ORIGINAL RESEARCH

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Impact of increased wood pellet demand on biodiversity in the south-eastern United States

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Abstract

Increasing wood pellet exports from the United States are projected to lead to changes in land use and timberland management, including a shift from natural timberland to pine plantations. These projected changes may impact biodiversity. This study aims to quantify potential biodiversity impacts of increased wood pellet demand in the south-eastern United States in a spatially explicit manner. We determined differences according to an index of potential species richness (for total, threatened and endemic species and four taxonomic groups) between scenarios of high and low demand for wood pellets, while taking into account potential developments in other wood markets and other land uses. Increased demand for wood pellets was projected to cause both positive and negative biodiversity impacts. Negative shifts in total potential species richness were projected for areas in Florida, coastal Virginia and North Carolina, and parts of the Gulf Coast. Positive shifts in total potential species richness were projected in parts of Oklahoma and Arkansas. In some locations, the direction of change differed per taxonomic group, highlighting the importance of analysing different taxonomic groups. Shifts in potential species richness due to increased wood pellet demand were considerably smaller compared to the changes due to other drivers, such as urbanization and increased timber demand. Biodiversity impacts due to wood pellet demand should therefore be considered in the context of other drivers of land-use change and biodiversity loss. Our results provide information that allows policymakers, industry and NGOs to focus on areas of concern and take appropriate mitigation measures to limit negative biodiversity impacts and promote positive impacts. The spatially explicit approach presented in this study can be applied to different regions and drivers of land-use change, to show how projected demand for an internationally traded commodity may lead to impacts on land use and biodiversity in the procurement region.

KEYWORDS
Bioenergy, land-use change, pine plantation, potential species richness, solid biomass, spatial analysis

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1 | INTRODUCTION

Exports of wood pellets from the United States (US) have grown from close to zero in 2008 (Dale, Parish, Kline, & Tobin, 2017) to 4.7 Mt per annum in 2016 (Copley & Lang, 2017). Of all exported pellets from the United States, about 98% is produced in the eastern United States (Wang, Dwivedi, Abt, & Khanna, 2015). The wood pellet producing sector in the United States is largely driven by demand from the European Union (EU) (Abt, Abt, Galik, & Skog, 2014; Joudrey, McDow, Smith, & Larson, 2012), which is in turn determined by policies within the EU member states (Abt et al., 2014). Future development in wood pellet production in the south-eastern United States is uncertain, partly because it relies heavily on European policies. Projections of wood pellet production in the region range from ~6 Mt (Pinchot Institute, 2013) to ~16 Mt per annum in 2020 (Abt et al., 2014). An increased demand for wood products, including wood pellets, has been projected to lead to a change in timberland[1] area (Abt & Abt, 2013; Abt, Cubbage, & Abt, 2009; Duden et al., 2017) and a shift from natural timberland[2] to pine plantation (Duden et al., 2017; Evans et al., 2013). These changes in timberland area and management may exacerbate changes due to trends in other wood using markets such as the US domestic housing market and the pulp and paper market (Duden et al., 2017). Changes in forest area driven by wood markets occur in addition to forest losses due to urbanization, which is identified by the US Department of Agriculture (USDA) Forest Service as a major risk to forest sustainability in the south-eastern United States (Wear & Greis, 2013). The area of pine plantations in the south-eastern United States increased by over 54 thousand km^2 between 1990 and 2013 (an average increase of 2% per year), reaching almost 180 thousand km^2 in 2013 (USDA Forest Service, 2017). In the same time period, natural timberland decreased by almost 34 thousand km^2 (0.1% per year) to 620 thousand km^2 in 2013 (USDA Forest Service, 2017). An increased demand for wood pellets may exacerbate the existing trend of loss of natural timberland, while at the same time promote the establishment of pine plantations. These projected shifts in land use and forest management due to increasing demand for wood pellets could impact (either positively or negatively) biodiversity in the south-eastern United States (Parish, Herzberger, Phifer, & Dale, 2018).

The sustainability of wood pellet production has been questioned and discussed extensively for several reasons including its potential contribution to changes in land use and forest management (Costanza, Abt, Mckerrow, & Collazo, 2015, 2017 ; Duden et al., 2017; Prestemon & Abt, 2002) and impacts on biodiversity (Evans et al., 2013; Olesen, Kittler, Price, & Aguilar, 2016; Pelkmans et al., 2014; Tarr, Rubino, & Costanza, 2016). Land-use change and subsequent habitat loss have been the major cause of biodiversity loss in terrestrial ecosystems over the last 50 years (Millenium Ecosystem Assessment, 2005; Pimm & Raven, 2000; Secretariat of the Convention on Biological Diversity, 2010; WWF, 2012), and are projected to continue to drive biodiversity losses in the future (Jetz, Wilcove, & Dobson, 2007; Sala et al., 2000). The south-eastern United States is rich in endemism (Jenkins, Houtan, Pimm, & Sexton, 2015), and a large part of the region, the North Atlantic Coastal Plain, has been identified as a global biodiversity hot spot (Noss et al., 2015). Two important land-use trends that may lead to biodiversity loss are expected in the south-eastern United States in the coming decades: urbanization (McKinney, 2008; Price, Dorcas, Gallant, Klaver, & Willson, 2006) and expansion of pine plantation (Duden et al., 2017). Pine plantations in the south-eastern United States are currently among the most intensively managed forests in the world (Fox, Jokela, & Allen, 2007). Some field studies have found lower species diversity in plantation forests compared to natural forests, although other studies have also found similar or higher diversity in forest plantations (Carnus et al., 2006; Stephens & Wagner, 2007). Biodiversity in pine plantations strongly depends on plantation age (Carnus et al., 2006), forest management (Carnus et al., 2006; Miller, Wigley, & Miller, 2009) and abiotic factors (Miller et al., 2009). Biodiversity impacts of plantation establishment may differ according to the initial land use and spatial heterogeneity of abiotic factors (Immerzeel, Verweij, Hilst, & Faaij, 2014; Tarr et al., 2016). As a result, it is important to assess impacts of land-use change due to wood pellet demand on biodiversity in a spatially explicit manner.

Several European countries have discussed sustainability guidelines that may limit the sourcing of wood pellets from the south-eastern United States (Galik, Abt, Latta, & Vegh, 2015). Biodiversity loss has been identified as one of the major EU policy risks regarding the wood pellet imports from the region (Olesen et al., 2016). The effect of biomass production on ecosystems has been identified as one of the

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[1]Timberland is defined as "forest land that is producing or is capable of producing crops of industrial wood and not withdrawn from timber utilization by statute or administrative regulation. (Note: Areas qualifying as timberland are capable of producing in excess of 20 cubic feet (1.4 cubic meters) per acre (0.4 hectare) per year of industrial wood in natural stands. Currently inaccessible and inoperable areas are included.)" following the US Forest Service (Oswalt et al., 2014).

[2]Natural timberland is defined as "Productive forests composed of trees established by natural regeneration of existing seed sources, root suckers, stump sprouts, etc. Establishment may be either afforestation on land that until then was not classified as forest or by reforestation of land classified as forest after a disturbance or following harvest" following the Forest Inventory and Analysis National Program (see for instance Oswalt et al., 2014).

40 scientific priorities to inform US conservation managers and policymakers (Fleishman et al., 2011). The European Renewable Energy Directive (RED) aims to increase the share of renewable energy, while ensuring that "areas of high conservation value are preserved" and "the impacts of forest harvesting on biodiversity are minimised" (European Commission, 2017). Therefore, to fulfil the requirements of the RED, it is necessary to identify areas of high conservation value throughout sourcing areas of the south-eastern United States, and determine potential impacts of wood pellet production on biodiversity. No uniform guidelines currently exist on how to quantify these biodiversity impacts. Apart from the political need for impact assessments of increasing wood pellet demand, biodiversity has an intrinsic and economic value that requires conservation.

A number of studies have assessed the potential impact of wood pellet production on biodiversity in the south-eastern United States using a spatially explicit approach. Some studies showed overlap of wood pellet sourcing areas with areas of high biodiversity value. For example, Evans et al. (2013) examined the overlap of habitats for 16 conservation priority species with sourcing areas surrounding six wood pellet mills in Virginia, North Carolina and Georgia. Procurement areas of existing and proposed pellet plants were also found to overlap with important bird areas (Olesen et al., 2016). The latter study focused on the entire south-eastern United States but only included areas important for bird species. In some locations, Galik and Abt (2015) found an overlap of over 40% between presently harvestable forest areas and areas of potential harvest restrictions based on proposed EU sustainability guidelines for wood pellets. This study spanned the entire south-eastern United States, but focused on areas that could be excluded according to the EU sustainability criteria without determining potential impacts outside these excluded areas. Finally, Tarr et al. (2016) assessed the impact of land-use transitions due to bioenergy production in North Carolina on the habitat area of 16 indicator species in a spatially explicit manner. Wood pellet production was projected to lead to habitat increases for some species but decreases for others, indicating that land-use transitions will lead to both winners and losers. However, Tarr et al. (2016) assessed just one state (North Carolina) within the wood pellet producing region. Studies have identified regions of potential biodiversity risk due to feedstock harvesting for wood pellets, including coastal Georgia and Virginia (Olesen et al., 2016), Florida and the Atlantic coastal area (Galik & Abt, 2015). Species at risk included threatened species such as the Cerulean Warbler (*Setophaga cerulean)* and Eastern Spotted Skunk (*Spilogale putorius*) (Evans et al., 2013; Tarr et al., 2016). The area of dense, shrubby vegetation may increase in bioenergy scenarios, providing additional habitat for some species including the southeast endemic Oak Toad (*Anaxyrus quercicus*) (Tarr et al., 2016). Spatially explicit studies on the impacts of woody biomass production on biodiversity in the south-eastern United States have assessed the impacts either for a limited number of species (Evans et al., 2013; Olesen et al., 2016) or for a restricted area in the wood pellet sourcing region (Galik & Abt, 2015; Tarr et al., 2016). Using broad taxonomic groups as indicators has been identified as a research need in the study of biodiversity impacts of bioenergy (Fletcher et al., 2011; Immerzeel et al., 2014; Riffell, Verschuyl, Miller, & Wigley, 2011a, 2011b ; Verschuyl, Riffell, Miller, & Wigley, 2011), partly because of contradictory findings for selected species groups as indicators (Dauber et al., 2003).

