each ES by the Baseline scenario at each year with the mean of each of the three other scenarios in that same year. In addition, we compared the mean provision of each ES in 2016 with the mean in 2040 for all four scenarios. Therefore, with respect to the provision of ESs, the first analysis compared the performance of all scenarios to the Baseline at a given year, while the second test compared the outcome of each scenario at the end of the study timeline to the starting point of the study area (2016). Moreover, we used two-sample weighted Student t-tests to compare the significance of the difference in the overall provision of ESs when using the AHP methodology and the equal weights. Due to problems associated with using multiple t-tests, i.e., finding false positives or wrongly rejecting a null hypothesis (Zaykin et al 2002), we used the Benjamini and Hochberg (BH) test (Benjamini and Hochberg 1995) to adjust the p-values of all hypotheses tested (see details in Appendix A11). We used a significance level of 5% (after adjusting with the BH test) in all analyses, although we provide tables displaying significance levels of 5% and 1% for each ES in the Appendix (tables A2 to A6 of appendix A). We used the software R 3.6.3 for undertaking statistical analyses. As for the economic values of ESs, we only compared their absolute values and reported their differences without further statistical tests.

Table 4. Land cover class distribution (ha) in the latest year that NLCD provides data (2016), the first projected year (2020), and the latest year (2040) of our simulation.

| Modified          | 2016 | 2020 | 2040 |
|-------------------|------|------|------|
|                   | Base/Plan | Urban | Cons. | Base/Plan | Urban | Cons. |
| Open water        | Open water | Open water | Open water | Open water | Open water | Open water |
| Developed: Open space | Developed: Open space | Developed: Open space | Developed: Open space | Developed: Open space | Developed: Open space | Developed: Open space |
| Developed: Low, medium, high | Developed: Low, medium, high | Developed: Low, medium, high | Developed: Low, medium, high | Developed: Low, medium, high | Developed: Low, medium, high | Developed: Low, medium, high |
| Barren land       | Barren land | Barren land | Barren land | Barren land | Barren land | Barren land |
| Mixed forest      | Mixed forest | Mixed forest | Mixed forest | Mixed forest | Mixed forest | Mixed forest |
| Evergreen forest  | Evergreen forest | Evergreen forest | Evergreen forest | Evergreen forest | Evergreen forest | Evergreen forest |
| Herb. and shrub   | Herb. and shrub | Herb. and shrub | Herb. and shrub | Herb. and shrub | Herb. and shrub | Herb. and shrub |
| Crops and pasture | Crops and pasture | Crops and pasture | Crops and pasture | Crops and pasture | Crops and pasture | Crops and pasture |
| Wetlands          | Wetlands | Wetlands | Wetlands | Wetlands | Wetlands | Wetlands |

|    | 2016 | 2020 | 2040 |
|----|------|------|------|
| Water | 1,165 | 1,165 | 1,165 |
| Developed: Open space | 65,362 | 66,532 | 66,532 |
| Developed: Low, medium, high | 75,229 | 77,908 | 77,908 |
| Barren land | 1,165 | 1,165 | 1,165 |
| Mixed forest | 187,904 | 185,912 | 185,912 |
| Evergreen forest | 21,738 | 21,127 | 21,127 |
| Herb. and shrub | 7,480 | 7,043 | 7,043 |
| Crops and pasture | 36,032 | 35,243 | 35,243 |
| Wetlands | 1,983 | 1,983 | 1,983 |

|    | 2016 | 2020 | 2040 |
|----|------|------|------|
| Water | 1,165 | 1,165 | 1,165 |
| Developed: Open space | 65,362 | 66,532 | 66,532 |
| Developed: Low, medium, high | 75,229 | 77,908 | 77,908 |
| Barren land | 1,165 | 1,165 | 1,165 |
| Mixed forest | 187,904 | 185,912 | 185,912 |
| Evergreen forest | 21,738 | 21,127 | 21,127 |
| Herb. and shrub | 7,480 | 7,043 | 7,043 |
| Crops and pasture | 36,032 | 35,243 | 35,243 |
| Wetlands | 1,983 | 1,983 | 1,983 |
storage and habitat quality than the Baseline in 2035 and 2040. Both ESs were significantly lower in all scenarios in 2040 compared to 2016.

According to the valuation techniques we applied in this study, values associated with water quality were highest compared to carbon storage (SCC) and wildlife habitat quality (Table 5). All values were higher under the Conservation scenario and lower under the Urbanization scenario. The Baseline and Plan 2040 scenarios yielded similar results.

