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Impacts of Land Ownership and Forest Fragmentation on Water-Related Ecosystem Services Provision, Dynamics and Their Economic Valuation in Kentucky

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Abstract: Ecosystem services assessment is vital for sustainable land management decision-making. However, ecosystem service responses to land ownership and forest fragmentation have rarely been incorporated into landscape management decision-making contexts. Such knowledge gaps pose a challenging conservation issue: how to incentivize landowners to ensure the sustainability of ecosystem services provision? This study provides new insights into integrating ecosystem services into landscape planning by illustrating the significant changes in ecosystem service value among different landowner types. The net ecological and economic consequences of forest land cover transition in Kentucky, USA, were assessed, as were the details of how each landowner type was affected, and the driving factors were analyzed. The results showed that the total value of water-related ecosystem services was USD 745.83 million in 2011, which had decreased by USD 19.38 million compared to the value in 2001. Forestland owned by family landowners contributed 75% of the total loss. Public landowners lost USD 0.08 million, corporate landowners lost USD 0.19 million and family landowners lost USD 0.55 million in terms of water retention value. In terms of nitrogen retention value, there was a loss of USD 1.57 million, USD 7.65 million and USD 1.69 million for public, family and corporate landowners, respectively. Family-owned forestland presented the highest mean value of water retention and the lowest mean value of soil, nitrogen and phosphorus retention. All landowners experienced a noticeable loss in water-related ecosystem services value. Land ownership and forest fragmentation exerted significant impacts on ecosystem services provision and change. Integrating land ownership into ecosystem service assessment may improve the landscape and regional planning, through which scientifically sound decision-making can be promoted by natural resource management agencies.

Keywords: ecosystem service; value; landowner; forest; landscape fragmentation

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Citation: Bai, Y.; Yang, J.; Ochuodho, T.O.; Thapa, B. Impacts of Land Ownership and Forest Fragmentation on Water-Related Ecosystem Services Provision, Dynamics and Their Economic Valuation in Kentucky. *Land* **2024**, *13*, 984. https://doi.org/10.3390/land13070984

Academic Editor: Shicheng Li

Received: 2 May 2024
Revised: 20 June 2024
Accepted: 26 June 2024
Published: 4 July 2024

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1. Introduction

Ecosystem services are the benefits that human communities obtain either directly or indirectly from ecological systems to sustain and satisfy human life [1–3]. Though the importance of ecosystem services to sustainability is recognized by scientists and policymakers, the economic valuation of ecosystem services is still controversial [4]. Some ecosystem services (such as food, timber and fuel) have quantifiable economic values and can be traded in market economies [5]. On the other hand, other services (such as water purification, sediment retention and biodiversity maintenance) are not well recognized in the current market economies and therefore offer little or no financial incentives to landowners to ensure their sustainable provision [5]. According to the Millennium Ecosystem Assessment and other studies, many types of ecosystem services are currently on the decline, and the trend may be accelerated in the future due to land transition or degradation [1,2].

Among various ecosystems, forests provide a remarkably wide range of ecosystem services that are indispensable to society [1]. However, stresses on the world's forests are

increasing. Timber harvest, urban expansion and climate change all play important roles in changing the size and configuration of forests in the landscape, ultimately influencing the continuous provision of future ecosystem services [3]. To restore and maintain forest ecosystem services, scientists researching ecosystem services and policymakers have worked closely to implement optimal forest management strategies [6]. These efforts include identifying critical areas of high-value or fragile forests for targeted conservation and restoration [7], revealing the mechanism by which forest fragmentation impacts the provision of multiple ecosystem services [8,9], and using scenario-based simulations of forest management plans to achieve better tradeoffs of multiple ecosystem services [10]. However, effectively maintaining forest ecosystem services requires special attention to land ownership and their management objectives, practices and goals [6]; unfortunately, this requirement has been largely overlooked in previous studies. From the perspective of private forest landowners, many of the ecosystem services provided by forests are public goods and have no direct market value; thus, the services are not fully considered in land use practices and decisions [3].

In the USA, more than 50% of all forestland is privately owned. Landowners' behaviors (especially those of private landowners) greatly impact the sustainability of forested landscapes across the country [11]. To encourage sustainable forest management, government and organizations have established a wide range of policies and programs, such as forest certification, cost-share, conservation easements, etc. However, most of these programs have addressed less than 10% of forest landowners to date [11]. Behavioral and cognitive factors such as landowner objectives and attitudes are known to have strong influences on management choices and program participation [12]. Many landowners have an aspiration of managing ecosystem services for their benefit and the benefit of society [6]. However, given the fact that many ecosystem services are nonmarketable, many private landowners are not actively managing their forests for these services because of potentially increased management costs, lower timber revenues and a lack of financial compensation [13–16].

One of the most challenging issues regarding forestland ecosystem services is how to create incentives for private landowners to participate in conservation efforts that prevent the loss of forestland to development [5]. Forestlands are rapidly being fragmented and lost to development because economic incentives that should enable landowners to protect these landscapes are either financially inadequate or nonexistent [17–19]. The situation is becoming worse as natural resource management agencies are experiencing significant budget reductions, resulting in cutbacks in spending for the protection of ecosystem services. In the US, these reductions are due not only to the ballooning federal budget deficit but also to the expanding scope (multiple objectives) and complexity of management problems [5,20]. For instance, federal wildfire suppression costs in the US have spiked from an annual average of about USD 425 million from 1985 to 1999 to USD 1.6 billion from 2000 to 2019 [21]. Such high expenditures on fire suppression and limited funding mean less funding for other forest conservation and management activities. Most policy decisions are based on economic tradeoffs as they are more direct and easier to understand for a broader audience. Thus, a better understanding of the net ecological and economic consequences of forest transition, in terms of ecosystem services gain and loss, is critical. The estimation of such gain and loss will aid landowners in appreciating and relating their course of action with consequences in terms of the value of the provided ecosystem services, which may encourage both landowners and natural resource management agencies to make better tradeoffs regarding forest management.

There have been several efforts to remedy the inadequacy of forest management incentives by estimating the total value of forests in different places and focusing on assessing trade-offs among different bundles of ecosystem services, using different valuation methodologies [22,23]. However, such efforts have not incorporated economic gains and losses of specific landowner types. Other researchers investigated whether landowners were willing to protect forests and analyzed the factors (e.g., gender, education, household income)

underlying the landowners' choices and motivation [16,24]. However, those analyses did not explicitly link these factors to changes in specific ecosystem services.

Land ownership produces distinct signatures on landscapes, creating patterns that, in turn, will influence a variety of ecological processes [25]. Ref. [26] found that the likelihood of the forest cover being disturbed was a function of the ownership type [25]. Different forestland ownership types (e.g., private, communal or public) can result in differing forest use [27]. Privately held forests exhibited much higher disturbance rates than state forests and national parks [27]. Land ownership types play an important role in shaping the landscape structure and the capacity of forest landscapes to provide ecosystem services [6]. Forest fragmentation has been increasing in the USA over recent decades and has had a significant negative influence on the provision of forest ecosystem services [9,16,28]. Landowner groups are motivated by a different set of values and beliefs. Some landowners may value their forestland more as a present-day asset (used to generate timber or other marketable ecosystem services), as an investment that can be sold and developed for a profit at a later date or simply as a quiet retreat [29]. By linking land ownership with ecosystem services, ecosystem service gains and losses can be analyzed by ownership type. Maintaining forestlands and their ecosystem services in the US hinges on both reducing penalties and creating incentives for responsible, long-term forest stewardship.

