

Projected gains and losses of wildlife habitat from bioenergy-induced landscape change

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Abstract

Domestic and foreign renewable energy targets and financial incentives have increased demand for woody biomass and bioenergy in the southeastern United States. This demand is expected to be met through purpose-grown agricultural bioenergy crops, short-rotation tree plantations, thinning and harvest of planted and natural forests, and forest harvest residues. With results from a forest economics model, spatially explicit state-and-transition simulation models, and species–habitat models, we projected change in habitat amount for 16 wildlife species caused by meeting a renewable fuel target and expected demand for wood pellets in North Carolina, USA. We projected changes over 40 years under a baseline ‘business-as-usual’ scenario without bioenergy production and five scenarios with unique feedstock portfolios. Bioenergy demand had potential to influence trends in habitat availability for some species in our study area. We found variation in impacts among species, and no scenario was the ‘best’ or ‘worst’ across all species. Our models projected that shrub-associated species would gain habitat under some scenarios because of increases in the amount of regenerating forests on the landscape, while species restricted to mature forests would lose habitat. Some forest species could also lose habitat from the conversion of forests on marginal soils to purpose-grown feedstocks. The conversion of agricultural lands on marginal soils to purpose-grown feedstocks increased habitat losses for one species with strong associations with pasture, which is being lost to urbanization in our study region. Our results indicate that landscape-scale impacts on wildlife habitat will vary among species and depend upon the bioenergy feedstock portfolio. Therefore, decisions about bioenergy and wildlife will likely involve trade-offs among wildlife species, and the choice of focal species is likely to affect the results of landscape-scale assessments. We offer general principals to consider when crafting lists of focal species for bioenergy impact assessments at the landscape scale.

Keywords: biodiversity, bioenergy target, biofuel, habitat, landscape change modeling, renewable energy, southeastern United States, wildlife, wood pellets

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Introduction

In the United States and Europe, government policies encourage the use of bioenergy as an alternative to fossil fuels. In the United States, the Energy Policy Act of 2005 and the Energy Independence and Security Act of 2007 set nationwide targets for a renewable fuel standard whose goals include replacing some dependence on fossil fuels through bioenergy production and reducing greenhouse gas (GHG) emissions (Energy Policy Act, 2005; Energy Independence and Security Act,

2007). The European Union set renewable energy targets that are expected to be met, in part, by harvesting forests in the southeastern United States to produce wood pellets (Abt *et al.*, 2014; Wang *et al.*, 2015; Galik & Abt, 2015). Besides alleviating fossil fuel usage and GHG emissions, bioenergy production can benefit rural, agricultural economies by diversifying markets, increasing demand, and supplementing profits for traditional crops (Dale *et al.*, 2010) or so-called purpose-grown bioenergy crops, such as switchgrass, sorghum, and short-rotation woody crops (SRWCs) composed of pine (*Pinus* spp.) or poplar (*Populus* spp.) (Coleman & Stanturf, 2006).

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As renewable fuel targets and incentives increase bioenergy production, landscapes and the wildlife that inhabit them will be affected through the intensification of agriculture and forest management, influences on land use change (direct and indirect), including conversion of land to forest and agriculture, and the prevention of urbanization (Groom *et al.*, 2008; Robertson *et al.*, 2008; Eggers *et al.*, 2009; McDonald *et al.*, 2009; Dauber *et al.*, 2010; Fargione, 2010; Wang *et al.*, 2015). However, the specific effects of this increase will likely vary across spatial and temporal scales, geographic regions, the wildlife species considered, feedstock types, feedstock management practices, and land use histories (Fletcher *et al.*, 2010; Wiens *et al.*, 2011; Rupp *et al.*, 2012; Immerzeel *et al.*, 2014).

The conversion of land from natural ecosystems (e.g., forests, woodlands, grasslands) to ones dominated by crop monocultures could be one of the most important ecological consequences of the increase in demand for bioenergy because such changes are associated with losses of wildlife habitat and biodiversity (Tilman *et al.*, 2002; Firbank, 2008; Dornburg *et al.*, 2010; Fletcher *et al.*, 2010; Wiens *et al.*, 2011; Rupp *et al.*, 2012; Immerzeel *et al.*, 2014). Although some crops grown for bioenergy such as *Miscanthus*, sweet sorghum, switchgrass, and short-rotation woody crops (SRWCs) provide food and/or cover for some species, the crops differ in their value for wildlife and if grown in monocultures will likely have little value to some species because of their lack of heterogeneity and structure (Fletcher *et al.*, 2010; Wiens *et al.*, 2011; Bonin & Lal, 2012). In addition, crop management practices, such as the timing of harvest relative to animals' annual and breeding cycles, could also limit their value for wildlife, as well as reduce survival and reproduction (Rupp *et al.*, 2012).

The degree to which land conversions to bioenergy crops negatively affect wildlife depends in part upon the initial land use or land cover. Whereas the conversion of many natural ecosystems to bioenergy crops would represent a loss of habitat value, there may be cases where it improves habitat (Meehan *et al.*, 2010; Wiens *et al.*, 2011; Rupp *et al.*, 2012). For example, if existing monocultures were replaced with structurally heterogeneous grasslands composed of diverse perennial plants, they could support more diverse wildlife communities and increase habitat availability for some grassland species (Rupp *et al.*, 2012; Uden *et al.*, 2015). Such benefits to wildlife would likely be lower in fields with less heterogeneity in plant species composition and structure and lower than in native grasslands, depending upon crop harvest and management practices (Hartman *et al.*, 2011; Wiens *et al.*, 2011; Robertson *et al.*, 2012). SRWC monocultures are associated with lower biodiversity than natural forest ecosystems but may have higher diversity

than agricultural croplands (Rupp *et al.*, 2012). Little is known about herpetofaunal diversity on SRWCs (Rupp *et al.*, 2012), but bird and mammal species richness and abundance in midwestern SRWC *Populus* plantations were lower than in forests and shrublands and higher than in nonhay croplands (Christian *et al.*, 1998).