5.3. Comparing the overall provision of ecosystem services
Both AHP and equal weights ESs integrated indexes significantly increased by 2040 under the Conservation scenario but declined in all other scenarios (Figure 4). The ESs integrated index of the Urbanization scenario was significantly lower than the Baseline scenario in all projected years when AHP weights were used. However, when using equal weights, the Urbanization scenario was not significantly different than the Baseline in 2020. In terms of economic values, the Baseline and Plan 2040 scenarios provided similar values over time. The Urbanization and Conservation scenarios provided lower and higher values, respectively than the Baseline scenario in all years. For instance, the total annual value under the Urbanization and Conservation scenarios were $31.96 million (2020$) less and $27.61 million (2020s) more than the Baseline in 2040, respectively.

Figure 2. New urban areas added to the UCW between 2016 and 2040 under the four simulated scenarios: (A) Baseline, (B) Urbanization, (C) Plan 2040, and (D) Conservation.
Overall, the aggregated values of ESs over the entire timeline of the study under the Baseline, Urbanization, Plan 2040, and Conservation scenarios were $19.06, $18.32, $19.06, and $19.44 billion (2020$), respectively.

6. Discussions

6.1. Land cover change in each scenario and public land use policies
All scenarios projected a substantial amount of new urban areas within the UCW by 2040, but our results showed that the larger and protected buffer areas of the Conservation scenario helped mitigating the impacts of
urbanization, indicating that sustainability at the watershed level is only achievable if a regional development plan accounts for conservation in addition to urban growth (World Bank 2018, Atlanta Regional Commission 2019). Moreover, our results revealed a clear trend of urban development on the northeastern portion of AMA, which was consistent with previous projections (Atlanta Regional Commission 2019).

Although there are some public policies in effect that would likely affect the provision of ESs within the UCW in the future, the ARC Plan 2040 was chosen for this study due to its regional importance and guidelines that are easily adapted for spatial analysis. Other policies, such as the Water Resource Management Plan (Metropolitan North Georgia Water Planning District 2017) and the Gainesville/Hall Urban Redevelopment Plan (City of Gainesville and Hall County 2012), focus on actions that are not spatially explicit, such as optimizing the use of water and wastewater treatment facilities. The implementation of these policies alongside the Plan 2040 has a great potential to boost conservation in the region and should be encouraged by policymakers.

Under the Urbanization scenario, new urban areas spread around Lake Lanier, creating a new large urban center on the west side of the lake. Additional protected areas could be placed in this region through the use of optimization programs to maximize the provision of ESs with respect to a budget (Newburn et al. 2005, Remme and Schröter 2016). An alternative to protected areas would be the implementation of larger and well-maintained riparian buffer areas, which was explored in the Conservation scenario. These new buffer areas are important to protect environmental sensitive zones (Dosskey et al. 2010, Pärn et al. 2012). However, state waters without a clear line of wrested vegetation (e.g., freshwater wetlands, and impoundments with vegetated banks) do not require a buffer area in Georgia (The State of Georgia 2011), even though the importance of these areas was already demonstrated in the literature (Tu et al. 2007) and in our results, especially when analyzing the substantial difference in the provision of ESs under the Conservation and other scenarios. Finally, public land use policies in effect within the UCW could address the establishment and maintenance of buffer areas more explicitly.

### 6.2. Impact of urban growth on the provision of ecosystem services

Our projections suggested that water quality within the UCW would decrease mainly because of the establishment of urban patches near water bodies, disrupting natural vegetation, and increasing exports of sediments and nutrients to water streams (Tu et al. 2007). Over time, only the Conservation scenario with its larger and reforested buffer areas prevented the increase of sediment and nutrients exports to water bodies. In fact, the forestation of pastures and agricultural fields located near water bodies, which are the leading nonpoint nutrient exporters within the UCW (Hancock et al. 2014, Cooperative Extension/University of Georgia 2017, Lee et al. 2017), resulted in an increase in water quality (Dosskey et al. 2010) and a potential decrease in water treatment costs (Grolleau and Mccann 2012). Moreover, we estimated that the extra buffer areas added in the
Conservation scenario would provide an additional $256.5 million (2020$) associated with water quality alone. Therefore, although the reforestation program proposed by this scenario would demand long-term investments (2025 to 2040) and new urban development would not be allowed inside buffer areas, the reduced water treatment costs and the overall increase in the provision of ESs might justify such investments. On the other hand, the higher displacement of natural areas simulated in the Urbanization scenario yielded an economic value associated with water quality $547.0 million (2020$) lower than the Baseline, which clearly shows the importance of public land use policies with explicit guidelines for urban growth accounting for the provision of ESs, especially water quality.