The main objective of this study is to enhance the understanding of forest ecosystem service characteristics by land ownership type. Such an understanding would be critical in designing incentive programs that would be appropriate for each forestland owner type in an effort for the continued provision of forest ecosystem services. Forest landowner incentive programs could include tax benefits, cost-share assistance and payments, the sale of carbon credits, direct subsidies, recognition, education and awareness. The provision of incentives based on the management objectives of different forest landowners could help them to become better stewards of the forest and subsequently their provision of ecosystem services (Figure 1).

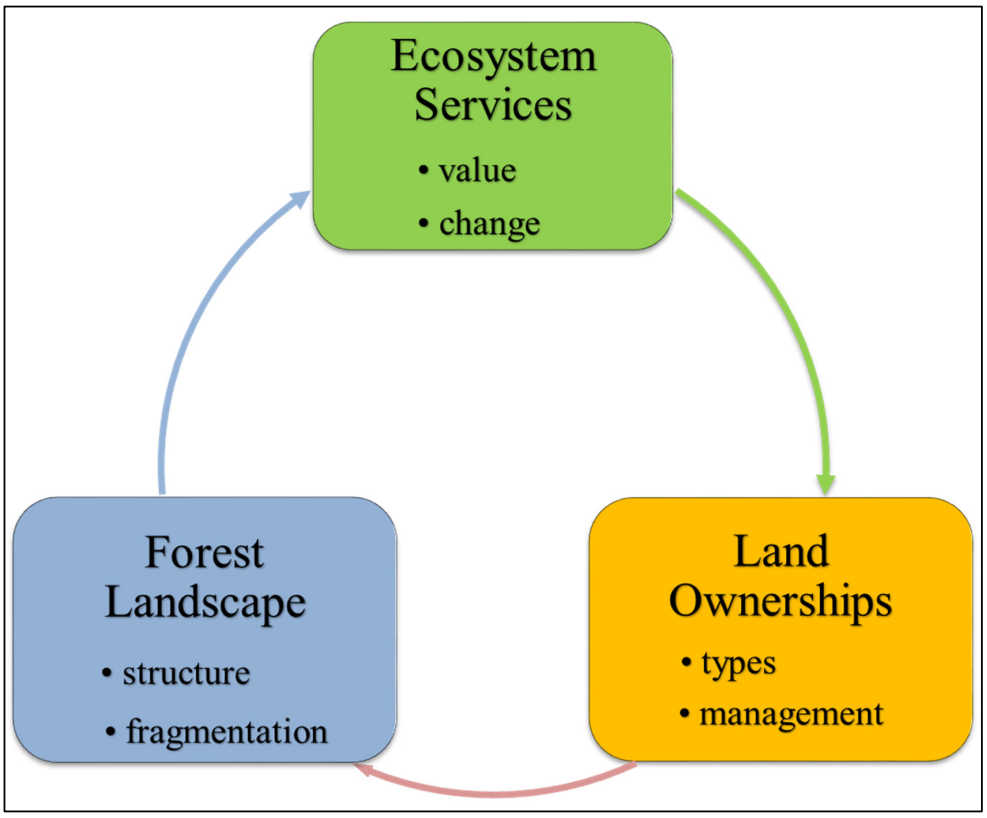


Figure 1. Relationships among landowners, forest landscapes and ecosystem services.

The Commonwealth of Kentucky was selected as a case study because it is composed of diverse forestland ownership types and is experiencing significant forest loss. The three specific objectives of this study were as follows: (1) spatially assess the biophysical quantity of water-related ecosystem services provision and account for the total value of the ecosystem services gained and lost as a result of changes in forestland; (2) explore how different landownership types are affected due to changes in forestland; and (3) identify how forest fragmentation and land ownership types impact the provision of water-related ecosystem services.

2. Materials and Methods

2.1. Study Area

Kentucky (36°30′ N–39°09′ N, 81°58′ W–89°34′ W, Figure 2), covering an area of 104,749 km^2, is located in the east south-central region of the USA and is bounded by the Appalachian Mountains to the east and the Ohio and Mississippi Rivers to the north and the west, respectively. The highest point in Kentucky is 1259 m and the lowest point is 78 m above sea level, the latter of which is located along the Mississippi River. Kentucky has a humid subtropical climate, with an average annual precipitation ranging from 1016 mm to 1397 mm and monthly average temperatures ranging from approximately −1 °C to 27 °C. Kentucky has an estimated population of 4,436,974, with a per capita gross domestic product of USD 42,386 as of 2019 [30]. Kentucky has some of the most diverse woodlands in the USA, with more than 100 tree species naturally occurring in the state [31]. These woodlands support clean water, wildlife habitat and recreation, and they provide the foundation for a large forest industry that makes a USD 13 billion output contribution to the state's economy [32].

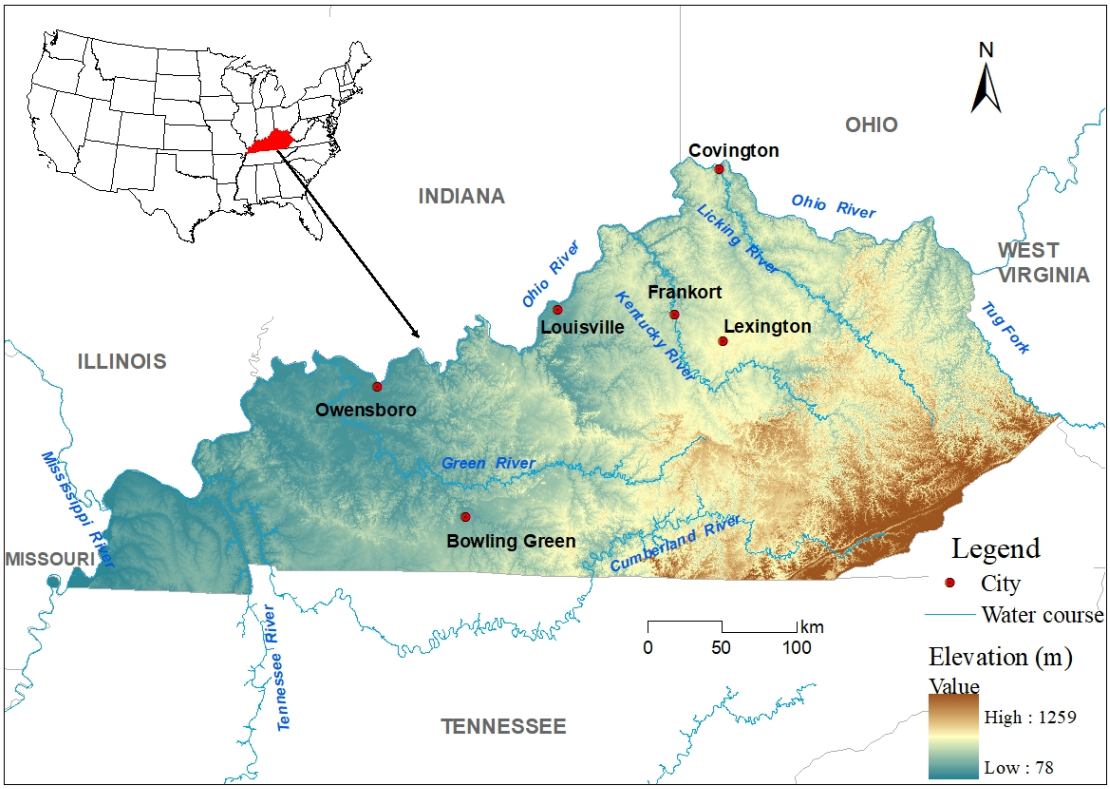


Figure 2. Spatial location of Kentucky and its topographic features.