In the southeastern United States, biomass could also be derived from forests through elevated rates of harvest and thinning and removal of forest harvest residues. The specific characteristics of these harvests will determine their impact on species, although an experimental study failed to detect relationships between the amount of retained woody biomass in clear-cut pine plantations and herpetofauna diversity, evenness, or richness (Fritts *et al.*, 2016) or shrew abundance (Fritts *et al.*, 2015). Decisions regarding which forest types to extract biomass from (e.g., forested uplands vs. wetlands, natural vs. planted stands) will influence the magnitude of effects on individual species (Evans *et al.*, 2013). Thinning forests can increase wildlife diversity by increasing structural complexity, but specific impacts vary across species, thinning technique, and thinning intensity (Rupp *et al.*, 2012).

Land conversions, bioenergy crop cultivation, and forest biomass harvests at individual sites may scale up to biodiversity impacts that are measurable at the landscape level. Once again, the degree and nature of such changes depend upon the initial landscape and bioenergy sources utilized. Heterogeneous landscapes support greater biodiversity (Wiens *et al.*, 2011), so the conversion of abandoned and marginal lands to crop agriculture, along with the concentration of biomass crops around processing facilities, could decrease heterogeneity on some landscapes and, therefore, negatively influence biodiversity (Fletcher *et al.*, 2010; Wiens *et al.*, 2011). Alternatively, planting perennial grasslands for bioenergy feedstocks could be an opportunity to increase the heterogeneity of landscapes that are currently dominated by agricultural crops (Wiens *et al.*, 2011). Effects of the establishment of SRWCs on landscape- and regional-scale biodiversity will vary depending on the degree to which landscapes are already forested and the configuration of SRWC stands, but even relatively small amounts of SRWC plantations in forested landscapes with little existing shrub-dominated land would enhance landscape-scale biodiversity. This effect would not necessarily be the same for landscapes dominated by grasslands or managed forests (Rupp *et al.*, 2012). Rupp *et al.* (2012) argued that forest thinning for biomass will likely increase the forest structural diversity in landscapes and consequently increase landscape-level species diversity and abundance.

The influence of biomass cultivation and harvest on wildlife habitat at large spatial and temporal scales has

rarely been demonstrated (Immerzeel *et al.*, 2014). Few long-term or landscape-scale studies of bioenergy, effects on wildlife in the southeastern United States exist other than Evans *et al.* (2013), which examined the concurrence of wildlife habitat and forest biomass harvest risk under scenarios that differed in factors such as the types of forests harvested, the spatially explicit probability of pine plantation establishment, and the intensity of biomass extraction at harvest sites. In our study, we examined the potential impacts of meeting realistic bioenergy demands on habitat availability across a landscape over a 40-year period. While previous large-scale studies examined single sources of biomass for bioenergy, we assessed impacts of bioenergy production under different portfolios of biomass sources that included land conversions to bioenergy crops and harvests of forest biomass. We accomplished this with deductive habitat modeling and results from a timber supply model that was integrated with a spatially explicit state-and-transition simulation model (Costanza *et al.*, 2016). Our results account for complex economic processes that link demand for forest biomass to landscape change, as well as other landscape change drivers, such as urbanization and vegetation dynamics. Furthermore, comparing across different scenarios allowed us to isolate some relationships between specific biomass sources or land conversions and individual species to gain insights into patterns that may emerge when bioenergy demand is applied to landscapes. Our objectives were to (i) determine whether realistic levels of bioenergy demand are large enough to translate into gains or losses in the amount of habitat for individual species at the landscape scale, (ii) assess the degree of similarity in habitat effects among diverse bioenergy portfolios, and (iii) find insightful patterns and connections between specific sources of biomass and individual species. Exploring these topics will inform future efforts to understand how bioenergy demand will impact wildlife and help policymakers craft bioenergy targets and feedstock portfolios that support sustainable wildlife populations.

Materials and methods

Study area

North Carolina (NC) is located in the southeastern United States and includes three distinct physiographic regions, each encompassing different types of land use and vegetation: the Mountains, Piedmont, and Coastal Plain. The majority of publicly owned forested lands lie in the Mountains, which are heavily forested with hardwood and mixed pine and hardwood forests. Most of the urban areas occur within the Piedmont where pine, hardwood, and mixed pine-hardwood forests form a matrix with agricultural lands. Much privately

owned agricultural use is spread over the eastern third of the state in the Coastal Plain where diverse plant communities such as bottomland hardwood forests, pine woodlands, and freshwater wetlands also occur. Landscape change due to attempts to meet future timber and agricultural demands, including demands for bioenergy, will likely occur within the Coastal Plain and Piedmont regions. Traditional silviculture that is dominated by loblolly pine (*Pinus taeda*) plantation growth and management is also common in the Coastal Plain, and much of the future urbanization is predicted to occur in the Piedmont (Terando *et al.*, 2014; Costanza *et al.*, 2015). A state goal of replacing 10% of petroleum sold in NC with locally grown and produced liquid biofuels through 2017 was chosen in 2007 (Burke *et al.*, 2007). Additionally, existing or announced wood pellet plants could consume 3.45 million green tonnes (3.8 million green short tons) of forest biomass annually (Forisk Consulting LLC, 2014). We used deductive species-habitat modeling to translate future projected landscapes into potential trends in habitat availability within NC between 2010 and 2050 for 16 wildlife species under five alternative bioenergy production scenarios.

Bioenergy scenarios and land use/land cover

In the southeastern United States, bioenergy will likely be produced from agricultural crop residues, bioenergy crops (e.g., sorghum and switchgrass), SRWC, the harvest or thinning of forests, and forest harvest residues (Rupp *et al.*, 2012; Fritts *et al.*, 2014; Wang *et al.*, 2015). We could not predict the exact portfolio of sources in NC's future, so we explored Costanza *et al.*'s (2016) five bioenergy production scenarios along with their baseline that did not include increased bioenergy production (Table 1). The bioenergy scenarios differed in how much they utilized three sources of biomass: (i) feedstock sourced from the harvest and thinning of planted pine, natural pine, hardwood, mixed pine-hardwood, upland, and bottomland forests (hereafter, 'conventional forestry'); (ii) the conversion of agricultural lands on poor soils to purpose-grown feedstocks and subsequent harvests (i.e., switchgrass, sorghum, SRWC; hereafter 'conversion of Ag lands'); and (iii) the conversion of forests on poor soils to purpose-grown feedstocks and subsequent harvests (hereafter, 'conversion of forests').