The aim of this study was to identify and quantify potential future biodiversity impacts of an increased wood pellet demand in the south-eastern United States in a spatially explicit manner. This study focuses on the whole south-eastern United States in order to include all areas where pellets will likely be sourced in the near future. We assess the impact of scenarios with and without increased wood pellet demand between 2010 and 2030 (according to land-use projections from a previous study by Duden et al. (2017)) on projected potential species richness for all terrestrial vertebrate species reported for the region.

2 | METHODS

2.1 | General approach

This study makes use of projections of land use up to 2030 following scenarios of demand for wood pellets and other wood products derived from Duden et al. (2017). In order to determine the impact of an increasing demand for wood pellets on biodiversity, we created potential species richness maps for the south-eastern United States by summing all terrestrial vertebrate potential habitat maps available at the USGS Gap Analysis Program database (McKerrow, Tarr, Rubino, & Williams, 2018). We then generated an index based on location-specific associations between land-use types and potential species richness by overlaying potential richness maps with land-use maps taken from Duden et al. (2017) (Figure 1). Finally, we projected the species richness index in 2030 according to projections of land-use change. To determine the potential impact of increased wood pellet demand on biodiversity, we focussed the analyses on: (a) spatial variation in species richness index within and among different land-use types, (b) projected baseline changes in species richness index between 2010 and 2030 in the absence of increasing wood pellet demand and (c) potential impacts of scenarios of increased pellet demand on biodiversity.
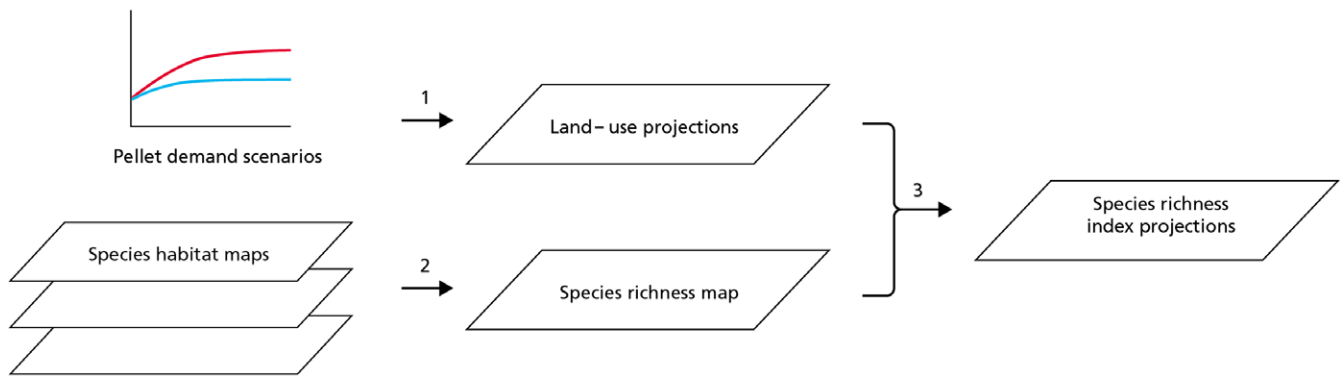
FIGURE 1 Overview of general approach to create projections of species richness index. Rectangles symbolize maps, and the stack of rectangles symbolizes individual species habitat maps of 811 species included in the analysis. Arrows symbolize adaptations to the maps; 1) modelling of wood pellet demand scenarios using the economic wood market model SRTS and land-use change model PLUC to create different land-use projections up to 2030 (see Duden et al., 2017), 2) summing of species potential habitat maps taken from US-GAP (McKerrow et al., 2018) to create maps of potential species richness and 3) spatial analysis using a moving window approach to create spatially explicit projections of species richness index up to 2030

2.2 | Pellet demand scenarios and land-use projections

In a previous study (Duden et al., 2017), we created land-use maps for 2010 and 2030 under different scenario assumptions using a combination of the subregional timber supply (SRTS) model (Abt, Cubbage, & Pacheco, 2000) and the PCRaster Land-Use Change (PLUC) model (Van der Hilst, Verstegen, Karssenberg, & Faaij, 2012; Verstegen, Karssenberg, Hilst, & Faaij, 2012). The scenarios consisted of four alternative trends of developments of wood markets (including wood pellets) in the south-eastern United States, as well as projections of developments in different land-use types. The wood market projections included a trend of increasing wood pellet demand (12.1 Mt by 2030) and a trend of stable low wood pellet demand (0.5 Mt by 2030, see Figure 2) (Duden et al., 2017). Because the wood pellet sector in the south-eastern United States is strongly influenced by developments in the domestic housing market (Duden et al., 2017), the biggest timber consuming sector, trends of high or low domestic housing demand were also included in the scenarios. Scenarios consisted of a combination of demand projections for wood pellets (high or low) and demand projections for timber (high or low), resulting in four different scenarios (see Figure 2): HhHp (high housing demand, high pellet demand), HhLp (high housing demand, low pellet demand), LhHp (low housing demand, high pellet demand) and LhLp (low housing demand, low pellet demand). Land-use projections were created in the PLUC model and based on projections of timberland (both natural and planted) area based on the wood market scenarios, as well as projections for agricultural land, pasture and urban land. The PLUC model allocates land use according to the projected area per land use,

an allocation order of the different land uses and suitability maps for each land use. The resulting land-use projections were at a 2×2 km resolution and include 11 different land-use types. These include four anthropogenic land-use types (urban, cropland, pasture and pine plantation); four natural forest types (natural pine, mixed forest, upland hardwood forest and lowland hardwood forest); nonforest vegetation, which comprises all natural land-use types not classified as forest; and two land-use types for which projections are stable over time (federal land and water). Between 2010 and 2030, 9.4% to 12.3% of the area is projected to be subject to a transition in land use (Duden et al., 2017). The most common land-use transitions are natural pine to pine plantation, pasture to natural pine and nonforest vegetation to pasture. We refer to Supporting information Appendix S12 and Duden et al. (2017) for a more detailed description of the wood product demand trends and the land-use projections.

2.3 | Species richness data

Species richness is an important determinant of ecosystem function and forest productivity (Liang et al., 2016). Ecosystems with higher species richness have shown to be able to sustain higher levels of multiple ecosystem functions (Lefcheck et al., 2015; Soliveres et al., 2016; Zavaleta, Pasari, Hulvey, & Tilman, 2010) and to maintain these ecosystem functions while subjected to changes in environmental conditions or loss of species (Folke, Holling, & Perrings, 1996; Potter & Woodall, 2014). Threatened species are those considered to be at relatively high risk of (local) extinction in the near future (Dobson, Rodriguez, Roberts, & Wilcove, 1997; Steck, Bürgi, Coch, & Duelli, 2007) and as such provide a group of species of
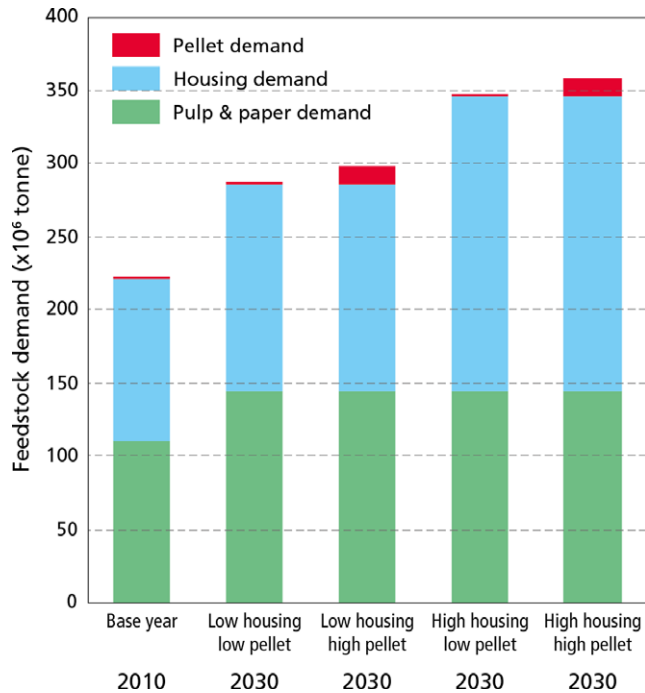
FIGURE 2 Demand for wood feedstock from the south-eastern United States for the US domestic housing market, the pulp and paper industry and the wood pellet sector in 2010 and as assumed in the four different scenarios (LhLp, LhHp, HhLp and HhHp) in 2030 (Duden et al., 2017)

conservation concern (Boyd et al., 2008). For this study, threatened species were defined as those species listed as 1) "vulnerable," "endangered" or "critically endangered" according to the IUCN Red List (International Union for Conservation of Nature, 2017), and/or 2) listed "threatened" or "endangered" by the Endangered Species Act (Department of the Interior U.S. Fish & Wildlife Service, 1979) and/or 3) listed globally in the category of "critically imperilled," "imperilled" or "vulnerable" by NatureServe (Master et al., 2012). When species consisted of several subspecies with an individual conservation status, these subspecies were included separately in the analysis. Endemic species are considered priorities for conservation because of their uniqueness and extinction risk (Kier & Barthlott, 2001; Laffan & Crisp, 2003). Endemism is the restriction of a species' range to a defined area, meaning it occurs uniquely within the boundaries of that area (Laffan & Crisp, 2003). In this study, endemic species were defined as those species that have their entire potential habitat area within the south-eastern United States.