2.2. Ownership of Kentucky Forestland

Because types of forestland ownership were the focus of this study, only Kentucky's forestland areas were considered; other areas were excluded (designated as "non-forestland"

in the following analysis). "Forestland" refers to forest-dominated parcels, which are composed of various land use and land cover types, including grassland, pasture, water, etc., with a total area of 50,185 km^2 [33]. According to the latest data from the United States Department of Agriculture Forest Service (US Forest Service), the majority (>84%) of forestlands in Kentucky are privately owned. Of these, 84% are considered family-owned and the remaining 16% are corporate-owned (Supplementary Table S1).

Data about landowner types were downloaded from the US Forest Service [33]. The original data contained six types of land ownership: federal, state, local, family, corporate and other private. For better data analysis, these categories were reclassified into three types: public (PU), family (FA) and corporate (CO). Federal, state and local ownership were grouped as PU while family and other types of private ownership were grouped as FA. Supplementary Table S1 presents detailed characteristics of each ownership type. For example, changes in land ownership and management are shown to most likely occur in the family and other private ownership types, as land is passed from one generation to the next or sold. Most corporate entities own land for investment purposes, i.e., farm/ranch and timber products [34]. Figure 3 shows the spatial distribution of each type of forest landowner.

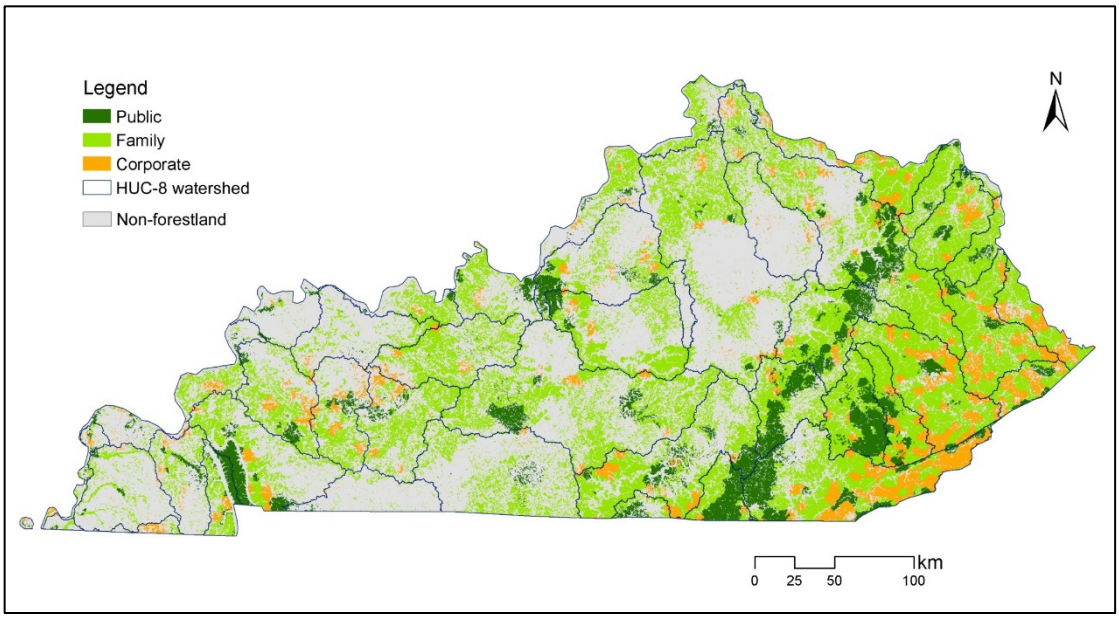


Figure 3. Distribution of forestland by ownership type in Kentucky, USA, and the 8-digit hydrologic unit code watershed (HUC-8).

2.3. Ecosystem Services Selection

Mountainous forest areas provide a wide range of essential ecosystem services to society, most notably through the supply of purified freshwater from upstream headwaters [10,35]. However, land use change has hindered the capacity of mountain areas to regulate the hydrologic cycle and control downstream water quantity and quality [36]. As stated by [35], water flow control and erosion control are the key water-related ecosystem service types in mountain regions, but water purification is another key ecosystem service that is lesser known. Given the importance of water-related ecosystem services in Kentucky, three water-related ecosystem services were selected for this study: water retention, soil retention and water purification.

2.4. Ecosystem Services Evaluation

The InVEST (Version 3.3.3) suite of simulation tools has been developed to enable decision-makers to assess trade-offs among ecosystem services and to compare the consequences of different future change scenarios, such as land use or climate change [37]. This study used InVEST's Water Yield model (for water retention), the Sediment Delivery Ratio

model (for soil retention) and the Nutrient Delivery Ratio model (for nitrogen and phosphorus export) to evaluate the corresponding ecosystem services provided by forestland in Kentucky. Data availability and sources are summarized in Supplementary Table S3. Related input parameters can be found in Supplementary Tables S4–S6. A detailed description of the estimation process can be found in Supplementary Information S1. The InVEST model parameterization and validations are described in Supplementary Information S2.

2.4.1. Water Retention

The service of water retention is defined as the ability of ecosystems to intercept or store water provided by precipitation and is calculated by subtracting runoff and evapotranspiration from precipitation [7]. First, the difference between precipitation and evapotranspiration (i.e., water yield) was estimated using the Water Yield model. Then, water retention was calculated by subtracting runoff from water yield. In InVEST, the annual water yield for each cell (pixel) in a grid of an area is estimated based on average annual precipitation and the Budyko curve [37]. This calculation is then used in conjunction with data on mean annual precipitation and annual reference evapotranspiration, with correction factors for vegetation type, soil depth and plant-available water content [37] (Supplementary Tables S3–S5; Supplementary Information S1.1). Then, an extended model was used to evaluate water retention as the difference between runoff and water yield (Supplementary Information S1.2).

2.4.2. Soil Retention

The InVEST Sediment Delivery Ratio model maps overland sediment generation and delivery to streams. For each cell in a grid, the model first computes the amount of eroded soil and then calculates the sediment delivery ratio, which is the proportion of total eroded soil that reaches a watershed outlet [37]. The amount of annual soil loss in each cell is computed using the Revised Universal Soil Loss Equation. Outputs from the Sediment Delivery Ratio model include the annual sediment load delivered to the stream, as well as the amount of sediment eroded in the watershed and retained by vegetation and topographic features. The input data for the Sediment Delivery Ratio model include maps of land cover and land use, digital elevation models, rainfall and soil erodibility, as well as biophysical attributes related to sediment retention based on land cover (Supplementary Tables S3–S5; Supplementary Information S1.3).