One of the bioenergy scenarios ('Conventional') assumed that 3.63 million green tonnes (4.0 million green short tons) of forest biomass would be harvested, and thus met the anticipated demand for biomass for wood pellet production (Table 1). Two other scenarios ('Conventional-Ag', and 'Conventional-Ag-Forest') also included the harvest of 3.63 million green tonnes of forest biomass for bioenergy, as well as enough purpose-grown feedstocks to meet or exceed the 10% liquid if the forest biomass was used for biofuels instead of wood pellets (Costanza *et al.*, 2016). Those two scenarios differed in the types of lands converted to purpose-grown feedstocks. Conventional-Ag assumed that only marginal agricultural lands would be converted to purpose-grown crops, whereas Conventional-Ag-Forest assumed that marginal agricultural and forest lands, in equal proportions, would be converted to purpose-grown crops. The remaining two bioenergy scenarios ('Ag' and

Table 1 Six bioenergy production scenarios modeled by Costanza *et al.* (2016) and assessed here for their impacts on species' habitats over 40 years in North Carolina, USA

Scenario name	Description	Conv. forest* (%)	Marg. Ag.† (%)	Marg. forest‡ (%)	NCRFS§ (%)	Forest biomass¶ (M gt)	Harvest residue** (%)	Ag. converted†† (ha)	Forest converted‡‡ (ha)
Baseline	Business-as-usual production of conventional forest products; no bioenergy production	0	0	0	0	0	n/a	0	0
Conventional	Increased conventional forestry for bioenergy	100	0	0	0	3.63	40	0	0
Conventional-Ag	Increased conventional forestry and conversion of agricultural lands for bioenergy	15	85	0	100–148	3.63	40	425 000	0
Conventional-Ag-Forest	Increased conventional forestry and conversion of agricultural lands and forests for bioenergy	15	42.5	42.5	100–148	3.63	40	212 500	212 500
Ag	Conversion of agricultural lands for bioenergy	0	100	0	100–140	0	n/a	504 000	0
Ag-Forest	Conversion of agricultural lands and forests for bioenergy	0	50	50	100–140	0	n/a	252 000	252 000

*Percentage of scenario derived from planted and natural forests.

†Percentage of scenario derived from conversion of agricultural lands on marginal soils to purpose-grown feedstocks.

‡Percentage of scenario derived from conversion of forests on marginal soils to purpose-grown feedstocks.

§Potential percentage of North Carolina's 10% biofuel target met (see text for description of 10% target). Ranges are presented instead of point estimates because the amount of energy that can be derived from feedstocks varies with refinement process (Costanza *et al.*, 2016).

¶Biomass from conventional forestry (in million green tonnes).

**Percentage of forest harvest residues removed.

††Amount of agricultural land converted to switchgrass, sorghum, or SRWC.

‡‡Amount of forest land converted to switchgrass, sorghum, or SRWC.

'Ag-Forest') would meet the 10% biofuel target without increased conventional forestry. Ag assumed that marginal agricultural lands would be the only lands converted to purpose-grown crops, whereas Ag-Forest incorporated conversion of equal proportions of marginal agricultural and forest lands. All five bioenergy scenarios and the baseline scenario included urbanization, natural disturbance such as fire, ecological succession, and conversion between forest and agricultural land use according to recent change trends (Wear & Greis, 2013; Costanza *et al.*, 2016).

Costanza *et al.* (2015, 2016) translated demand for bioenergy into potential changes in NC's land cover using the Sub-regional Timber Supply Model (SRTS) (Abt *et al.*, 2009) and state-and-transition simulation model (STSM) software ST-Sim (ApexRMS, 2014). The SRTS model produced aspatial time series of areal amounts of forest thinning and harvest, as well as conversion between agricultural and forest land uses. For the three scenarios without biomass from conventional forests (including Baseline), the SRTS model incorporated demand for broad forest types and age classes according to recent trends in USDA Forest Service Forest Inventory and Analysis (FIA) data. For the three scenarios that included conventional forest biomass, the SRTS model projected forest harvest and thinning amounts that would result in 3.63 million green tonnes of

biomass extracted from forests, while accounting for demand displacement of other forest products (Costanza *et al.*, 2015).

The spatial STSM incorporated time series outputs of forest thinning, harvest, and land use conversion from SRTS and aspatial state-and-transition model pathways for each of 52 vegetation and land use classes. It also included several spatial inputs: maps of initial vegetation, land use, successional stages, and vegetation structure; an existing time series of future urbanization probability (Terando *et al.*, 2014); marginal agricultural and forest lands to be converted to purpose-grown crops for bioenergy; and lands under conservation ownership to be excluded from biomass production (Costanza *et al.*, 2016). To ensure that adjacent pixels were more likely to undergo disturbance, management, and land conversion transitions together, rules were set in the STSM that governed the sizes of these events so that no transitions were smaller than 2.0 ha (approximately six pixels), which was approximately the minimum patch size in the initial vegetation and land use raster for NC (Costanza *et al.*, 2016).

Initial vegetation and land use maps and most of the aspatial model pathways are detailed in Costanza *et al.* (2015). Costanza *et al.* (2016) added model pathways corresponding to each of the three purpose-grown crops. Switchgrass and sorghum pathways each consisted of a single state class, whereas the

SRWC pathway contained three state classes: a recently harvested class, an early-succession class, and a mid-succession class. When lands were converted to SRWC, they transitioned to the early-succession class. The planted pine pathway was divided into nine state classes: one recently harvested class, three recently thinned classes, and five classes that represented various age classes of closed (not recently thinned) forest (Table S1). The accuracy of the initial vegetation and land use map has not been assessed, but some accuracy metrics have been measured for the datasets from which it was created (Costanza *et al.*, 2016). The current version of the GAP Land Cover Map has not been formally assessed, but a similar land cover mapping effort for the North Carolina Gap Analysis had per-class accuracies between 74% and 95% for general land cover classes and overall accuracy of 58% for the detailed classes based on conditional probabilities (McKerrow *et al.*, 2006). Validation of the NLCD canopy cover model with canopy cover values from field data by Coulston *et al.* (2012) showed an R^2 value of 0.8 in the southeastern United States. LANDFIRE s-class data are based on LANDFIRE vegetation height data, which have a spatial bias of 3.8%, and LANDFIRE cover data had an overall agreement of 74% when compared with field data across the U.S. (Toney *et al.*, 2012).