Even though the projected population growth is equal in all scenarios, the higher urban growth projected under the Urbanization scenario reduced the carbon storage of the watershed over time compared to the Baseline, demonstrating the negative correlation between urbanization and carbon storage (Eigenbrod et al 2011, Li et et al 2016). We estimated an aggregated loss of $88.2 million (2020$) from 2016 to 2040 due to damages caused by carbon emissions under these circumstances. An alternative for addressing this issue is to incentivize better management practices in the remaining forestlands of the UCW to maximize their carbon storage capacity, such as prioritizing species that fix more carbon (Hernández et al 2017) or selecting an optimal rotation age for forest plantations (Abdi et al 2018). The reforestation program simulated in the Conservation scenario added an extra $51.6 million (2020$), compared to the Baseline, due to the avoidance of carbon emissions caused by the larger and reforested buffer areas. Although this hypothetical program would imply costs, there are some alternatives to facilitate its implementation, such as payment for ESs programs (Derissen and Latacz-lohmann 2013) or a mix of payment for ESs and protected areas (Robalino et al 2015).

The negative correlation between urban areas and wildlife habitat quality reported in previous studies (He et al 2017, Sun et al 2018) was observed in our analysis as well, especially when additional urban growth was simulated in the Urbanization scenario, where a loss of $101.8 million (2020$) was estimated compared to the Baseline over the timeline of this study. Although the projected overall habitat quality under the Baseline and Plan 2040 scenarios were not significantly different over time, the higher urban growth rates in heavily urbanized areas projected by the Plan 2040 scenario induced lower habitat quality in those areas but allowed for better habitat quality in less urbanized counties, especially the ones located north of Lake Lanier. Although the larger and reforested buffer areas projected by the Conservation scenario were not enough to prevent a decline in wildlife habitat quality over the timeline of this study, the aggregated value associated with these ESs was $66.0 million (2020$) more than the Baseline scenario, demonstrating the importance of those sensitive areas for providing wildlife habitat (Zhang et al 2005). Even though we did not model habitat connectivity, those buffer areas serve as corridors of habitat connecting protected areas, which possibly guarantee the flow of species through the landscape and could boost the economic values associated with biodiversity and habitat quality (Lanzas et al 2019).

Please, note that several studies have evaluated the economic value of ESs provided by urban forests in the United States. For example, Nowak et al (2002) reported that the compensatory value of trees in urban areas of lower 48 states in the United States was $2.4 trillion based on the method developed by the Council of Urban and Landscape Appraisers. Majumdar et al (2011) estimated willingness to pay for the benefits provided by urban trees to the tourists in Savannah, Georgia. They found that tourists are willing to pay between a minimum of $81 million to a maximum of $167 million for the benefits provided by urban trees. McPherson et al (2017) reported that the annual value of ecosystem services provided by trees in urban areas of California was $8.3 billion and the urban forests asset value was $181 billion. For estimating economic values, value transfer techniques were used. It was also found that urban forests provided $2.52 in benefit relative to every dollar spent on upkeep and management. A closer look into the existing studies clearly reveals that the current studies do not evaluate the changes in ESs at the watershed level over space and time relative to expected changes in land use under various policy scenarios. In this context, our study provides a fresh approach for policymakers for evaluating the economic and environmental tradeoffs in light of various land use options for an effective decision making.

6.3. Using different indexes to analyze the overall provision of ecosystem services

Although the AHP weights used in this study can be subjective, previous studies showed that it is more robust than assigning random or equal weights to ESs (Durbach, Lahdelma, and Salmiinen 2014, Sun et al 2018). The ESs integrated index using equal weights tested in this study yielded very similar results than the index using the AHP weights, even though less significant results were found when using equal weights to compare the Baseline and the Urbanization scenarios. The results of both indexes indicated that the Conservation scenario would provide significantly higher overall provision of ESs, but these indexes do not show the magnitude of these differences. Including the economic values of ESs into decision-making processes is important to demonstrate economic tradeoffs associated with ESs due to different public land use policies. In addition, incorporating these values could potentially prevent the displacement of natural ecosystems in sensitive areas by allowing for
arguments to retain undeveloped lands, leading to a more sustainable watershed (Tammi, Mustajärvi, and Rasinmäki 2017).