2.4.3. Water Purification

The InVEST Nutrient Delivery Ratio model maps nutrient sources in watersheds and nutrient transport to streams [37]. The model uses a mass balance approach, describing the movement of nutrient mass through space. Sources of nutrients across the landscape, also called nutrient loads, are determined based on the land use map and associated loading rates. Nutrient export from each cell in a grid of a watershed is represented by the product of the load and the nutrient delivery ratio. Each cell's load is modified to account for the local runoff potential, which can be divided into surface runoff and subsurface runoff [37]. Although there are multiple potentially significant impairments in water quality, this study focused on nitrogen (N) and phosphorus (P). The input data for the Nutrient Delivery Ratio model includes maps of land cover and land use, digital elevation models and rainfall, as well as biophysical attributes related to the nutrient loading and retention efficiency for each land use and land cover class (Supplementary Tables S3–S5; Supplementary Information S1.4).

2.5. Ecosystem Services Valuation

Ecosystem services contribute to economic welfare in two ways: through contributions to the generation of income and well-being and through the prevention of damages that inflict costs on society [38]. Economic valuation techniques must be used to value ecosystem services. Supplementary Table S7 presents a review of the most commonly used ecosystem

services valuation methods, together with their norms, advantages and disadvantages. Supplementary Table S8 lists the valuation methods and parameters used to estimate the ecosystem services values in this study.

The value of water-retention services reflects two economic benefits. Firstly, the amount of water retained in ecosystems can be used for vegetation growth during the dry season, thereby avoiding or reducing irrigation costs. Secondly, water that is retained in ecosystems can eliminate or mitigate flooding, thereby avoiding or reducing flood damage costs. Researchers have used a large variety of methods and proxies to estimate the value of water retention services. These techniques include (1) the market price of water supply or electricity (e.g., [39,40]); (2) the economic cost of storing water in constructed reservoirs or dams (e.g., [22,23,41,42]); (3) benefit transfer (e.g., [43–46]); (4) the averted flood damage costs to assess the flood mitigation benefits (e.g., [47–49]).

In this study, the economic cost of storing water in a constructed reservoir was used as a shadow price to estimate the value of water retained in ecosystems because of this estimation technique's relative ease of computation and data availability compared to alternative methods. Information about the construction cost and capacity of Dix Dam, Green River Lake Dam and Taylorsville Lake Dam, all of which are located in the research area, was collected. The costs of the three dams were USD 7 million (Dix Dam, constructed in 1927), USD 33.4 million (Green River Lake Dam, 1938) and USD 103 million (Taylorsville Lake Dam, 1983), and the capacity of the dams were 0.66 billion m^3, 0.89 billion m^3 and 0.36 billion m^3, respectively. A 5% discount rate was applied to reflect inflation and a dam life of 80 years was assumed, based on the average long-term United States Treasury bond rate. Thus, the discounted cost per m^3 of water stored in the three dams was estimated to be USD 0.011, USD 0.023 and USD 0.020, respectively, all in 2018 dollars. The arithmetic mean of these values yielded an estimated value for water retention service of USD 0.018/m^3. This value is comparable with those calculated in other case studies that used the same valuation methods, e.g., USD 0.001/m^3 [23], USD 1.14/m^3 [22] and USD 1.47/m^3 [41].

The value of soil retention also reflects various economic benefits. For example, erosion carries topsoil from fields, reducing the land's productivity and increasing the need for (and cost of) fertilizer; thus, preventing soil erosion also controls fertilizer costs. Soil erosion also increases the potential cost of sediment removal at downstream reservoirs or in rivers and streams. Several approaches have been used to evaluate soil retention services. These techniques include (1) estimating the amount of potential soil nutrients lost due to soil erosion and then using the value of chemical fertilizers needed to replace these lost nutrients (e.g., [42,50]); (2) transferring available information from already completed studies to another location and/or context (e.g., [43–46]); (3) estimating the avoided cost of sediment removal at a downstream reservoir (e.g., [37]); and (4) estimating how losses in soil quality or productivity impact forestland prices (e.g., [23,41]).

In this study, the benefit transfer method from [51] was used; these researchers generated unit values for 14 different categories of soil conservation benefits for all 8-digit hydrologic unit code (HUC) watersheds in the contiguous USA. The travel cost, damage function, replacement cost and aversion of expenditures were used to estimate the values of the 14 soil conservation benefits [51], which (in 2000) ranged from USD 1.46/ton to USD 7.12/ton. Based on these data for the Kentucky study area, a value of USD 2.47/ton was adopted and a 5% discount rate was used to adjust for inflation, leading to a final value (in 2018 prices) of USD 5.94/ton. We chose 2018 as the reference year for valuation as it was the latest common year for all the data variables in this study.

The value of nutrient retention reflects the economic benefits of avoiding water contamination in receiving waters (e.g., downstream rivers and lakes). Several approaches have been used to value the nutrient retention service, including (1) calculating the average price of mixed chemical fertilizers (e.g., [23,42]); (2) transferring available information from studies that were already completed in another location and/or context (e.g., [43–46]); and (3) estimating the removal treatment cost of pollutants (e.g., [23,40]). In the current study, a value of USD 12.69/kg of nutrients was used, based on the treatment costs in a Virginia,

USA, case study [52]. Considering inflation of 5%, the final value (in 2018 prices) used in the current study was USD 15.42/kg of nutrients.

2.6. Landowner and Forest Fragmentation Indicators

2.6.1. Patch Density by Land Ownership

Patch density by land ownership (PDL) represents the degree to which landowner types are fragmented in a given landscape. A higher value of landowner fragmentation indicates a greater number of fragmented landowners. PDL is calculated as follows:

$$PDL = \frac{n_i \cdot n_k}{\sum_j^1 S_{ij}} \quad (1)$$

where n_i is the number of landowner types in a given area i (i.e., a watershed in this study); n_k is the number of land ownership patches in a given watershed i; S_{ij} is the area owned by landowner type j (j = 1, 2, 3. . .N) in watershed i. "Land ownership patches" refer to spatial polygons differentiated by landowner types.

2.6.2. Mean Landownership Patch Size

Mean landowner patch size (MLPS) represents the average size of landowner patches in a given landscape. As the name implies, a higher value of MLPS indicates a larger average size of each landowner patch. MLPS is calculated as follows:

$$MLPS = \frac{\sum_j^1 S_{ij}}{n_k} \quad (2)$$

where n_k is the number of land ownership patches in a given watershed i, and S_{ij} is the area of patches owned by landowner type j (j = 1, 2, 3. . .N) in watershed i.

2.6.3. Patch Density by Forestland Cover

Forest fragmentation, as indicated by the patch density of forestland cover (PDF), represents the degree to which forest area is fragmented in a given landscape. A higher value of PDF indicates a more fragmented forest landscape, and this parameter can be calculated as follows:

$$PDF = \frac{n_f}{S_f} \quad (3)$$

where n_f is the number of forest patches in a given landscape, and S_f is the total area of forest in the landscape.