Costanza *et al.* (2016) identified marginal agricultural and forest lands as those that were row crop, pasture/hay, or forest in NLCD 2006, and were in nonirrigated soil capability classes 3 or 4 according to gridded Soil Survey Geographic Database (gSSURGO). They used GAP PAD-US data (U.S. Geological Survey National Gap Analysis Program, 2013) to identify lands under conservation ownership that would be ineligible for biomass production and conversion to urban or agricultural land. The STSM simulated changes in each of the mapped vegetation and land use classes due to forest management and land conversions specific to bioenergy production described above, as well as natural disturbance processes, ecological succession, urbanization, forest management, and background conversion between forest and agricultural lands.

The spatial outputs resulting from the STSMs had 60 m × 60 m cells. Each cell was assigned a land use or vegetation class. Vegetation classes were also assigned a successional stage (early, mid, or late succession) and a structure class (open or closed canopy, thinned, or recently harvested) (Table S2). Hereafter, we refer to these unique combinations of vegetation class, successional stage, and structure class as 'land cover classes'. Whereas SRTS and the STSMs were run on an annual time step, spatial outputs were produced every 10 time steps. In order to match the resolution of other model inputs, we resampled these spatial vegetation and land use outputs to 30 m × 30 m cell resolution using a nearest neighbor technique that assigned the values of larger cells to the each of the smaller, nested cells and used these finer resolution raster layers in the habitat models.

Focal species

While modeling all terrestrial vertebrate species within NC would have been ideal for our study, doing so was impractical given the effort and information required to build a habitat

model. Therefore, we carefully selected 16 species that, collectively, allowed us to meet our objectives, focusing on birds and amphibians because numerous published studies on the habitat associations of these taxa existed (Table 2). To meet our objective of determining whether bioenergy demand could translate into changes in habitat amount, given the economic and ecological processes at play on the landscape, we sought a list that included species that were, collectively, associated with each of the major systems that will be affected by bioenergy demand (e.g., forests, woodlands, and agricultural lands) and could be sensitive to the changes in successional stage and vegetative structure likely resulting from biomass harvests, such as decreases in percent canopy cover or the development of dense shrub layers. To meet our objective of exploring the diversity of responses, we sought a list of species with diverse habitat preferences. Therefore, we chose species that contributed a unique suite of land cover class associations with the list and that select habitat in part on the basis of vegetative composition or structure at the level of detail concomitant with the land cover classification and changes in forest successional stage and vegetative structure in the STSM. We presumed species which do not select habitat based on vegetative structure would be unlikely to experience changes in habitat amount from forest biomass harvests and did not consider them for inclusion. Similarly, landscape change would be less likely to manifest in changes in habitat availability for habitat generalists than for habitat specialists, so we avoided adding individual species associated with the majority of the land cover classes in the land cover map. Our list included 10 species of greatest conservation need, three species of special concern, and two species listed as threatened by NC (NCWRC, 2015; Table S4). In addition, seven are on the Partners in Flight Watch List and four are on the IUCN Red List (Rich *et al.*, 2004; IUCN, 2015; Table S4).

Habitat models

We built a deductive habitat model for each species that categorized STSM land cover classes as either inhabitable or uninhabitable in order to estimate future trends in species' habitat availability under each scenario (Table S2). Our deductive approach (habitat models) followed that of the United States Geological Survey's National Gap Analysis Program (GAP; Scott *et al.*, 1993; Corsi *et al.*, 2000; Overmars *et al.*, 2007). GAP models are based on species-habitat requirement information gleaned from scientific literature and expert opinion. They associate individual species with land cover/land use classes (Gergely & McKerrow, 2013) and incorporate ancillary parameters (e.g., elevation, distance from forest edge, and distance from water) and range data to map the distribution of individual species' habitat at a landscape scale (1 : 100 000 map scale). Whereas national GAP models are based upon the GAP National Land Cover, ours corresponded with the land cover classification used in STSM, which has a finer thematic resolution than GAP's classification due to the specification of multiple states and stages (Table S2). For several species, we added constraints on model output related to maximum elevations and distances to water features as specified by the species' GAP model (Table S5) (<http://gapanalysis.usgs.gov/>). We

Table 2 We built deductive habitat models for 16 focal species to explore bioenergy demand effects on wildlife habitat in North Carolina, USA

Common name	Scientific Name	Regions	Forest	Open Woodland	Open Grass	Dense Shrub	Urban	% Habitat
Eastern Tiger Salamander	<i>Ambystoma tigrinum</i>	P, CP	X	X				4
Gopher Frog	<i>Lithobates capito</i>	CP		X				<1
Mole Salamander	<i>Ambystoma talpoideum</i>	M, P	X	X				1
Oak Toad	<i>Anaxyrus quercicus</i>	CP		X				1
Brown-headed Nuthatch	<i>Sitta pusilla</i>	M, P, CP	X	X			X	33
Cerulean Warbler	<i>Dendroica cerulean</i>	M, CP	X					14
Field Sparrow	<i>Spizella pusilla</i>	M, P, CP				X		20
Grasshopper Sparrow	<i>Ammodramus saviannarum</i>	M, P, CP			X			13
Hairy Woodpecker	<i>Picoides villosus</i>	M, P, CP	X					54
Kentucky Warbler	<i>Oporornis formosus</i>	M, P, CP	X			X		27
Loggerhead Shrike	<i>Lanius ludovicianus</i>	P, CP		X	X			17
Prothonotary Warbler	<i>Protonotaria citrea</i>	CP	X					3
Red-headed Woodpecker	<i>Melanerpes erythrocephalus</i>	M, P, CP		X			X	12
Swainson's Warbler	<i>Limnothlypis swainsonii</i>	M, CP	X			X		19
Wood Thrush	<i>Hylocichla mustelina</i>	M, P, CP	X					41
Yellow-breasted Chat	<i>Icteria virens</i>	M, P, CP				X		16

The regions field indicates which ecoregions the species occurs in within NC: 'M' for Mountains, 'P' for Piedmont, and 'CP' for Coastal Plain. X's indicate that the species is associated with a land cover or vegetative structure type, but we caution that the associations listed here are highly generalized from the associations in the habitat models, which are described in Tables S2 and S3 and include specification of suitable successional stages and vegetative structure classes. Percent habitat refers to the proportion of the land area of North Carolina that our species-habitat models predicted as suitable for the species in 2010.

excluded some GAP model parameters because they are dependent upon habitat patch characteristics that had a high degree of uncertainty in the STSM output (e.g., patch size, whether a cell is forest edge or forest interior). We used ArcGIS 10.1 (ESRI, 2012) for all geoprocessing.