The GAP data set includes potential habitat maps for all terrestrial vertebrate species in the United States at 30 × 30 m resolution (McKerrow et al., 2018). GAP potential species habitat maps were created using a deductive modelling approach in which potential species habitat is determined by selecting only areas within its range, that is

where a species could be expected to persist according to scientific literature. Potential habitat is determined by selecting areas within the species' range that are considered suitable based on land use (as classified in the GAP land-use map), hydrology, elevation, patch size, human disturbance, canopy cover and forest edge effects. The GAP land-use map was derived from 1999–2001 Landsat satellite imagery at 30 × 30 m resolution and contains 280 different land-use types within our study region (U.S. Geological Survey, 2011). The GAP potential habitat maps of individual species reflect potential distribution. We generated maps of potential species richness by summing potential habitat maps of individual species (McKerrow et al., 2018). While these individual GAP species potential habitat maps were not assessed for accuracy, previous iterations of GAP modelling efforts related to species potential distribution projects at state and regional level using nearly identical deductive methodologies produced accuracy rates of 65%–80% (Aycrigg et al., 2015; Maxwell & Gergely, 2005; Maxwell, Gergely, Aycrigg, & Davidson, 2009; McKerrow, Williams, & Collazo, 2006). The latest update of the complete data set has undergone a USGS data review and has been included in McKerrow et al. (2018. Since abundance data are not available for all species in the study area, we were not able to assess biodiversity indicators based on species abundance, such as the Simpson index (Simpson, 1949), Shannon index (Shannon-Wiener, Weaver, & Weater, 1950) or the mean species abundance (Alkemade et al., 2009).

2.4 | Biodiversity indicators

Many different biodiversity indicators exist, and no single indicator can measure the full complexity of biodiversity (Noss, 1990). The impact of wood pellet demand on biodiversity was quantified in this study using a combination of the following biodiversity indicators:

- Total potential species richness (of all terrestrial vertebrate species and of the individual taxonomic groups of amphibians, birds, mammals and reptiles)
- Threatened potential species richness
- Endemic potential species richness

Species richness is widely used (Fleishman, Noss, & Noon, 2006) and is well established as an indicator for biodiversity because of its relative ease to calculate and interpret (Dobson et al., 1997; Lamb et al., 2009; Orme et al., 2005; Williams & Gaston, 1994). A downside of using total species richness as a biodiversity metric is its inability to differentiate among species with different conservation values based on, for example, endemism or sensitivity to threat (Fleishman et al., 2006). Therefore, we selected total

species subsets using endemic and threatened species as separate biodiversity indicators. We assessed total potential species richness, as well as potential species richness specific for different taxa, because spatial richness patterns and response to environmental and land-use change have been shown to vary among taxa in the United States (Gregory et al., 2005; Hof, Araújo, Jetz, & Rahbek, 2011; Jenkins et al., 2015; McKerrow et al., 2018). Using species' habitat distribution data from the US Geological Survey's Gap Analysis Project (GAP) (McKerrow et al., 2018), all terrestrial vertebrate species (amphibians, birds, mammals and reptiles) whose habitat lies within the study area were included in the analysis (see Supporting information Appendix S1 for a detailed description of the study area). A total of 811 terrestrial vertebrate species were reported for the study region, including 135 threatened species and 140 endemic species (see Table 1). Supporting information Appendix S2 provides a definition and a list of threatened and endemic species in the south-eastern United States.

2.5 | Spatial analysis

Our aim was to estimate changes in potential species richness driven by the projected land-use change in the south-eastern United States (Duden et al., 2017). To do this, maps of potential species richness were combined with maps of land-use types by assigning spatially variable values of potential species richness (McKerrow et al., 2018) to each land-use type, in a spatial analysis described in more detail below and in Supporting information Appendix S4. Due to the difference in resolution and years of origin of the GAP data (30×30 m, 2001) and PLUC land-use output (2×2 km, 2010), we had to develop a method to align these two maps (see Figure 3). This method consists of a spatial analysis, which assigned values of potential species richness from the GAP habitat suitability data to land-use projections. Land-use projections were based on modelling with the PLUC model at a county-sized resolution (on average ~1,520 km^2 or 39×39 km) because sensitivity analyses indicated that PLUC output finer than that resolution was less accurate (Duden et al., 2017). Our method links two spatial data

sets with different spatial resolutions to produce results at a resolution coarser than either one, while retaining location and land-use-specific information on potential species richness.

Potential species richness maps were created by aggregating all individual species habitat maps from the GAP database (step 1 in Figure 3). As a result, potential species richness of a cell is defined in this study as the number of species for which the cell contains suitable habitat according to the GAP data. The resulting potential species richness maps reflect the land use in 2001 as characterized by the GAP program (as well as other factors including elevation and hydrology). Because the PLUC land-use projections and GAP habitat maps utilize different land-use classifications, we reclassified the 280 GAP land-use types to match PLUC land-use types (see Supporting information Appendix S3 for a more detailed description of the reclassification process) (step 2). Spatially variable land-use-specific richness values were then determined by creating separate potential species richness maps for each land use, using the reclassified GAP land-use map and the potential species richness map (step 3, see Supporting information Appendix S4 for a more detailed description). A species richness index (SRI) for each PLUC cell (2×2 km) for 2010 was then calculated using a spatial neighbourhood analysis (step 4, see Supporting information Appendix S4 for a more detailed description of the neighbourhood analysis). The process was repeated for the 2030 PLUC land-use maps. This resulted in 2×2 km resolution maps of SRI for each biodiversity indicator (total, endangered and endemic potential species richness and the four taxonomic groups) and scenario (HhHp, HhLp, LhHp and LhLp, step 5 in Figure 2).

The species richness index map for 2010 created in step 4 was used to assess differences within and among projected SRI values per land-use type, by calculating average SRI and ranges of SRI for each land-use type (step 6). Species richness index in 2010 was then compared to species richness index in 2030 for the low pellet demand scenarios (HhLp and LhLp), to assess the baseline changes in SRI occurring between 2010 and 2030 in the absence of increased wood pellet demand (step 7). This was done by subtracting the 2010 SRI map from the 2030 SRI maps, creating $\Delta SRI_{2010\text{-}2030}$ (see Equation (1). The resulting maps of species richness index change ($\Delta SRI_{2010\text{-}2030}$) were then divided into positive and negative changes, and aggregated (mean) to 39×39 km resolution. SRI values in 2030 for scenarios with and without an increased pellet demand were then compared to determine the differences in species richness index attributable to increased wood pellet demand (see Equation (2) (step 8)). This was calculated both for the high housing and low housing demand scenarios. Each of the resulting maps of the differences in

TABLE 1 Species count for total, threatened and endemic species in the south-eastern United States, separated per taxonomic group

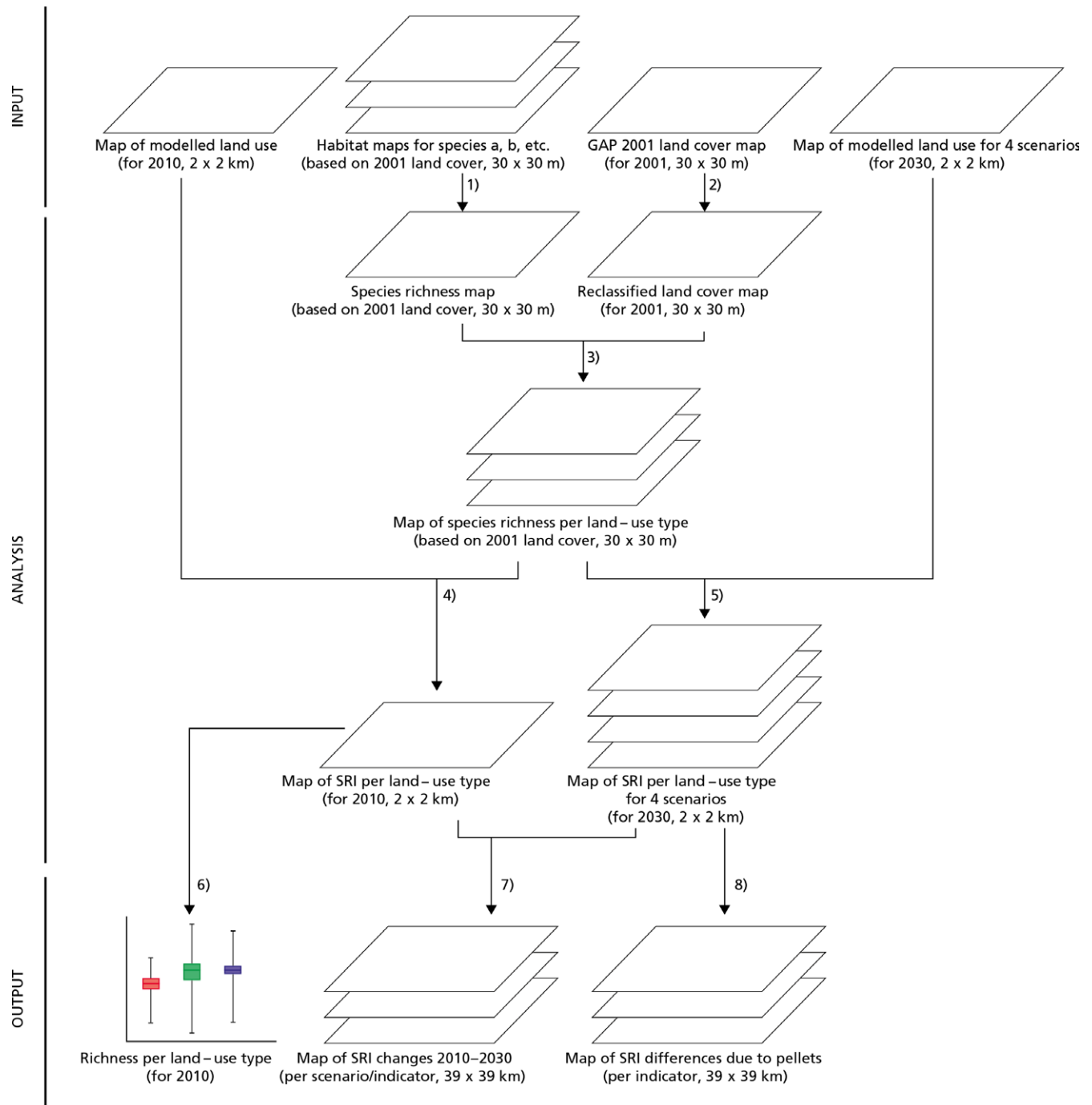|	Total	Threatened	Endemic
Mammal	123	36	23
Bird	372	19	3
Reptile	145	33	39
Amphibian	171	47	75
Total taxa	811	135	140