2.6.4. Mean Forest Patch Size

Mean forest patch size (MFPS) represents the average size of forest patches in a given landscape. A higher value of MPFS indicates a larger average size of forest patches in the landscape. MPFS is calculated as follows:

$$MFPS = \frac{S_f}{n_f} \quad (4)$$

where S_f is the total area of forest in a given landscape, and n_f is the number of forest patches in the landscape.

2.7. Statistical Analysis

Differences in the mean value of each service by each landowner type were tested for statistical significance using the independent samples t-test. Samples were extracted from the original landowner parcels (Figure 3). There were 3918 ownership patches for PU, 24,087 parcels for FA and 3532 parcels for CO.

The relationships between ecosystem services and natural environment factors were tested using Pearson correlation analysis. Landowner types and forest fragmentation factors influenced ecosystem services provision, and the changes were also analyzed through Pearson correlation analysis. A total of 42 samples were extracted using an 8-digit hydrologic unit code (HUC-8) (Figure 3). All statistical analyses were conducted using SPSS statistical software.

2.8. Data Requirement and Preparation

The InVEST model requires multiple gridded datasets combined with specific biophysical data as inputs. Digital elevation model (DEM) data with a spatial resolution of 10 m were downloaded from the Kentucky Geoportal. Land use layers with a spatial resolution of 30 m were downloaded from the National Land Cover Database (NLCD) maintained by the United States Geological Survey. Climate data containing the average annual precipitation and temperature from 1981 to 2016 were downloaded from PRISM Climate Type. Spatial data for the state of Kentucky and other relevant data collected for this study are listed in Supplementary Table S3, which includes summaries of each dataset by source, a short introduction and the associated models. Also, Supplementary Tables S4 and S5 list key parameters used in the InVEST model. All spatial layers were resampled to a 30 m resolution and assigned to the Kentucky State Plane FIPS 1600 reference system.

3. Results

3.1. Land Use Composition and Transition

Forests occupied 81.09% of the forestland in 2011. The forest area decreased from 41,299 km^2 in 2001 to 40,697 km^2 in 2011 (a 1.46% reduction, Table 1). From 2001 to 2011, 98.43% of forestland was retained, but 0.8% was converted into grassland and 0.41% was converted into barren land (Supplementary Table S9).

Table 1. Land use composition in 2001–2011 as a proportion of total forestland area (%) and as absolute area (km^2).

Land Use Types	2001		2011		Change Rate
	Area (km^2)	Percentage	Area (km^2)	Percentage	
Water	188.38	0.38%	186.25	0.37%	−1.13%
Developed	1781.38	3.55%	1815.13	3.62%	1.89%
Barren	59.38	0.12%	242.25	0.48%	308.00%
Forest	41,299.25	82.29%	40,696.75	81.09%	−1.46%
Shrubland	190.25	0.38%	249.00	0.50%	30.88%
Grassland	2241.75	4.47%	2518.13	5.02%	12.33%
Pasture	2976.25	5.93%	3011.38	6.00%	1.18%
Cultivated	813.50	1.62%	830.63	1.66%	2.11%
Wetlands	634.38	1.26%	635.00	1.27%	0.10%

Land use and land cover composition and transition varied among the three ownership types (Supplementary Figure S1). Forests under PU ownership covered an area of 6643 km^2 in 2011, or 84.76% of the total public land area. From 2001 to 2011, 99.08% of the forestland under PU ownership was retained, 0.47% was converted into grassland and 0.2% was converted into barren land (Supplementary Tables S10 and S11). Forests under FA ownership covered an area of 27,500 km^2 in 2011, or 77.44% of the total FA land area. From 2001 to 2011, 98.54% of the forestland under FA ownership was retained, 0.77% was converted into grassland and 0.34% was converted into barren land (Supplementary Tables S12 and S13). Forests under CO ownership covered an area of 5419 km^2 in 2011, or 79.29% of the total CO land area. From 2001 to 2011, 97.05% of the forestland under CO ownership was retained, 1.38% was converted into grassland and 1.12% was converted into barren land (Supplementary Tables S14 and S15).

Generally, forest area was lost under all three types of land ownership. However, forests under CO ownership showed the highest loss (2.66%), the average annual loss rate of which was 14.79 km^2 from 2001 to 2011.

3.2. Ecosystem Services Comparison and Factors

Forestland under FA ownership presented the highest total biophysical amount of water retention, soil retention, N retention and P retention. For example, the total biophysical amount of water retention for all landowners was 8.04 billion m^3 in 2011, of which FA-owned forestland contributed 5.99 billion m^3 (74.5%). In terms of monetary value, FA-owned forestland also had the highest water retention value (USD 119.74 million), followed by PU (USD 22.24 million) and CO (USD 18.82 million). Likewise, the total biophysical amount of soil retention service for all landowners was 22.58 million t in 2011, of which FA-owned forestland contributed 14.41 million t (63.82%) worth USD 85.57 million. FA-owned forestland also retained 15.67 million kg N and 4.12 million kg P (Table 2). The total value of water-related ecosystem services was USD 745.83 million in 2011, of which 47.67% was contributed by N retention and 21.56% was contributed by water retention. The spatial distributions of each ecosystem service value are presented in Figure 4. The relatively high value of water retention was distributed in the central area of Kentucky, while the relatively high values of soil retention, N retention and P retention were mainly distributed in eastern Kentucky.

The ranking of the mean value of each ecosystem service according to each landowner type was largely different from the ranking based on the total value of the ecosystem services. FA-owned forestland still ranked first in terms of water retention, with a mean value of USD 30.4/ha, and was followed, as in the ranking based on total value, by PU-owned forestland (USD 25.5/ha) and CO-owned forestland (USD 24.8/ha). The mean values of water retention according to each landowner type were significantly different from each other (independent samples *t*-test; $p < 0.01$). CO-owned forestland achieved the highest mean soil retention value (USD 37.6/ha), while FA-owned forestland presented the lowest mean value (USD 23.8/ha). The mean values of soil retention for each landowner type were also significantly different ($p < 0.01$). PU-owned forestland achieved the highest mean values for both N retention (USD 97.1/ha) and P retention (USD 27.8/ha). In contrast, FA-owned forestland achieved the lowest mean values for nutrient retention (USD 87.8/ha for N retention and USD 23.1/ha for P retention). The mean values of both N retention and P retention for PU-owned forestland were significantly different from those for FA- and CO-owned forestland ($p < 0.01$; Figure 5).

Table 2. Ecosystem services and their values by landowner types in 2011.