Model assumptions

Our modeling methods necessarily involved a set of assumptions about bioenergy feedstocks and habitat use patterns because some feedstocks are not yet widely present in NC or their characteristics and suitability for wildlife have not been documented. First, we assumed that forest thinning, whether under Baseline or in the bioenergy scenarios, removed 50% of biomass and therefore 50% of canopy cover. In the SRTS model, forest thinning was largely used for nonbioenergy products including sawtimber, so larger diameter trees would be included in the thinnings with and without bioenergy demand. Second, we assumed forest harvest under bioenergy scenarios removed 40% of residual biomass while retaining some snags (Costanza *et al.*, 2015). Third, Conventional-Ag, Conventional-Ag-Forest, Ag, and Ag-Forest included the cultivation of sorghum, switchgrass, and SRWC, but the extent to which wildlife will use these crops in NC has not been thoroughly studied. We assumed that none of our species will breed in sweet sorghum because they rarely breed in traditional row crops in NC; switchgrass crops will be tall, dense monocultures that may only be suitable for Yellow-breasted Chats; and our focal species would inhabit SRWC at rates similar to those in young planted pine stands (Dhondt *et al.*, 2007).

Another set of assumptions are related to species-specific complexities in habitat suitability. First, Cerulean Warblers are sensitive to the proportion of forest in landscapes where they occur but can benefit from commercial thinning within heavily forested landscapes (Wood *et al.*, 2013). We assumed thinned forests in the Mountains, which is heavily forested, would remain suitable for Cerulean Warblers but thinned forests in the Coastal Plain, which is largely agricultural, would render them unsuitable. However, we explored the sensitivity of our model to this assumption and determined that this component made little difference in the results in terms of total amount of habitat for the species. Second, Brown-headed Nuthatches and Red-headed Woodpeckers are known to inhabit some urbanized areas, but only where large trees and suitable nest cavities typically found in snags occur (Slater *et al.*, 2013). Therefore, we assumed that sites that transitioned from treeless land cover types to 'urban' remained unsuitable for 20 years and removed such sites from models. Third, Grasshopper Sparrows reportedly inhabit traditional row crops and switchgrass fields in Nebraska (Uden *et al.*, 2015). However, we did not attribute them to these classes because we assumed switchgrass fields will be too dense and homogenous for them in NC; we found no indication that they regularly breed in crops in NC; and the species exhibits regional variation in habitat preferences (Vickery, 1996).

Analysis and summary of habitat model outputs

The six scenarios (Baseline and five bioenergy portfolios) and five time steps (10-year increments from 2010 to 2050) yielded

30 separate model outputs for each of 16 focal species. We quantified the influence of each bioenergy feedstock portfolio on habitat availability for each species by calculating the differences in the amounts of habitat available between each bioenergy scenario and Baseline. To do this, we converted the number of cells in the habitat model output from each species-scenario-time step combination into hectares (Table S6) and then calculated the areal differences between individual bioenergy scenarios and baseline values at each time step for each species. We also summarized changes in habitat amount under Baseline in order to provide background references for both the amount of habitat initially available and projected trends given drivers of landscape change other than bioenergy production. However, we limit our presentation of baseline trends because they were somewhat difficult to interpret for some species due to modeled increases in new thinned and harvested land cover classes on the landscape during the first ten annual time steps of the STSM (Costanza *et al.*, 2015). Those large modeled landscape changes had the capacity to affect trends for species that use intact, thinned, or harvested forests during the same time period. We discuss this issue further in the Discussion.

Results

Our habitat models estimated that in 2010, the proportion of NC that was suitable ranged from <1% to 54% among our focal species (Table 2). For most species, between 11% and 20% of the state was suitable, but only three to four percent was suitable for the Prothonotary Warbler and Eastern Tiger Salamander, one percent for the Oak Toad and Mole Salamander, and less than one percent for the Gopher Frog. Among the species with the most habitat were Kentucky Warbler (27%), Brown-headed Nuthatch (33%), Wood Thrush (41%), and Hairy Woodpecker (54%).

Under Baseline, trends in the projected amount of habitat varied by species. Habitat was lost with each time step for the Eastern Tiger Salamander, Mole Salamander, Cerulean Warbler, Grasshopper Sparrow, Hairy Woodpecker, Swainson's Warbler, and Wood Thrush (Fig. 1). In addition, a net loss of habitat occurred by 2050 for the Kentucky Warbler, Loggerhead Shrike, and Prothonotary Warbler, but these species experienced gains and losses during the 40-year period. The Gopher Frog also experienced gains and losses during the 40-year period, but 2050 habitat amounts were similar to initial amounts. Mole and Eastern Tiger Salamanders experienced habitat losses in both upland and bottomland hardwood and conifer forests. Habitat was gained by 2050 for the Red-headed Woodpecker, Brown-headed Nuthatch, Oak Toad, Yellow-breasted Chat, and Field Sparrow (Fig. 1).

For several species, trends in habitat amount remained similar to Baseline under scenarios with bioenergy production; that is, the amount of habitat available did not deviate more than five percent from the amount

available under Baseline at any time step under any scenario (Fig. 2). Nevertheless, some patterns emerged for those species. Conventional, Conventional-Ag, and Ag were similar to Baseline for the Mole Salamander, Brown-headed Nuthatch, Eastern Tiger Salamander, Hairy Woodpecker, and Wood Thrush, but Conventional-

Ag-Forest and Ag-Forest, which included the conversion of marginal forests, offered less habitat than Baseline for those species. Cerulean Warbler habitat amounts were most similar to Baseline under scenarios without increased conventional forestry (Ag and Ag-Forest) and declined over the 40-year period in scenarios with increased conventional forestry. However, those declines were never more than 4% below Baseline. Kentucky Warbler habitat amounts were most similar to Baseline under the scenario without conversion of agricultural lands or forests (Conventional) but alternated gains and losses of up to 5% from Baseline in scenarios with conversion of agricultural lands or marginal forests. Habitat for Swainson's Warbler also varied among time steps under each bioenergy scenario, but never deviated more than 4% from Baseline.