FIGURE 3 Overview of the method of the spatial analysis to determine species richness index in 2030. Rectangles symbolize maps, while arrows indicate adaptations to the map. SRI, species richness index

species richness index ($\Delta SRI_{pellets2030}$) was then divided into positive and negative changes, and aggregated (mean) to 39 × 39 km resolution.

$$\Delta SRI_{2010-2030} = SRI_{2030-LP} - -SRI_{2010} \quad (1)$$

$$\Delta SRI_{pellets2030} = SRI_{2030-HP} - -SRI_{2030-LP} \quad (2)$$

$\Delta SRI_{2010-2030}$: Change in richness between 2010 and 2030 for low wood pellet demand scenario.

$\Delta SRI_{pellets2030}$: Difference in richness between high and low wood pellet demand scenarios by 2030.

SRI_{2010}: Potential species richness index in 2010.

$SRI_{2030-LP}$: Potential species richness index in 2030 in the low pellet demand scenario (HhLp or LhLp).

$SRI_{2030-HP}$: Potential species richness index in 2030 in the high pellet demand scenario (HhHp or LhHp).

The SRI output maps were aggregated from a resolution of 2 × 2 km to 39 × 39 km to reflect uncertainty of the

input map as well as uncertainty introduced in the analysis. This means that SRI calculations at 2 × 2 km are based on averages at a 39 × 39 km window in the analysis, after which the maps are aggregated to a 39 × 39 km resolution for visualization of the results. Because our spatial processing methods introduced uncertainty by averaging and aggregating richness, while assuming land-use change to be the sole driver of changes in potential species richness, our results were expressed as "index" of species richness. Therefore, species richness index values do not signify absolute numbers of species projected to be lost or gained at a given location. Rather, they give an indication of the possible intensity of shifts in potential species richness that is based on what we know about the spatial distributions of potential species richness in each of the land cover classes. The unit of the potential species richness index is average number of species per 30 × 30 m, which is the original resolution of the species habitat maps. However, as mentioned above, this is provided at a 39 × 39 km resolution. All subsequent mentions of species richness refer to this species richness index or SRI. When visualizing changes in species richness index, we provided figures for species richness increases and species richness decreases separately. Due to the aggregation to a large cell size (39 × 39 km) and the fact that land-use transitions resulting in SRI losses and as well as those resulting in SRI gains coincided in some areas, both positive and negative changes were projected within some single 39 km cells. As a result, simply averaging SRI values could result in a loss of information. For instance, if within a cell large positive shifts and large negative shifts in SRI occur, the average may even out and result in a low average SRI change.

3 | RESULTS

The potential impact of increased demand for wood pellets on biodiversity is discussed in four steps. First, we show spatial variation in potential species richness index in 2010. We then discuss species richness index (SRI) values in 2010 per land-use type. Next, we discuss projected changes between 2010 and 2030 in SRI without an increased pellet demand. Finally, we examine the additional impacts due to an increased demand for wood pellets.

3.1 | Species richness in the south-eastern United States

The mean total species richness index in the south-eastern United States in 2010 was 108 species (standard deviation = 24), and the total SRI ranged from 48 to 183 (see Table 2). Of the four taxonomic groups, birds had the highest SRI (mean = 62) and amphibians the lowest (mean = 7). The threatened and endemic SRI ranged from 0 to 15 (threatened species) and 0 to 20 (endemic species). The total SRI in the south-eastern United States in 2010 was generally highest along the Coastal Plains (Figure 4a), with highest values mostly overlapping with the presence of lowland hardwood forest (Figure 4b). Some areas with high SRI occurred in the western part of the study area, corresponding with species-rich pasture land (see Supporting information Appendix S5 for SRI patterns per land-use type). The high SRI in Tennessee and surrounding areas to the south and east are due to high densities of species-rich mixed forest and upland hardwood forest in this area. Low species richness index in eastern Arkansas, northern Louisiana and western Mississippi reflects the abundance of cropland. Low SRI in eastern North Carolina and Virginia is the result of high densities of cropland and pine plantations. Threatened species richness index was relatively high in the northern part of the study area, in Tennessee and Arkansas (see Supporting information Appendix S6 for SRI for different indicators). This region contains concentrations of upland hardwood forest and mixed forest, two land-use types with the highest SRI of threatened species. Endemic SRI was strongly concentrated in the south-eastern part of the study region, particularly in Florida and Georgia, a pattern illustrating the gradient of increasing SRI towards the south-eastern part of the study area within all land-use types (Supporting information Appendix S5). SRI for the different taxonomic groups generally showed similar patterns. The patterns described above were more pronounced for amphibians, which showed high SRI values in lowland hardwood forest but much lower values in other land-use types. Mammal SRI was more concentrated in the northern part of the study region due to high density of upland hardwood forests.
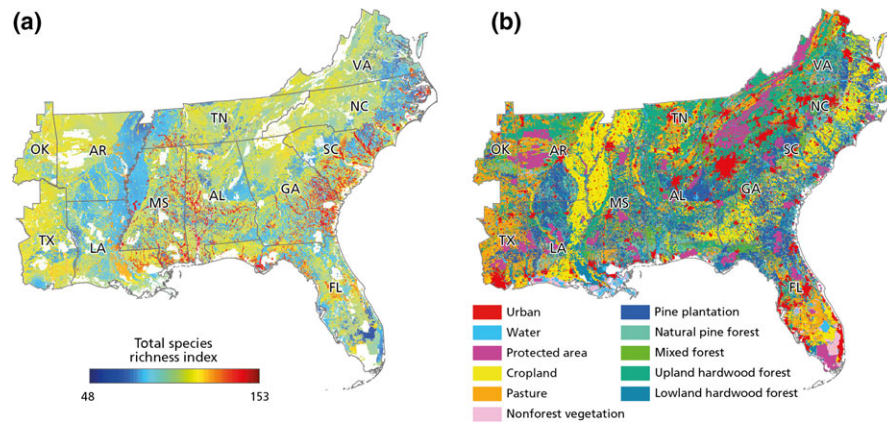
Minimum (min), maximum (max) and mean potential species richness index in the south-eastern United States in 2010 for seven biodiversity indicators

Statistics	Threatened	Endemic	Mammal	Bird	Reptile	Amphibian	Total
Mean	6	2	22	62	16	7	108
Min	0	0	5	27	3	0	48
Max	15	20	39	104	43	36	183
SD	3	4	6	10	7	7	24

SD: standard deviation.

FIGURE 4 (a) Projected total species richness index in the south-eastern United States for 2010. White areas contain Federal lands or open water, which were not projected to change over time and were not included in the biodiversity analysis. Figure 4b: Land use in the south-eastern United States for 2010. Black lines show state boundaries, and abbreviations indicate state names. AL: Alabama; AR: Arkansas; FL: Florida; GA: Georgia; LA: Louisiana; MS: Mississippi; NC: North Carolina; OK: Oklahoma; SC: South Carolina; TN: Tennessee; TX: Texas; VA: Virginia