	Water Retention		Soil Retention		Nitrogen Retention		Phosphorus Retention		Aggregated
	Biophysical (10^9 m^3)	Monetary (Million USD)	Biophysical (10^6 t)	Monetary (Million USD)	Biophysical (10^6 kg)	Monetary (Million USD)	Biophysical (10^6 kg)	Monetary (Million USD)	Monetary (Million USD)
PU	1.11 (1.4 ± 1.3)	22.24 (25.5 ± 23.5)	3.75 (4.8 ± 9.7)	22.28 (28.4 ± 57.4)	4.18 (6.3 ± 2.7)	64.53 (97.1 ± 41.6)	1.19 (1.8 ± 1.9)	18.44 (27.8 ± 29.3)	127.49 (178.8 ± 151.8)
FA	5.99 (1.7 ± 1.4)	119.74 (30.4 ± 25.1)	14.41 (4 ± 8.1)	85.57 (23.8 ± 48.2)	15.67 (5.7 ± 3.1)	241.70 (87.8 ± 47.8)	4.12 (1.5 ± 1.9)	63.61 23.1 ± 29.3	510.62 (165.1 ± 150.4)
CO	0.94 (1.4 ± 1.3)	18.82 (24.8 ± 22.6)	4.42 (6.3 ± 12.1)	26.23 (37.6 ± 71.9)	3.20 (5.9 ± 2.8)	49.30 (90.9 ± 43.2)	0.88 (1.6 ± 1.8)	13.37 (24.7 ± 27.7)	107.72 (178.0 ± 165.4)
Total	8.04	160.80	22.58	134.08	23.05	355.53	6.19	95.42	745.83

Note: PU refers to public landowners; FA refers to family landowners; CO refers to corporate landowners. Numbers in brackets represent the mean value and standard deviation for each ecosystem service. The unit for water retention in the bracket is 10^3 m^3/ha; for soil retention, it is t/ha; for both nitrogen and phosphorus retention, it is kg/ha. The units for all monetary values in brackets are USD/ha.

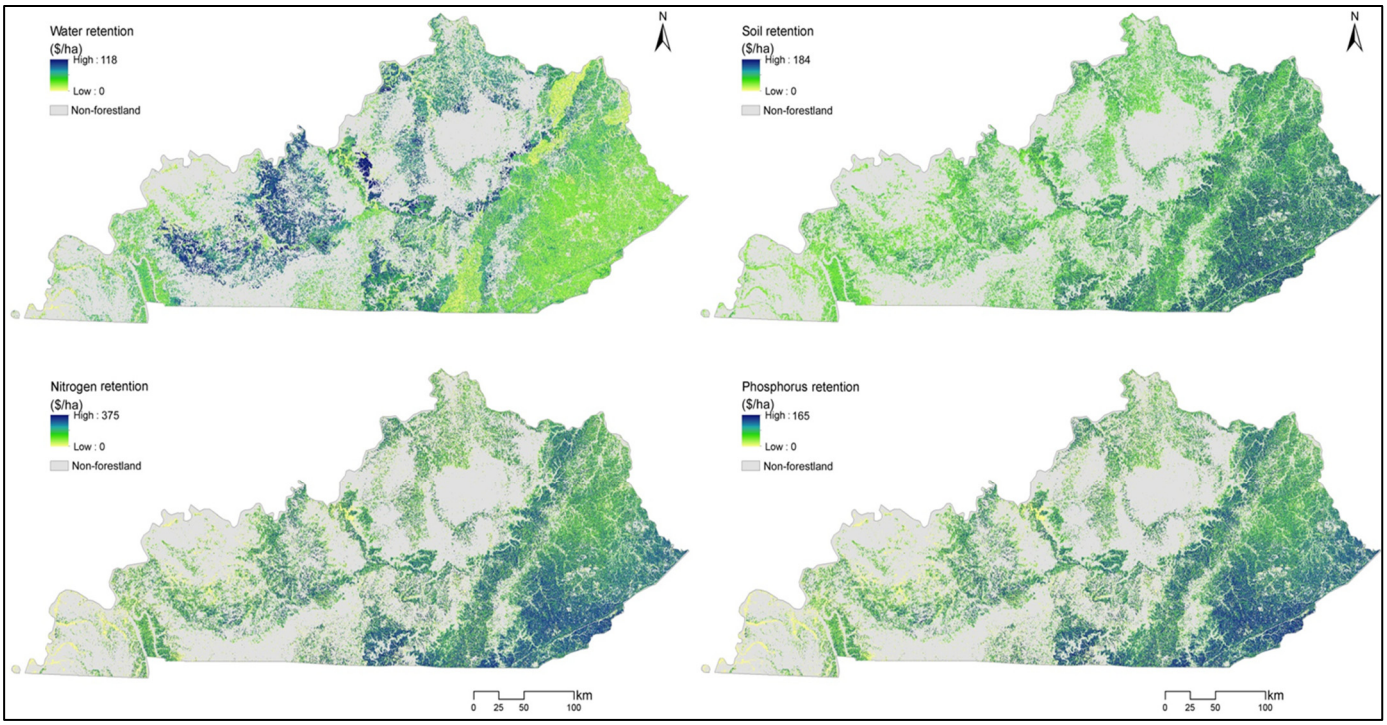


Figure 4. Spatial distribution of ecosystem services values in 2011.

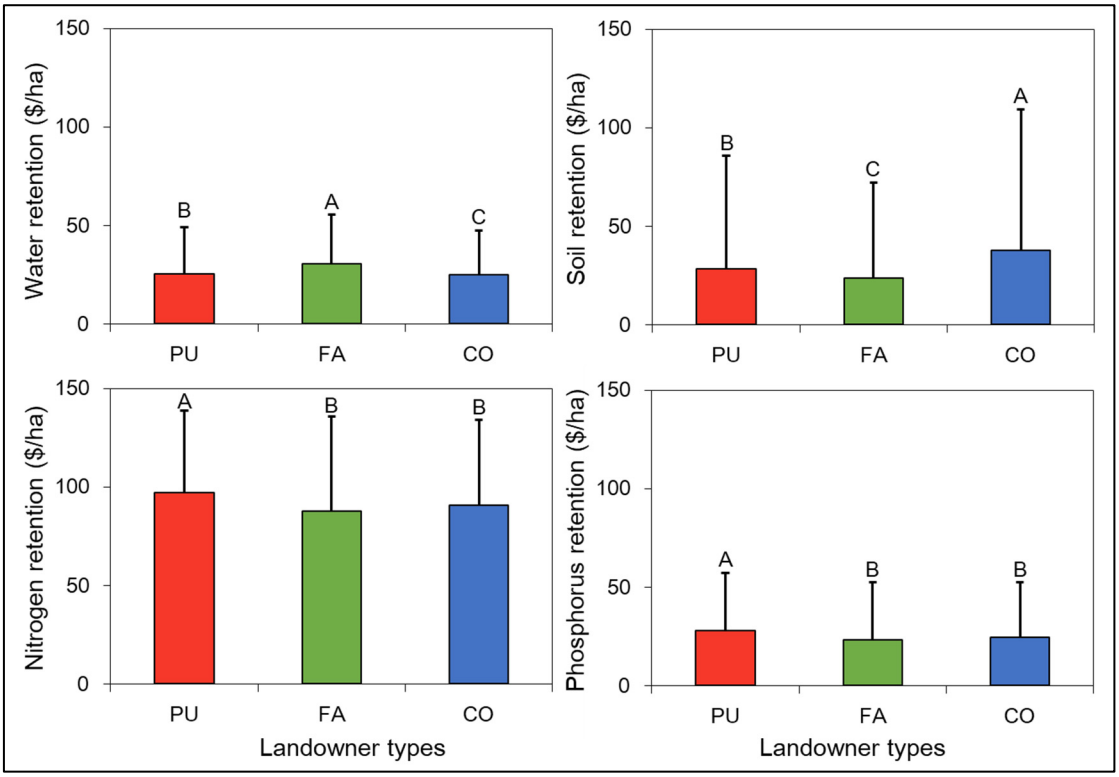


Figure 5. Comparison of the mean value of each ecosystem service (A, B, C) by landowner types. (In the independent samples *t*-test, the number of patches for PU was 918; for FA was 24,087 and for CO was 3532, respectively).