Among species that experienced >5% deviation from Baseline under a bioenergy scenario, the trends and magnitudes of differences varied among species, scenarios, and time steps (Fig. 2). The Gopher Frog, Grasshopper Sparrow, Loggerhead Shrike, and Yellow-breasted Chat experienced the greatest differences from Baseline with Gopher Frog habitat falling 20–25% below Baseline by 2050 under scenarios that included increases in conventional forestry. Grasshopper Sparrow habitat area was almost 20% below Baseline and Loggerhead Shrike habitat area was 12% below Baseline under Ag, in which all bioenergy was derived from the conversion of agricultural lands to purpose-grown feedstocks. The Yellow-breasted Chat gained 14% more habitat by 2040 under scenarios that included added conventional forestry and conversion of agricultural lands to bioenergy feedstocks.

Several noteworthy relationships between specific bioenergy feedstocks and species' habitat were identifiable. First, for a few species, the trends in habitat availability under scenarios with increased conventional forestry differed from trends under the other scenarios. For the Oak Toad, those scenarios were better than all other scenarios before 2040, although in 2050, scenarios without added conventional forestry (Ag and Ag-Forest) were also better than Baseline (4% and <1%, respectively). For the Gopher Frog, which has similar habitat associations with the Oak Toad except that it is not associated with hardwood forests (Table 2), increasing conventional forestry *decreased* the habitat amount to as much as 25% below Baseline in 2050 (25% under

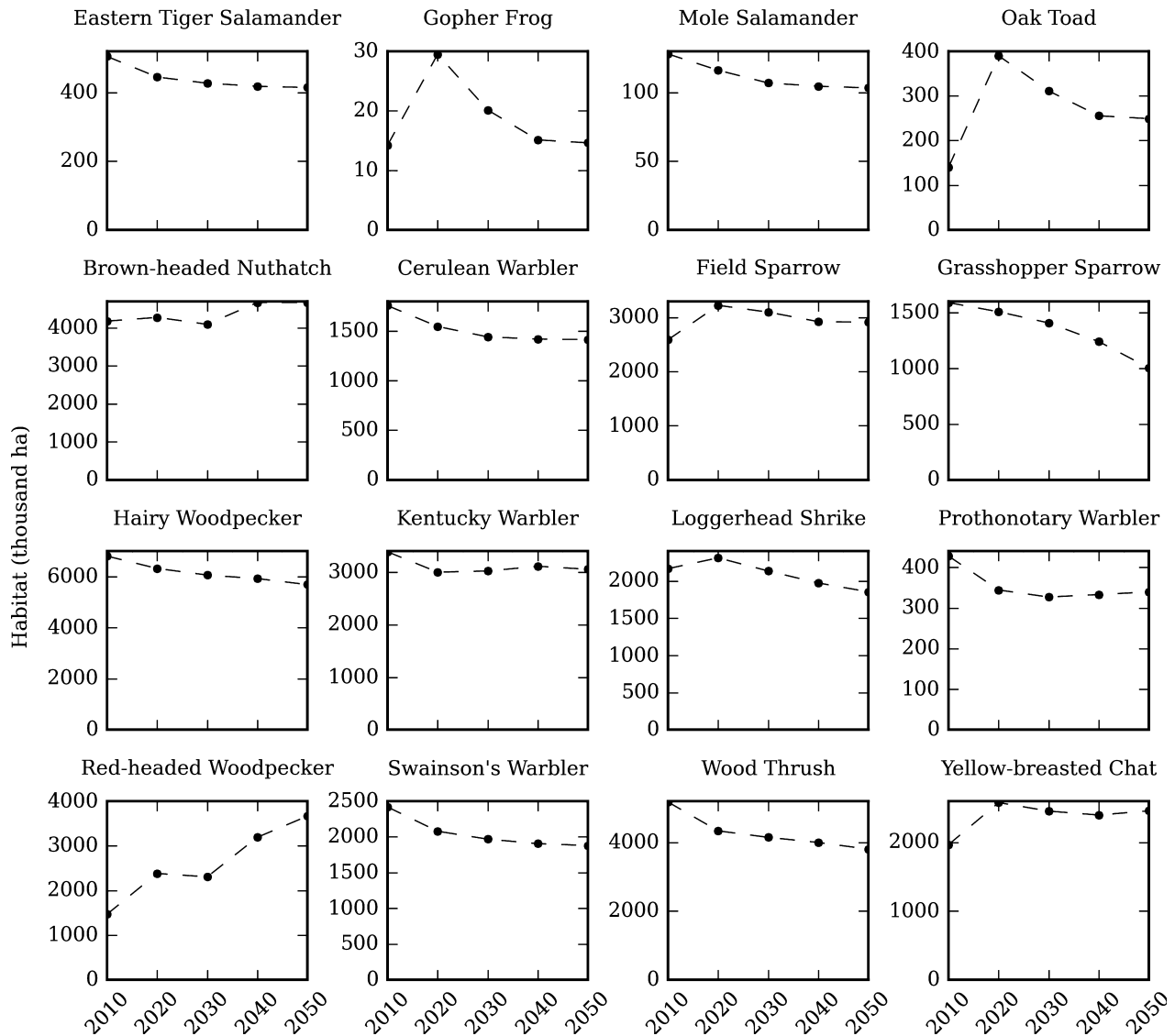


Fig. 1 Projected area of habitat in North Carolina, USA, under a ‘business-as-usual’ scenario (Baseline) that included urbanization, forestry, and vegetation dynamics (Costanza *et al.*, 2016).

Conventional-Ag-Forest; 23% under Conventional; 20% under Conventional-Ag). For the Prothonotary Warbler and the Red-headed Woodpecker, scenarios that included increased conventional forestry caused greater deviation from Baseline while remaining similar to each other, whereas habitat amounts under the other scenarios (Ag and Ag-Forest) were very similar to Baseline. However, Prothonotary Warbler habitat decreased to almost 10% below Baseline, whereas Red-headed Woodpecker habitat *increased* to around 5% above Baseline by 2030 and then decreased to 2–3% above Baseline by 2050 under Conventional, Conventional-Ag, and Conventional-Ag-Forest. Scenarios that included increased conventional forestry were better for the Field Sparrow than scenarios that did not, and converting marginal

forests to bioenergy feedstocks also increased the amount of its habitat. Conventional provided a steady gain in habitat relative to Baseline, but habitat amounts under the other scenarios varied among time steps; Conventional-Ag-Forest was consistently better than Baseline with up to 8% more habitat, whereas Ag was consistently worse with 8% less habitat in 2030.