3.2 | Potential species richness in land-use classes

Total species richness index was highest in lowland hardwood forests (mean = 156) followed by the other natural forest types (natural pine (mean = 115), mixed forest (mean = 117), upland hardwood forest (mean = 118)) and pasture (mean = 117; Figure 5 and Table 3). Natural forests (apart from lowland hardwood forest) and pasture showed comparable mean and range in potential species richness index. As a result, land-use changes from forest to pasture are projected to result in either increased or decreased potential species richness, depending on the potential species richness values for these land-use types in a particular area. Natural forest types showed a large range of total SRI over the study area (range = 52 (minimum

mixed forest) – 183 (maximum lowland hardwood)), reflecting the spatial variation between and within these land-use types. Nonforest vegetation had the largest range of total SRI (range = 48–172). Urban area showed relatively high total species richness index compared to the other anthropogenic land-use classes (mean = 103). Cropland and pine plantation had the lowest total SRI (cropland (mean = 81) and pine plantation (mean = 79). Pine plantation showed little variation in total SRI (range = 64–93). Similar patterns of total species richness index per land-use type were found for the taxonomic groups (Table 3, Supporting information Appendix S7). Total SRI of pine plantations and cropland was low, and this was due to low index values for all taxonomic groups. Lowland hardwood forest had high total SRI across all taxonomic groups, especially amphibians. Bird species showed a slightly different
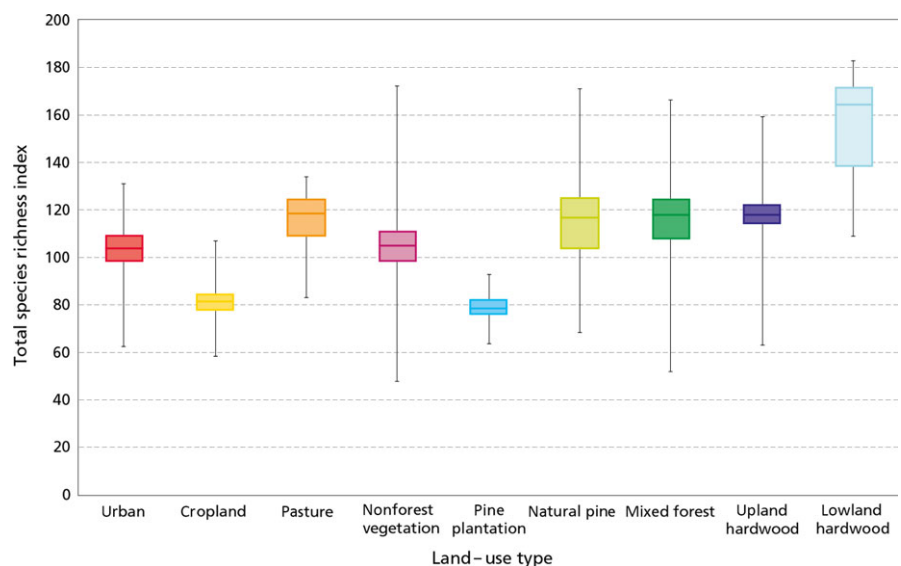
FIGURE 5 Potential total species richness index per land-use type in the south-eastern United States for 2010. Boxplots show median values (middle horizontal bar) and first (upper limit) and third quartile (lower limit) values, while vertical bars show maximum and minimum values

TABLE 3 Mean and standard deviation (*SD*) of species richness index within land-use classes of the south-eastern United States for seven biodiversity indicators

Species richness	Statistic	Urban	Cropland	Pasture	Pine plantation	Natural pine	Mixed forest	Upland hardwood	Lowland hardwood	Nonforest vegetation
Total	Mean	103.0	81.1	116.9	79.3	114.5	117.2	118.1	156.4	105.6
	SD	9.0	5.9	8.4	4.6	16.3	14.1	6.9	17.2	10.8
Threatened	Mean	3.1	3.5	5.8	4.5	8.2	8.5	9.1	7.6	4.7
	SD	0.8	1.1	1.1	1.3	2.3	1.7	1.7	1.6	1.5
Endemic	Mean	0.4	0.4	1.3	0.4	5.9	3.3	1.0	6.3	2.5
	SD	0.7	0.7	2.3	0.7	4.9	4.4	2.6	6.5	3.2
Mammal	Mean	20.3	13.1	22.6	18.1	22.6	26.3	29.8	29.7	19.2
	SD	2.0	2.4	2.0	1.7	3.2	3.8	2.5	2.8	5.9
Bird	Mean	67.3	57.1	71.1	50.1	57.4	59.3	59.8	76.7	69.0
	SD	6.0	4.1	4.3	3.4	7.0	5.9	4.4	5.5	10.8
Reptile	Mean	11.5	9.3	17.1	9.2	26.4	22.2	18.6	26.6	13.0
	SD	1.8	1.2	2.2	0.9	5.4	6.1	3.5	5.9	4.0
Amphibian	Mean	4.0	1.6	6.1	1.9	8.1	9.4	9.9	23.5	4.4
	SD	1.1	0.9	2.4	0.8	3.4	5.6	2.0	6.2	3.4

pattern compared to other taxonomic groups, with relatively high SRI in urban areas, cropland, pasture and nonforest vegetation. In fact, bird species richness index was so high in urban areas that transition from forest to urban in some locations led to a projected positive change in potential bird SRI. The highest SRI for both threatened and endemic species was found for natural forest types, with the exception of endemic species in upland hardwood forests, which was relatively low. The lowest SRI values for threatened and endemic species occurred in urban areas, cropland and pine plantation.

3.3 | Species richness changes between 2010 and 2030 without additional wood pellet demand

We assessed the (positive and negative) changes in potential species richness index between 2010 and 2030 due to drivers other than an increasing wood pellet demand to assess which changes in species richness index would be projected to occur even in the absence of projected increase in wood pellet demand. The changes in total SRI ranged from −21 to 37 (mean increase = 1.1, mean decrease = −2.0) for the high housing scenario (HhLp). Changes in threatened species richness index ranged from −2 to 1 (mean increase = 0.1, mean decrease = −0.2), while changes in endemic species richness index ranged from −4 to 5 (mean increase = 0.1, mean decrease = −0.3).

Decreases in total, threatened and endemic species richness index were projected to occur along the Gulf Coast, particularly in Mississippi, Alabama and Florida, as well as along the Atlantic Coast, particularly in South Carolina (Figure 6). These decreases are mainly due to decreases in reptile species richness index (Figure 7). Decreases in total SRI were also projected for the area around the Ouachita Mountains in Arkansas, and this is mainly due to a decrease in SRI of birds and reptiles. Additionally, threatened species richness index was projected to decrease in the Appalachian area on the eastern borders of Tennessee, and this was likely primarily due to a lower SRI of threatened mammals and reptiles. The most common land-use transitions between 2010 and 2030 for the high housing demand and low pellet demand scenario (HhLp) were transitions from pasture and natural forest to urban area, pasture and natural forest to (other types of) natural forest, natural forest and pasture area to pine plantations and nonforest vegetation to pasture area (Duden et al., 2017). The most common land-use transitions between 2010 and 2030 that resulted in a reduced SRI were transitions from upland hardwood forest and pasture to urban, and from pasture to pine plantation (see Supporting information Appendix S9 for more information on specific land-use transitions and related changes in SRI).

Increases in SRI were generally low in the study area, apart from the southern tip of Florida, where relatively larger increases in species richness index were projected. This was due to increasing species richness index for all taxonomic groups. South-western Louisiana showed a decrease in bird SRI, but an increase in mammal, reptile and amphibian SRI (Figure 7). Results for the low housing scenario (LhLp) showed a similar pattern (Supporting information Appendix S8), indicating the same areas where

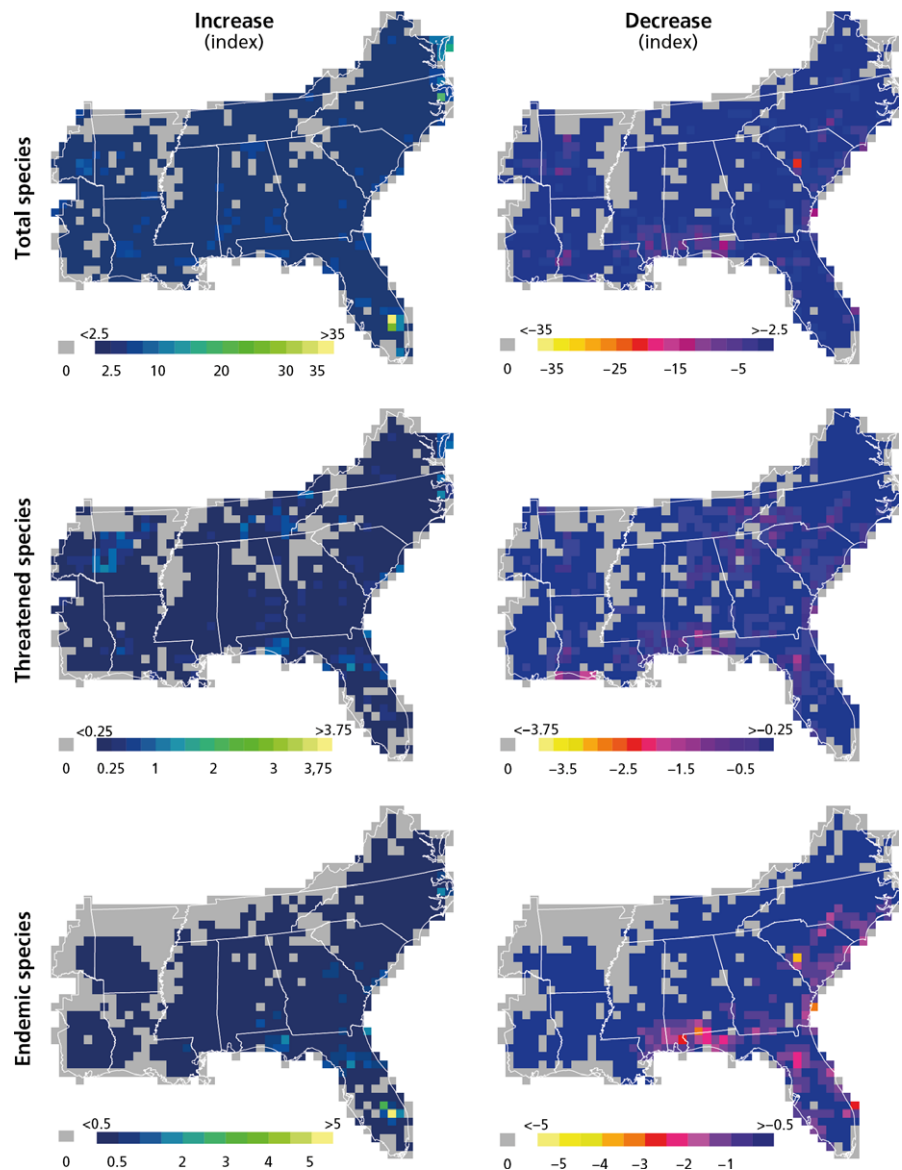**FIGURE 6** Projected changes in species richness index ($\Delta SRI_{2010\text{-}2030} = SRI_{2030\text{-}LP} - SRI_{2010}$) between 2010 and 2030 without additional wood pellet demand for different species richness indicators at 39×39 km resolution for the high housing scenario. Positive ($\Delta SRI > 1$) and negative ($\Delta SRI < 1$) changes were separated before aggregation.