Natural environmental factors imposed diverse impacts on ecosystem services. From Pearson correlation analysis, precipitation presented a significant positive correlation with

water retention (degrees of freedom (df) = 42, $p < 0.05$) but an extremely significant negative correlation with soil retention (df = 42, $p < 0.01$) and significant negative correlations with N retention and P retention (df = 42, $p < 0.05$). Elevation and slope were not significantly correlated with water retention, mainly due to scale effects such that the values used for analysis were averaged. However, elevation and slope presented extremely significant positive correlations with soil retention, N retention and P retention (df = 42, $p < 0.01$) (Table 3).

Table 3. Relationships between ecosystem services and natural environmental factors.

	Precipitation	Elevation	Slope
Water retention	0.331 *	−0.05	−0.089
Soil retention	−0.476 **	0.907 **	0.949 **
Nitrogen retention	−0.328 *	0.74 **	0.757 **
Phosphorus retention	−0.38 *	0.785 **	0.805 **

* Significant level at 0.05 ($p < 0.05$); ** significant level at 0.01 ($p < 0.01$); n = 42 (number of HUC-8 watersheds within Kentucky).

3.3. Ecosystem Service Changes

Land use change induced changes in ecosystem services during the period 2001–2011, which resulted in gains and losses among landowner types. A comparison of the ecosystem service changes from 2001 to 2011 showed that the value of water retention, soil retention, N retention and P retention all decreased under each landowner type and amounted to a total loss of USD 19.38 million. For example, in terms of water retention services value from 2001 to 2011, PU lost USD 0.08 million, FA lost USD 0.55 million and CO lost USD 0.19 million (Table 4). The losses of these services were mainly in eastern Kentucky, where FA-owned forestland is mainly concentrated.

Table 4. Ecosystem services value changes from 2001 to 2011 and their significance by landowner types (million USD).

Landowner Types	Water Retention	Soil Retention	Nitrogen Retention	Phosphorus Retention
PU	−0.08	−0.07	−1.57	−0.15
FA	−0.55	−1.14	−7.65	−5.18
CO	−0.19	−0.38	−1.69	−0.71

Factors underlying the provision, gain and loss of ecosystem services are valuable for land use decision-making. PDL had an extremely significant negative impact on water retention provision (Pearson correlation; df = 42, $p < 0.01$) and a significant positive impact on water retention change (df = 42, $p < 0.05$). Both PDL and PDF had an extremely significant negative impact on soil retention provision (df = 42, $p < 0.01$), but only PDF showed a significant positive impact on soil retention change (df = 42, $p < 0.05$). In addition, both PDL and PDF showed extremely significant negative impacts on N and P retention (df = 42, $p < 0.01$). Both PDL and PDF exhibited significant positive impacts on N and P retention (df = 42, $p < 0.05$) (Table 5). In addition, PDL and PDF also had an extremely significant positive correlation with each other (df = 42, $p < 0.01$) (Supplementary Table S16).

Table 5. Relationships between ecosystem services provision and change as a function of land ownership and forest fragmentation.

Ecosystem Services	Fragmentation	Water Retention	Soil Retention	Nitrogen Retention	Phosphorus Retention
Provision	PDL	−0.478 **	−0.425 **	−0.769 **	−0.757 **
	PDF	−0.259	−0.63 **	−0.788 **	−0.793 **
	MFPS	0.278	0.879 **	0.548 **	0.594 **
	MLPS	0.26	0.832 **	0.512 **	0.558 **
Change	PDL	0.381 *	0.292	0.345 *	0.342 *
	PDF	0.357 *	0.35 *	0.374 *	0.379 *
	MFPS	−0.345 *	−0.584 **	−0.605 **	−0.612 **
	MLPS	−0.268	−0.496 **	−0.555 **	−0.559 **

* significant level at 0.05 ($p < 0.05$); ** significant level at 0.01 ($p < 0.01$); n = 42 (number of HUC-8 watersheds within Kentucky); PDL: patch density by landownership; PDF: patch density by forestland cover; MFPS: mean forest patch size; MLPS: mean landownership patch size.

These results indicated that PDL and PDF have an important impact on ecosystem services provision and change. The more fragmented the forested areas and land ownership, the less ecosystem services provision and the more ecosystem service change. In addition to PDL and PDF, MFPS and MLPS also exerted important impacts on ecosystem services provision and change. Generally, the larger the forest size and size of holdings by a specific landowner type, the greater the ecosystem services provision and the smaller the ecosystem service change (Table 5).

4. Discussion

4.1. Landowners Experience a Major Loss of Ecosystem Service Values

A major goal of the ecosystem services assessment approach is to integrate ecosystem protection into development decisions to increase sustainability [53]. However, the majority of ecosystem service assessments have focused on biophysical or monetary accounting, and their scientific policy suggestions have been theoretical and seldom integrated into a social process (i.e., stakeholders) [4]. The current study assessed the net ecological and economic consequences of forest transition and details about how each landowner type was influenced; the factors underlying the changes were also analyzed. The results suggest that including the value of ecosystem services in landscape management could be a persuasive and practical incentive for prompting both landowners and natural resource management agencies to make better land management tradeoffs. This approach is also a good starting point for linking natural resource considerations with social considerations in the land use decision-making process.

The study showed that precipitation was significantly positively correlated with water retention. FA-owned forestland was mainly distributed in areas that experienced high precipitation, which resulted in FA-owned forestland exhibiting the highest mean value of water retention service. However, FA-owned land also presented the lowest mean values of soil retention and nutrient retention services. The analysis showed that land ownership and forest fragmentation have significant influences on ecosystem services provision and changes. Changes in land ownership and management practices among FA owners are most likely to occur as land is passed from one generation to the next or when some or all of the land is sold or transferred [34]. A generational shift in land ownership (i.e., inheritance, sale) has been shown to usually result in landscape parcelization [54]. The results from the current study demonstrate that FA ownership exhibited the greatest fragmentation among the three landowner types (Supplementary Table S1), and FA-owned forestland represented the lowest proportion of the total landscape area (Supplementary Table S12). These factors resulted in the greatest degree of forest fragmentation among the three landowner types and also meant that FA-owned forestland exhibited the lowest mean values of soil retention and nutrient retention.