Second, two species clearly responded negatively to the conversion of Ag lands to purpose-grown feedstocks: the Grasshopper Sparrow and the Loggerhead Shrike. For the Grasshopper Sparrow, sourcing biomass for wood pellets (Conventional) did not cause added habitat loss, but the conversion of agricultural lands did; ranking scenarios based on the proportion of bioenergy derived from agricultural land conversion (i.e.,

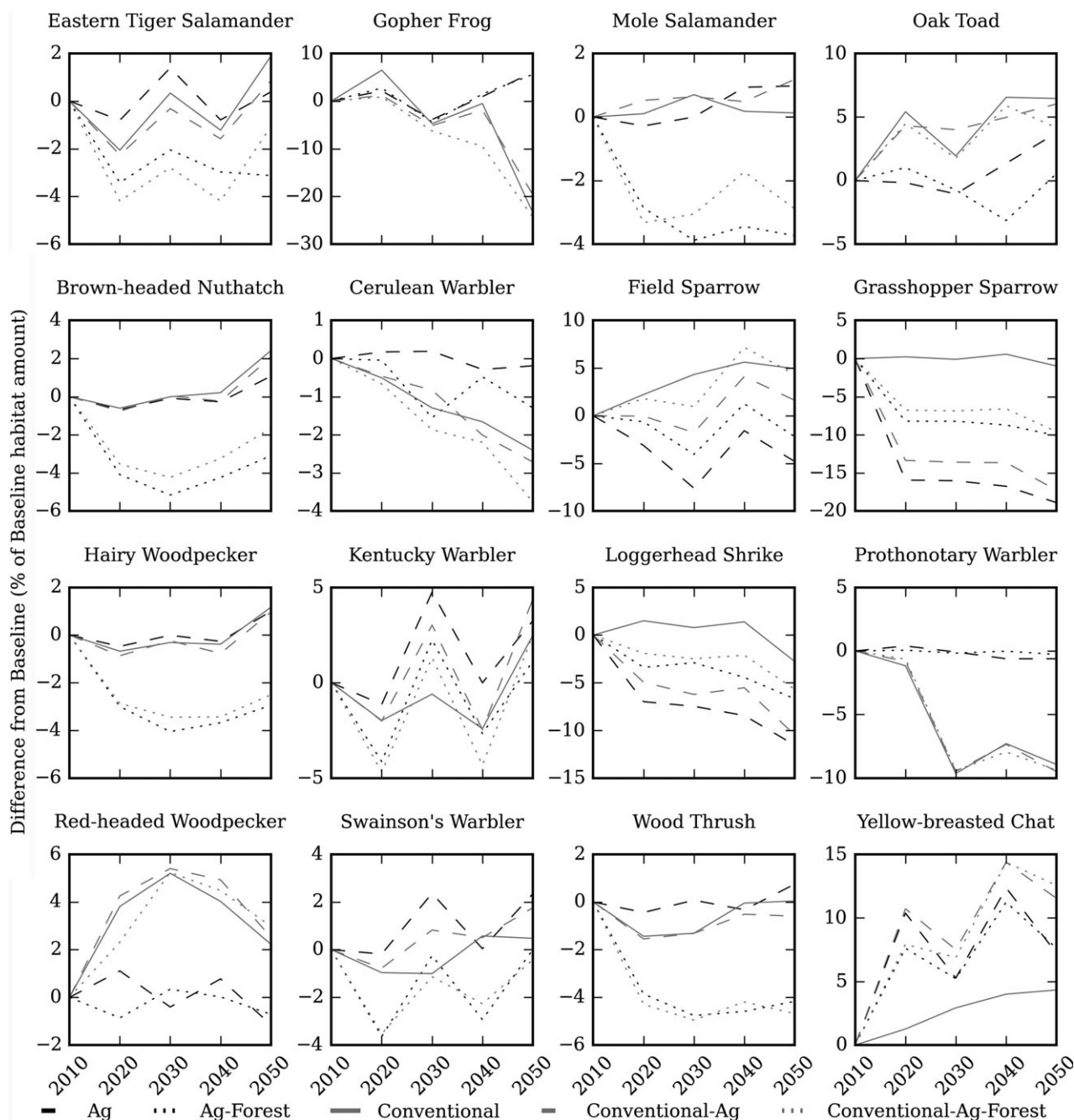


Fig. 2 Trends in habitat amount relative to Baseline for 16 species and five bioenergy production scenarios. Habitat amounts are presented as percent differences from Baseline amounts, calculated at each time step. For example, the models projected that in 2030 there would be almost ten percent less habitat for the Prothonotary Warbler under Conventional than under Baseline.

Conventional, Conventional-Ag-Forest, Ag-Forest, Conventional-Ag, Ag; Table 1) corresponded to their ranking in terms of habitat loss (Fig. 1; Conventional-Ag-Forest and Ag-Forest between 5% and 10%; Ag and Conventional-Ag between 13% and 19%). Patterns in the differences among scenarios for the Loggerhead Shrike were similar to the Grasshopper Sparrow, but

mutated; the worst scenario, Ag, was 12% below Baseline by 2050.

Finally, all bioenergy scenarios provided more habitat for the Yellow-breasted Chat than Baseline. This species experienced a steady increase relative to Baseline over the 40-year period with increased conventional forestry to meet wood pellet demand and was

4% greater than the Baseline in 2050. Under scenarios capable of meeting the biofuel target (Conventional-Ag, Conventional-Ag-Forest, Ag, and Ag-Forest), habitat amounts were between 7% and 13% greater than Baseline in 2050.

Discussion

We projected future changes in habitat availability for 16 species under alternative scenarios of bioenergy production and found that realistic levels of demand for biofuels and wood pellets in NC would be capable of causing considerable changes in the amount of habitat available for some species. These changes could be positive or negative, depending upon a species' habitat associations and the sources of biomass utilized to meet demand. Several patterns emerged that will inform future research and policy regarding bioenergy and wildlife.