Increase = window ($1,500$ km^2) average of positive changes in SRI.

Decrease = window ($1,500$ km^2) average of negative changes in SRI. Grey cells show areas where no change in species richness was projected to occur. AL: Alabama; AR: Arkansas; FL: Florida; GA: Georgia; LA: Louisiana; MS: Mississippi; NC: North Carolina; OK: Oklahoma; SC: South Carolina; TN: Tennessee; TX: Texas; VA: Virginia

the intensity of SRI changes is relatively high. Increases in total species richness index were the result of land-use transitions into nonforest vegetation (from cropland, pasture and pine plantation) and to natural forest (from nonforest vegetation, pasture and other natural forest types). Some transitions to urban area also resulted in some areas showing positive changes in total SRI, particularly for birds.

3.4 | Impact of wood pellet demand on biodiversity

We assessed the projected difference in potential species richness index by 2030 between the scenarios with and without an increased wood pellet demand (see Equation

(2). Projected differences in total species richness index due to an increased wood pellet demand ranged from -11 to 5 (mean increase = 0.4, mean decrease = -1.0) for the high housing (Hh) scenarios. For the low housing (Lh) scenarios, difference in total SRI ranged from -23 to 33 (mean increase = 0.7, mean decrease = -1.4) (see Supporting information Appendix S10 for results of the low housing scenario). Small differences in SRI were found throughout the study region, particularly in the Coastal Plains (Figure 8), while some areas show comparatively larger differences due to an increased pellet demand. In the high housing scenario, impacts of increased wood pellet demand lead to relatively large decreases in total SRI in patches in the southern tip of Florida, due to low richness index of bird, reptile and amphibian species (Figure 9).
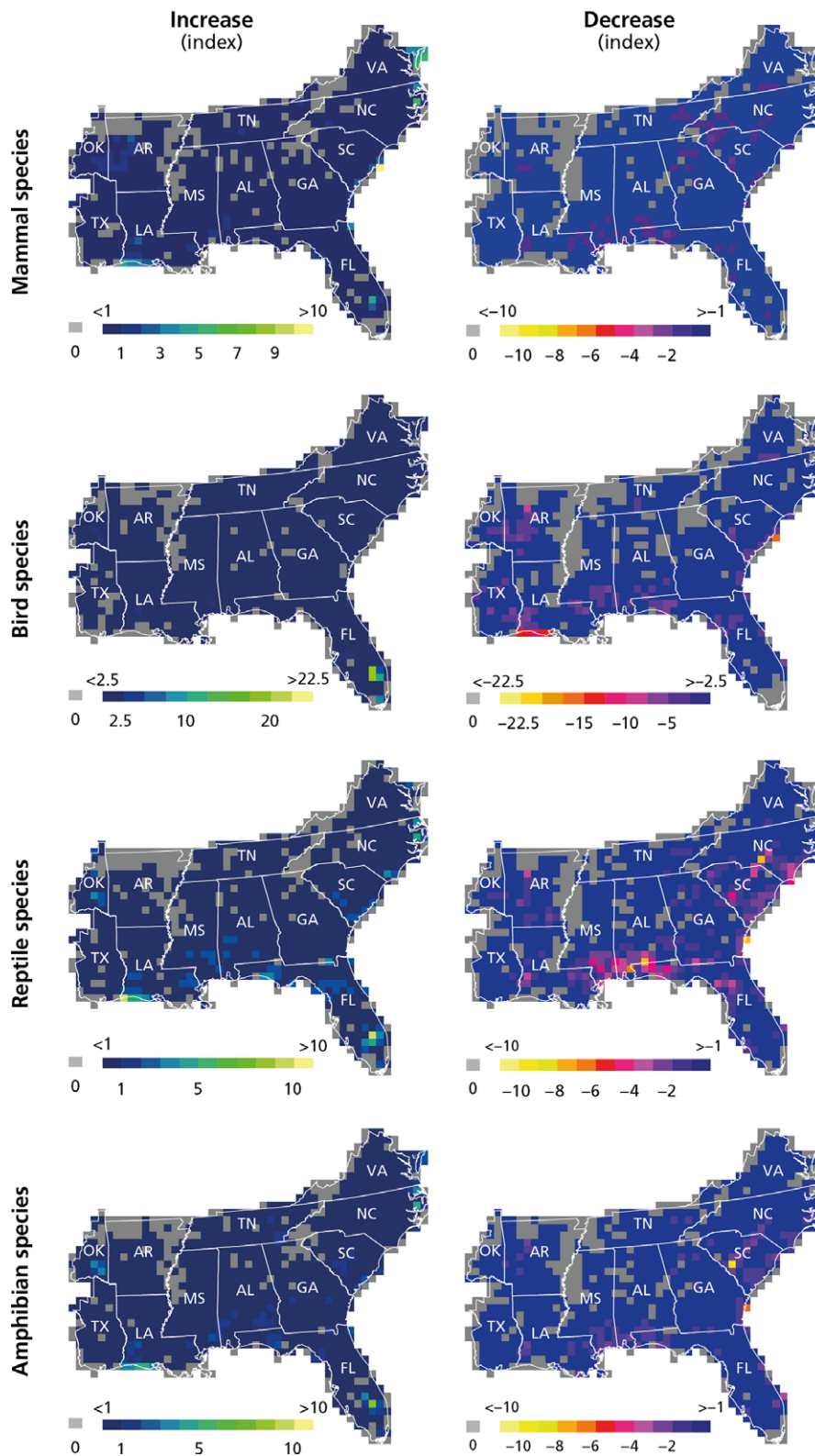
FIGURE 7 Projected changes in species richness index ($\Delta SRI_{2010\text{-}2030} = SRI_{2030\text{-}LP} - SRI_{2010}$) per taxonomic group between 2010 and 2030 without additional wood pellet demand for the high housing scenario. Positive ($\Delta SRI > 1$) and negative ($\Delta SRI < 1$) changes were separated before aggregation. Increase = window (1,500 km^2) average of positive changes in SRI. Decrease = window (1,500 km^2) average of negative changes in SRI. Grey cells show areas where no change in richness was projected to occur. AL: Alabama; AR: Arkansas; FL: Florida; GA: Georgia; LA: Louisiana; MS: Mississippi; NC: North Carolina; OK: Oklahoma; SC: South Carolina; TN: Tennessee; TX: Texas; VA: Virginia