The overall results of the provision of ESs under the Baseline and Plan 2040 scenarios were similar across all years of the analysis, with no significant differences found between them in any analysis, except for their economic values, which were very similar as well. For instance, the difference in their total aggregated values was only $0.57 million (2020s), which represents less than 0.01%. These two scenarios used the same Markov matrix, and therefore, the number of land cover classes was the same in both, with differences only on the spatial allocation of these classes. Even though the importance of land cover in ESs modeling is well known in the literature (Bagstad et al. 2018, Benez-Secanho and Dwivedi 2019), we showed that changes in the land cover (e.g., reforestation of buffer areas and additional urban areas) and the use of different Markov matrices are important to create a broader range of scenarios when analyzing the impacts of public land use policies on the provision of ESs. Nevertheless, models such as InVEST can be used to map and reveal environmental information that could potentially be incorporated into regional development plans (Grêt-Regamey et al. 2017).

6.4. Uncertainties and limitations

Even though we used the best possible driving factors that we could find for each sub-model when projecting future land cover, some limitations are inherent to this technique, such as understanding of underlying processes (e.g., changes in policies and environmental conditions over time) and data limits and constraints (e.g., spatial and temporal resolutions and restricted access to primary data, Committee on Needs and Research Requirements for Land Change Modeling 2014). Overall, the uncertainty of the outputs increases with an increase in the projection period. We made several projections (one every five years) to analyze the consistency of the results over time. The longest projection we have made was for 24 years into the future (2016 to 2040), which is consistent with the literature (Kumar et al. 2016, Sun et al. 2018). Also, we focused on the impact of land cover changes on the provision of ESs over time, but we acknowledge that other policies would affect the provision of ESs as well, such as regulations on the use of fertilizers in agricultural fields, emissions by different sectors of industry, and water consumption (Blackman et al. 2018). Future similar studies could include other transitions that we did not account for, such as land cover changes due to energy development (Trainor et al. 2016) if data are available. This study included models quantifying and mapping several biophysical mechanisms, such as carbon storage and sediment exports, and all of them are subject to uncertainties, even though this methodology is widely used in the literature (Butsic et al. 2017, Sharps et al. 2017, Neugarten et al. 2018). To address these uncertainties, we have used the most plausible parameters as inputs to the developed models to ensure that the provisions of ESs yielded from these models are the best mean estimates. We acknowledge that our economic valuation would have been more accurate if we had conducted primary valuation techniques (Champ et al. 2017). In addition, the economic values of ESs used in this analysis are an underestimation of the total values associated with ESs, which would include other services, such as recreation. We only used one discount rate for simplicity, but other rates could be used in future research. Finally, we used the state’s official population growth rate from the Governor’s Office of Planning and Budget. We acknowledge that a different population growth rate would change the absolute numbers of our results, but the trends and conclusions presented in this study would most likely remain the same.

7. Conclusion

A high urban growth rate is expected in the UCW in the future. The approach policymakers will take to address this growth will directly impact the environment. We simulated the impact of different public land use policies on the provision of ESs within the UCW. A Baseline scenario was set to represent business as usual for future urban growth, and three other scenarios were created with different assumptions. Better water quality with higher economic values could be obtained through the implementation of larger and reforested buffer areas and the enforcement of the existing ones, as projected by the Conservation scenario. In fact, projections of carbon storage and wildlife habitat quality were overall better under this scenario as well, yielding more economic values. In case the UCW experiments a higher urban growth, as simulated in the Urbanization scenario, a loss of almost three-quarters of a billion dollars associated with ESs would happen over the timeline of the study compared to the Baseline. Both the integrated indexes and economic values indicated that the Baseline and Plan 2040 scenarios would experience an overall decline in ESs by 2040, but the differences between these scenarios were minimal, indicating that the spatial pattern of development that is happening within the UCW is already following the Plan 2040 guidelines.

We provided various potential alternatives that could be implemented to facilitate the establishment of larger and forested buffer areas. For instance, the Plan 2040 Development Guide suggests implementation priorities for a variety of sectors, which includes the environment. It could reinforce the importance of buffer
areas and proper allocation of urban development considering the provision of ESs. Moreover, the present study can assist decision-makers when developing land use policies. It can serve as a starting point for future studies investigating other policies that affect the provision of ESs within the UCW. We suggest that, when data are available, the economic values of ESs should be used in decision-making processes, making the understanding of the impact of public land use policies on the provision of ESs more straightforward. However, integrated indexes, such as the AHP methodology, are easier to interpret compared to multiple indicators and could be used to indicate tradeoffs of selecting different scenarios when data are not available.

**Data availability statement**

The data that support the findings of this study are openly available at the following URL/DOI: https://www.mrlc.gov/.

**Conflict of interest**

The authors declare no conflict of interest.

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