FA owners possess the largest amount of area in Kentucky (and in the entire USA). Thus, FA-owned forestland had the highest total amount of water retention, soil retention and nutrient retention. However, from 2001 to 2011, the collective forest area owned by all landowner types decreased by 1.46%, and FA-owned forestland losses represented 63.25% of the total loss. The value of ecosystem services lost due to the decrease in the forested area amounted to USD 0.82 million in terms of water retention, USD 1.59 million in soil retention, USD 10.92 million in N retention and USD 6.05 million in P retention, of which losses from FA-owned forestland represented 67.54%, 71.88%, 70.09% and 85.71%, respectively. Furthermore, FA-owned forestland experienced an extremely significant loss in soil retention, N retention and P retention value. Because the provision of ecosystem services from PU- and CO-owned forestland remained in a relatively good condition, whereas that from FA-owned forestland, as well as the actual area, suffered large losses, the study results suggest that FA-owned forestland should be considered a priority for efforts to promote sustainable forest management and the continued provision of ecosystem services.

4.2. Integrating Ecosystem Services Information into Landscape Planning

Ecosystem management requires cross-jurisdictional problem-solving. Some ecosystem services (such as food production and pollination) are provided in situ, i.e., the services are provided and the benefits are realized in the same location. However, most ecosystem services (such as water conservation and water purification) are directional, omnidirectional or decoupled, i.e., the services are generated in one location, but the benefits are realized in other locations [55]. Thus, the provision and flow of ecosystem services are complex and mixed among landowners. Practices that influence ecosystem services provision in one landowner parcel may influence ecosystem services provision in another landowner parcel. Landowners have their unique management practices that are driven by different social, political and legal factors; these differences in management can drive changes in land use and, ultimately, in land cover and ecosystem services provision across the landscape [56]. FA- and CO-owned land is often fragmented into multiple parcels, resulting in decisions that are made at the parcel scale by diverse landowners [57]. The results from our study highlight the problem of fragmentation of land ownership types, which was shown to have a significant impact on forest fragmentation and ecosystem services provision.

Given the aforementioned relationships, the results from our study support cross-boundary cooperation among landowners as one means by which to achieve multi-scalar management under fragmented ownership in a forested landscape [57]. The benefits of coordinated forest management are well documented in the landscape impact literature and include the protection of ecosystem services [58]. Although many landowners have demonstrated a willingness to consider cross-boundary cooperation [59], successful and efficient cooperation is greatly influenced by incentives such as shared ecosystem service values and shared common management objectives [54,60]. Landowners should be well informed, able to realize the value of the ecosystem services they conserve and recognize the opportunities to access multiple sources of revenue. Ecosystem service valuation promotes greater economic incentives for landowners to practice conservation and, in turn, provides greater ecological benefits because it increases the number of landowners to engage in conservation across a broad landscape [5]. If properly implemented, ecosystem service value information could promote landscape-scale conservation and help allocate necessary financial incentives toward a more holistic approach to protecting ecosystems.

4.3. Summary, Limitations and Future Directions

Most of the existing ecosystem services assessments are general and theoretical, and either do not connect information about specific ecosystem service gains and losses with those responsible for providing the services (e.g., landowners) or do not connect to the providers' decision-making contexts [4]. The current study addressed this specific gap by utilizing information about ecosystem services to inform landowners' activities and to provide the basis for incentives for forest management decision-making. Nevertheless,

there are notable inherent limitations in this study. First, only limited water-related ecosystem services were considered, and land ownership was assumed to be unchanged for the study period (2001–2011). Second, uncertainties and a lack of consensus regarding the techniques employed in ecosystem service valuation exist, even though economic information provides an important basis for public awareness and ecological compensation. Another limitation of the InVEST hydrologic models are their inability to account for seasonal or sub-seasonal variability, groundwater and water resource infrastructure that redistributes water flow [61]. Despite these limitations, this study addressed the fundamental arguments of whether ecosystem services valuation can provide information that could be used as a basis to provide incentives to influence stakeholders' decision-making. However, putting these analytical results into practice requires monitoring and stakeholders' participation. The challenge is to accurately measure ecosystem services from providers and link those ecosystem services with beneficiaries' needs for the same. The next step in fostering effective policy actions is to build actual connections that link forest landscape characteristics with stakeholders' benefits.

5. Conclusions

This study provides new insights about integrating ecosystem service information into landscape planning by illustrating the significant variations in ecosystem services among different forest landowners. Although forestland owned by all landowner types experiences decreases in the value of water retention, soil retention and nutrient retention services, the losses are unevenly distributed among landowner types. For example, the total value of water-related ecosystem services in the study area decreased from 2001 to 2011, but 75% of the total loss occurred in FA-owned forests. Therefore, efforts to promote sustainable forest management should be targeted at specific types of landowners (in this study, FA landowners). Landowner types and forest fragmentation exert significant impacts on ecosystem services provision and change. Thus, cross-boundary cooperation among landowners provides one method for achieving multi-scalar management under fragmented forestland ownership in a forest landscape. Including ecosystem service values in forest landscape management decisions is an evidence-based basis to provide incentives to both landowners and natural resource management agencies to make better forestland management tradeoffs. The use of ecosystem service valuation is also a good starting point for linking natural resource considerations with social considerations in the decision-making process.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/land13070984/s1, Figure S1: Land use and land cover in 2001 (a) and 2011 (b).; Table S1: Characteristics of ownership groups; Table S2: Runoff Coefficients; Table S3: Data requirement for the InVEST model (Water yield model = WY; Nutrient delivery ratio model = NDR; Sediment delivery ratio model = SDR); Table S4: Key parameters used in the current study; Table S5: Critical parameter settings in the biophysical table; Table S6: LULC class definitions from NLCD 2001 and 2011, used in the maps for Kentucky; Table S7: The valuation methods and parameters used to estimate the values in this study; Table S8: Land use transition in the whole forestland parcels; Table S9: Land use composition in 2001–2011 as percent and total area in the Public forestland parcels; Table S10: Land use transition in the public forestland parcels; Table S11: Land use composition in 2001–2011 as percent and total area in the Family forestland parcels; Table S12: Land use transition in the Family forestland parcels; Table S13: Land use composition in 2001–2011 as percent and total area in the Corporate forestland parcels; Table S14: Land use transition in the Corporate forestland parcels; Table S15: Relationships between landowner fragmentation and forest fragmentation; Table S16: Methods and norms used to estimate the value of ecosystem services; S1.1: Water yield (WY) model; S1.2: Extension model for water retention; S1.3: Sediment Delivery Ratio (SDR) model; S1.4: Nutrient Delivery Ratio (NDR) model.

Author Contributions: Conceptualization, T.O.O. and J.Y.; methodology, T.O.O., J.Y. and Y.B.; software, J.Y. and Y.B.; validation, J.Y. and Y.B.; formal analysis, Y.B.; investigation, J.Y. and Y.B.; resources, T.O.O. and J.Y.; data curation, J.Y. and Y.B.; writing—original draft preparation, Y.B. and J.Y.; writing—

review and editing, T.O.O., B.T. and J.Y.; visualization, Y.B.; supervision, T.O.O. and J.Y.; project administration, T.O.O.; funding acquisition, T.O.O. All authors have read and agreed to the published version of the manuscript.

Funding: The McIntire-Stennis Program of the National Institute of Food and Agriculture, U.S. Department of Agriculture, supported this work under Project Number 1018771.

Data Availability Statement: The authors confirm that some of the original data used to support this study that are not already provided as Supplementary Materials may be shared upon request.

Conflicts of Interest: The authors have no relevant financial or non-financial interests to disclose.

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