Threatened species richness index, on the other hand, showed gains in this area, which is mainly due to high richness index values of mammal species. Endemic SRI is also low in the southern tip of Florida when wood pellet demand increases. Species richness index was also low due to increased wood pellet demand in the coastal areas of
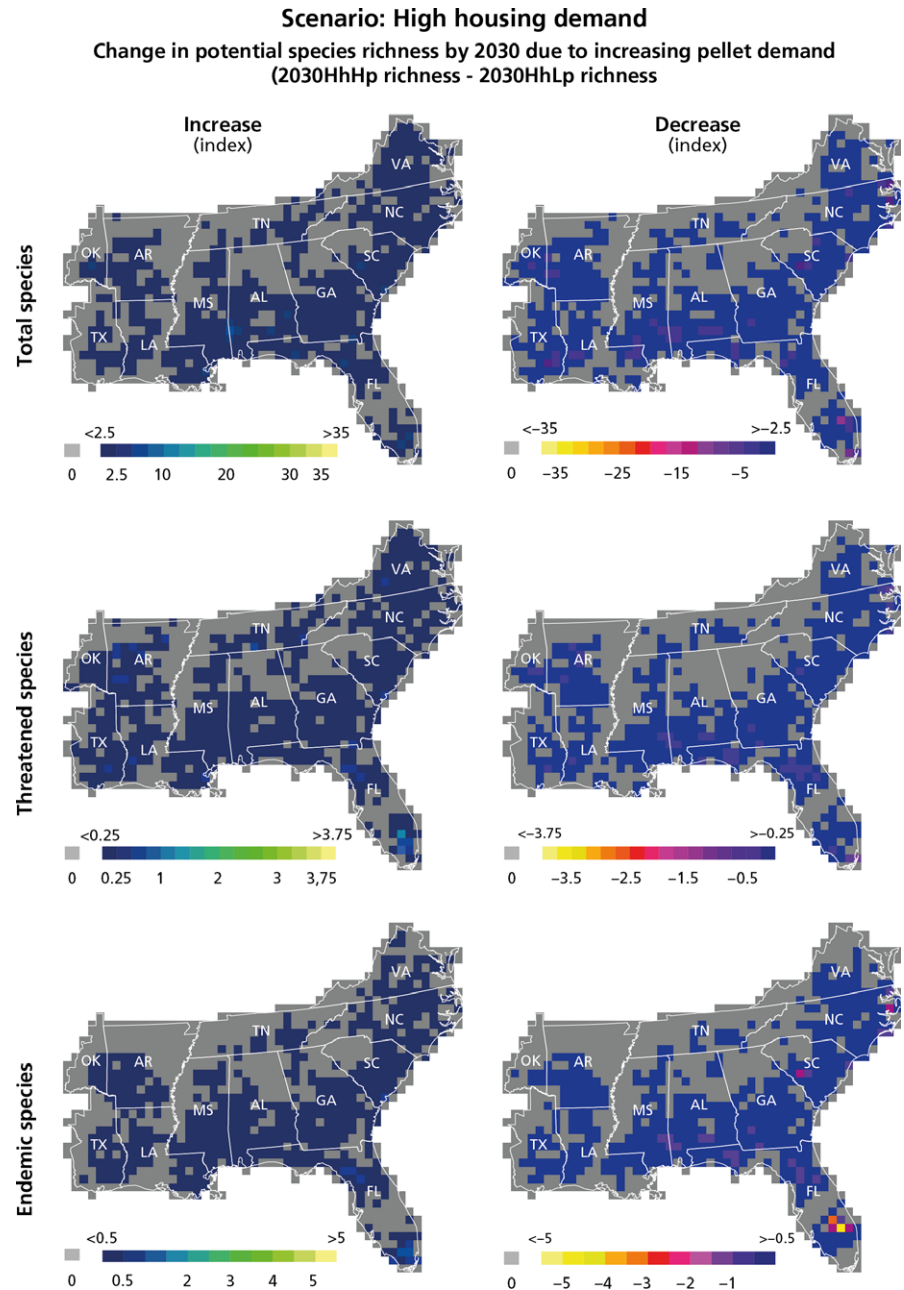
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FIGURE 8 The difference in species richness index ($\Delta SRI_{pellets2030} = SRI_{2030\text{-}HP} - SRI_{2030\text{-}LP}$) for total, threatened and endemic species richness by 2030 due to increased wood pellet demand for the high housing scenario. Positive ($\Delta SRI > 1$) and negative ($\Delta SRI < 1$) changes were separated before aggregation. Increase = window (1,500 km2) average of positive changes in SRI. Decrease = window (1,500 km2) average of negative changes in SRI. Grey cells show areas where no change in richness was projected to occur. AL: Alabama; AR: Arkansas; FL: Florida; GA: Georgia; LA: Louisiana; MS: Mississippi; NC: North Carolina; OK: Oklahoma; SC: South Carolina; TN: Tennessee; TX: Texas; VA: Virginia

north-eastern Florida and southern Alabama. This is mainly caused by a low SRI of reptiles.

When the demand for housing is assumed to be low, a small area of eastern Oklahoma shows relatively large positive differences in total SRI by 2030 due to increased wood pellet demand (Supporting information Appendix S10). This area showed high species richness index of birds under the high pellet and low housing scenario. Threatened species richness index in the Ouachita area in Arkansas is also relatively high, coinciding with high mammal and bird SRI. In the low housing scenario, relatively large negative differences were found in total, threatened and endemic species richness index in coastal Virginia—coinciding with low species richness index of birds—and North Carolina—

coinciding with low species richness index of mammals and amphibians. Negative differences in total SRI increased demand for wood pellets were mainly the result of transitions from natural forest, pasture and nonforest vegetation to pine plantations in the eastern and southern coastal regions, as well as transitions from species-rich pasture to natural forest in the western states in the study area (see Supporting information Appendix S11 for an overview of land-use transitions for the different pellet scenarios). Shifts from natural forest to pine plantation occurred throughout the study area but were more prevalent in southern Alabama and Mississippi, as well as in North and South Carolina. Positive differences in total SRI were caused by transitions to natural forest and pasture occurring along the coastal areas. The

Scenario: High housing demand

Change in potential species richness by 2030 due to increasing pellet demand
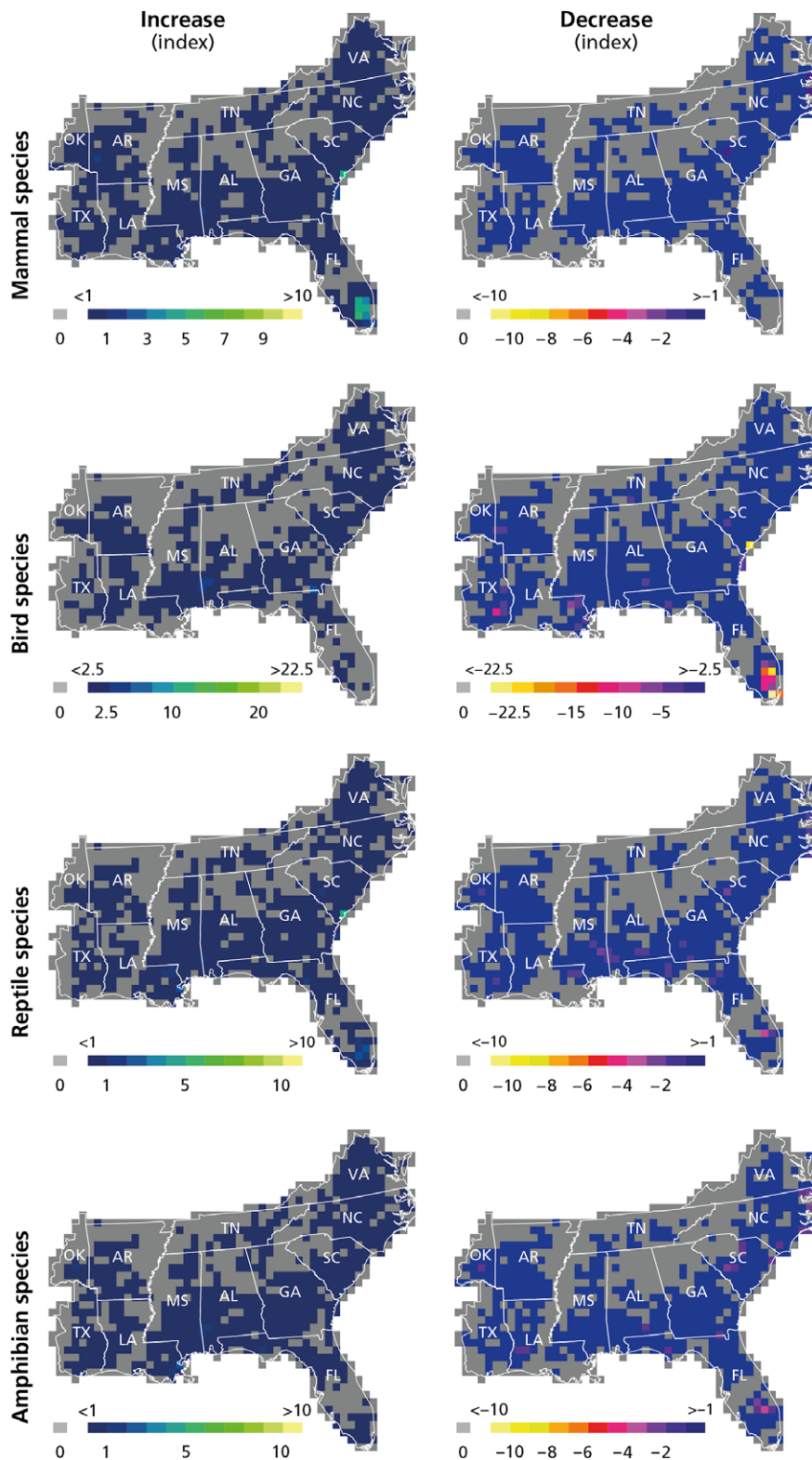(2030HhHp richness - 2030HhLp richness

FIGURE 9 Projected differences in species richness index $(\Delta SRI_{pellets2030} = SRI_{2030\text{-}HP} - SRI_{2030\text{-}LP})$ per taxonomic group by 2030 with additional wood pellet demand for the high housing scenario. Positive $(\Delta SRI > 1)$ and negative $(\Delta SRI < 1)$ changes were separated before aggregation. Increase = window $(1,500 \text{ km}^2)$ average of positive changes in SRI. Decrease = window $(1,500 \text{ km}^2)$ average of negative changes in SRI. Grey cells show areas where no change in richness was projected to occur. AL: Alabama; AR: Arkansas; FL: Florida; GA: Georgia; LA: Louisiana; MS: Mississippi; NC: North Carolina; OK: Oklahoma; SC: South Carolina; TN: Tennessee; TX: Texas; VA: Virginia

differences in SRI values were also high in some areas where a transition from natural pine forest and mixed forest to hardwood forest was projected to occur.

4 | DISCUSSION

An increasing demand for wood pellets was projected to cause both positive and negative biodiversity impacts in the south-eastern United States, and these impacts were location-specific. The effect of an increased wood pellet demand on the species richness index was projected to be relatively small and was not restricted to a particular area. We found that the SRI was relatively high in the Coastal Plains, particularly for the land-use type lowland hardwood forest. The Coastal Plains were also the region in which biodiversity impacts were likely to occur. Our study therefore agrees with earlier findings indicating the Coastal Plains as a region of potential biodiversity impacts of wood pellet demand (Galik & Abt, 2015; Olesen et al., 2016). Negative shifts in total SRI were projected to occur in southern Florida, along the coast of Virginia and North Carolina and along parts of the Gulf Coast. Positive shifts in total SRI were projected to occur in parts of Oklahoma and Arkansas. Threatened species showed positive shifts in SRI values in Tennessee, while endemic species index values showed negative shifts in coastal South Carolina. Some of the areas showing large shifts in SRI do not source wood pellets. The projected land-use change in these areas is the result of indirect land-use changes following wood pellet demand-driven expansion of natural and planted forest. The area of negative SRI changes was larger for reptiles than for birds, mammals and amphibians. In some areas, the direction of change differed per taxonomic group, and positive shifts were projected for one taxonomic group while negative shifts were projected for the other. The coastal area of Virginia in particular showed potential SRI shifts for several taxonomic groups due to increased pellet demand when housing demand was assumed to be low, positive impacts for mammals and reptiles and negative impacts for birds. When housing demand was assumed to be high, the southern tip of Florida showed potential for positive biodiversity impacts for mammals but potential negative impacts for birds, reptiles and amphibians. This shows that impacts on biodiversity differ per taxonomic group and highlights the importance of analysing different taxonomic groups in biodiversity impact assessments. When compared to the changes due to other drivers, including urbanization and increased demand for timber, the shifts in potential species richness due to an increased demand for wood pellets were small. Furthermore, the differences in SRI between scenarios of high and low demand for wood pellets were smaller when the demand from the housing market was assumed to be high. This shows that

the impact of wood pellet demand on biodiversity is influenced by developments in other sectors. The scenarios for wood pellet demand taken from Duden et al. (2017) run from 2010 to 2030. Between 2010 and the present, actual wood pellet demand has been closest to the trajectory of the "high pellet demand" scenario.

We found that, depending on the location, a specific land-use transition can lead to both increases and decreases in species richness index. This was for example the case for transitions from pasture to nonforest vegetation and from pasture to natural pine forest. This finding emphasizes the added value of a spatially explicit assessment of biodiversity impacts. Areas of positive biodiversity impacts overlapped partly with areas of negative impacts due to data aggregation. In these areas, land-use transitions that result in increases in SRI are projected to occur alongside transitions that result in decreases in SRI. Land-use transitions from natural timberland forest to urban area and pine plantations are responsible for the majority of the negative shifts in total SRI. Of these transitions, conversions from natural pine forest to pine plantation and from upland hardwood forest to urban area are most prevalent. Conversely, positive changes in SRI occurred when land was converted to natural forest or pasture, with transitions of nonforest vegetation and cropland to pasture being most common. These transitions accounted for about half of the areas with SRI increases. Therefore, we recommend land managers and planners to minimize biodiversity impacts by avoiding conversion of natural habitats to pine plantations, that is by focusing wood pellet sourcing from existing plantations.

The aggregation of data in this study has influenced the results. The PLUC land-use classification of the GAP data (280 land-use types) was aggregated to the classification of the land-use projections (11 land-use types). The number of land-use types included in an analysis has shown to influence the assessment of species richness (Lawler et al., 2004). Reclassification was however unavoidable, as it was unfeasible to create land-use projections of all 280 land-use types up to 2030. Reclassification resulted in a loss of detail, but the variation in potential species richness between the GAP land-use classes is included indirectly in the analysis by using local averages of potential species richness for that land use. Furthermore, land-use projections contain high levels of uncertainty that may influence subsequent environmental impact assessments (Prestele et al., 2016). Land-use projections were found to be most accurate for the land use types lowland hardwood forest, urban and cropland, and results were less accurate for mixed forest, natural and plantation pine forest, pasture and nonforest vegetation. The accuracy of PLUC is reduced with increasing spatial resolution, and sensitivity analyses indicated that PLUC output at a resolution higher than a county-sized level is less accurate (Duden et al., 2017).

This necessitated the aggregation of species richness index to a window size comparable to the average county in the study region (about 1,500 km^2), in order for the visualization of the results to be consistent with the uncertainty in the land-use projections. This resulted in a loss of detail. Species habitat maps also introduce additional uncertainty, and previous GAP modelling efforts had accuracy rates of 65%–80%. The effects of uncertainty of habitat maps on the overall pattern of our results may however be limited, due to aggregation of the potential habitat maps from 30 m to 1,500 km^2.

Data limitations necessitated a number of assumptions in the analysis. Firstly, we assume that the species richness index at a certain location does not vary over time if the land use remains the same. However, species' ranges could shift over time due to drivers other than land-use change, most notably due to climate change. Future research may want to incorporate both land-use change and climate change as drivers of biodiversity changes (Titeux et al., 2016). The temporal scope of this study is short, so no large changes due to climate change are expected. Secondly, the method applied in this study is based on the assumption that if land use changes in a certain location, changes in species richness index will reflect the difference in richness of the two land uses in that location. By using land use as the determinant factor for species richness, issues of landscape heterogeneity and connectivity are not included. However, landscape configuration is often noted as an important determinant of regional species richness (Dale et al., 2000; Dauber et al., 2003). In addition to land use and landscape configuration, other factors such as hydrology or human disturbance, are expected to affect the presence or absence of species. These factors were not explicitly modelled in this analysis but were considered implicitly, as they are expected to influence local spatial variation in species richness index values. Furthermore, these factors are assumed to stay stable over time between 2010 and 2030. Finally, the species-specific time lag, for instance due to regeneration or migration, necessary for the area to actually reach that potential species richness, is disregarded in this approach. This is a disadvantage of using potential rather than actual species richness. However, the GAP database provides uniform biodiversity data in the form of potential habitat maps for a large geographical area and allows us to include a large number of species in the analysis. Field observations of species richness before and after land-use transitions would be helpful to validate the assumptions made in this study.

This study used potential species richness index as an indicator of biodiversity. Species richness alone is often considered to be insufficient as an indicator to determine priority areas of conservation because it provides no information of species' functional roles and contribution to ecosystem processes, which is necessary to reflect the concept of biodiversity (Fleishman et al., 2011; Lamb et al., 2009). Furthermore, two cells in a map showing a similar value of potential species richness may differ completely in species composition; that is while alpha diversity is similar, beta diversity may be very different. In addition, a location that shows similar potential species richness over time may in fact have undergone changes in species composition due to species turnover. This is an inherent shortcoming of using potential species richness as an indicator for biodiversity. However, by also including selected groups of species in this study (according to taxonomy, level of threat and endemism), we are able to provide some information on which species groups may be impacted in which areas. In areas where biodiversity shifts due to wood pellets are projected, future studies should assess actual species richness and habitat suitability at a higher resolution. Some projected changes in SRI may partly reflect the assumptions about heterogeneity within land cover classes in the GAP data. For example, the low species richness in pine plantations reflects the assumption that all pine plantations that are identifiable with remotely sensed imagery are intensively managed, as many unmanaged plantations would have mixed pine-hardwood canopies. In reality, pine plantations in the region have a range of management intensities. Unfortunately, we were unable to include the impacts of intensification of existing management plantations in our analysis, because projections of intensification of existing plantations for the study region were not available, and the GAP habitat suitability models do not distinguish between different management intensities in pine plantations. Forest management can have either positive effects, for instance due to thinning (Gottlieb et al., 2017; Verschuyl et al., 2011), or neutral or negative impacts, for instance due to residue removal (Fritts et al., 2016; Gottlieb et al., 2017; Grodsky, Moorman, Fritts, Castleberry, & Wigley, 2016; Grodsky, Moorman, Fritts, & Hazel, et al., 2016; Riffell et al., 2011b) on species richness. Therefore, impacts on species richness due to pine plantation establishment may be overestimated if new plantations are not managed intensively. On the other hand, urban areas had a relatively high species richness compared to the other anthropogenic land-use classes, which may be explained by the fact that the land-use type "urban" includes the GAP land-use types "developed: low intensity" and "developed: open space," which includes suburban areas that contain a variety of vegetative communities and structures, including very small patches of forest. "Nonforest vegetation" had the largest range of SRI, a reflection of the land-use type's heterogeneous composition which includes nonforest and nonanthropogenic land uses such as marshes, dunes and grasslands. This means that a land-use transition involving nonforest vegetation can

result in very different changes in SRI, depending on the location.

The spatially explicit method developed and presented in this study provides a way to combine available spatial data sets of land-use projections and biodiversity. The method is applicable to different regions and for different drivers of land-use or biodiversity changes. Because of the uncertainty and aggregation of data in this study, it is not possible to provide solid answers about the exact magnitude and location of potential biodiversity impacts related to developments in the wood pellet market in the south-eastern United States. More data and research are therefore required to serve as a basis for policy development. However, our study provides trends in biodiversity related to increased wood pellet demand. These trends do not indicate strong additional biodiversity impacts compared to the impact of other drivers, apart from a small number of areas where higher changes in potential species richness were projected, including the coastal area of Virginia and the southern tip of Florida. These results provide information that allows policymakers, industry and NGOs to identify areas of concern where appropriate mitigation measures may need to be taken to limit negative biodiversity impacts, and areas where positive impacts can be promoted. Positive impacts could be promoted by stimulating land-use transitions that may increase biodiversity in particular areas such as the southern tip of Florida. Avoiding negative impacts could be achieved through restrictions of conversion of lowland hardwood forest, particularly in high biodiversity areas such as the Coastal Plains. Our results show that it is not straightforward to determine whether wood pellet production is good or bad news for biodiversity. Biodiversity impacts are location-specific, and different taxonomic groups may show different responses. Because biodiversity impacts are spatially variable, it will be vital to monitor the origin of wood pellets that are being produced in the south-eastern United States in order to quantify wood pellet-related biodiversity impacts. Shifts in species richness index values due to wood pellet demand were considerably smaller than biodiversity impacts caused by trends in other sectors. Therefore, biodiversity impacts due to wood pellet demand should be considered in the context of other drivers of land-use change and biodiversity loss.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

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