Species that inhabit dense, shrubby vegetation gained habitat in some bioenergy scenarios because forest biomass harvests (clear-cutting and thinning) encouraged development of shrubby vegetative structure (shrubs and saplings), and conversions of forests on marginal land to purpose-grown crops converted unsuitable forests to dense bioenergy crops that we assumed would be suitable for shrubland species. For example, the Field Sparrow experienced habitat gains when conventional forestry was increased, and the Yellow-breasted Chat gained habitat by 2050 under each of the bioenergy scenarios. Chats are associated with open, shrubby areas characteristic of regenerating and thinned forests (Ecklerle & Thompson, 2001), and they were associated with SRWC and switchgrass in our models. Furthermore, it was unclear whether future chat habitat would be gained or lost compared with 2010 under Baseline because of their sensitivity to modeled landscape changes described below in *uncertainty and limitations*, but the four scenarios that included the conversion of marginal agricultural lands compensated for baseline losses between 2020 and 2040. This suggests that bioenergy feedstock harvest and cultivation could compensate for habitat losses for species that inhabit regenerating forests.

Clear-cutting and thinning natural and planted forests for bioenergy, such as to meet the foreign demand for wood pellets, would result in gains for some species and losses for others. Our results suggest that species associated with clear-cut, thinned, or regenerating forests could benefit from increases in forest biomass harvest within NC. Red-headed Woodpeckers are associated with woodlands and open areas containing snags, which we assumed would exist in harvested forests, so they benefited from increased thinning and

clear-cuts. Likewise, the Oak Toad benefited from thinning closed canopy forests. Interestingly, Gopher Frog habitat, which consists of mostly open longleaf pine communities, initially increased relative to Baseline with additional forest thinning, but decreased when suitable forest types transitioned from an open to closed canopy state during the final time step. This highlights the fact that forest biomass harvests result in somewhat temporary changes because of vegetative successional processes, and, therefore, repeated harvests may be necessary to sustain forests in a state beneficial to species preferring open canopies and regenerating forests.

Increased conventional forestry resulted in habitat losses for some mature forest species including Prothonotary, Swainson's, and Cerulean Warblers. These species are associated with bottomland hardwood forests, which are important habitats because they harbor high levels of biodiversity (Mitchell *et al.*, 2009). Prothonotary Warbler habitat decreased under scenarios that included the harvest and thinning of these forest types (increased conventional forestry). Similarly, Evans *et al.* (2013) found that large amounts of forested wetlands are likely to be harvested in NC's Coastal Plain and among their indicator species, the Prothonotary Warbler had the greatest relative risk of habitat changes due to biomass harvest for wood pellets. There was also a large amount of overlap between Swainson's Warbler habitat and areas of high harvest risk when forested wetland harvests were allowed in their models, but we found little difference from Baseline for the Swainson's Warbler under any bioenergy scenario. Our models indicated a response similar to Prothonotary Warbler's for the Cerulean Warbler, but the magnitude of changes relative to Baseline was smaller. However, this muted response may have been a result of the Cerulean Warbler's distribution within NC. The species mostly occurs in the Mountains, with a small, isolated population inhabiting forests along the Roanoke River in the Coastal Plain (Fig. S1). Any changes for the coastal population were likely diluted by habitat dynamics occurring in the Mountains. Had our area of interest for this study been limited to the Coastal Plain, results for the Cerulean Warbler may have been different. The case of these bottomland hardwood-associated species suggests a need for further research and monitoring of how species depend upon them and underscores the need to identify important vegetative communities and understand their value to species when crafting policy that could result in changes to the landscapes where they occur.

Converting pasture and croplands on poor soils to purpose-grown bioenergy feedstocks exacerbated habitat loss for two pasture-associated species, the Grasshopper Sparrow and Loggerhead Shrike. These

results support concerns about turning to marginal agricultural lands for bioenergy feedstock cultivation (Meehan *et al.*, 2010; Wiens *et al.*, 2011). Grasshopper Sparrow and Loggerhead Shrike habitats, which are characterized by open, grassy conditions (Vickery, 1996; Yosef, 1996), were lost under all scenarios that included the conversion of marginal agricultural lands to purpose-grown feedstocks. That is, the greater the proportion of bioenergy that came from conversion of marginal agricultural lands, the worse the scenario was for these species. This result was specifically associated with the conversion of pasture located on marginal soils. Our analysis indicates a clear relationship where increasing the cultivation of marginal lands for bioenergy would decrease the amount of habitat for grassland-associated species. Such a decrease is troubling because, in NC, pasture is already expected to be lost with urbanization, a process which was evident in loss of habitat under our 'business-as-usual' scenario. Furthermore, grassland birds have undergone the steepest national population decline of any group of birds in the last 40 years (Wiens *et al.*, 2011; Rupp *et al.*, 2012), and species associated with open vegetation may be most vulnerable to future land use change (Martinuzzi *et al.*, 2015). The importance of two drivers of landscape change for the Grasshopper Sparrow in our study indicates that considering bioenergy impacts within the context of other drivers of landscape change, rather than alone, will improve the accuracy of assessments of their impacts on biodiversity.

The conversion of forests located on poor soils to purpose-grown bioenergy feedstocks, including SRWC, was beneficial to the Yellow-breasted Chat, Field Sparrow, and Kentucky Warbler which will presumably inhabit dense, shrubby growth, and/or low canopy cover that will likely exist in SRWC stands. Conversely, converting forests resulted in habitat loss for forest species, such as the Eastern Tiger Salamander, Mole Salamander, Brown-headed Nuthatch, Hairy Woodpecker, Wood Thrush, and Swainson's Warbler. The diverse responses of species to land conversions in our study echo a point that others have made that impacts of land conversions on wildlife depend upon the initial types of land cover involved in the conversions (Meehan *et al.*, 2010; Wiens *et al.*, 2011; Rupp *et al.*, 2012). Our results suggest that the choice of which species to examine deserve to be added to the list of factors that influence whether land conversions for bioenergy will be deemed 'good' or 'bad'.

By virtue of numbers, species with small ranges, such as rare and range restricted species, are more sensitive to small changes in habitat acreage than common species with large ranges. We estimated that <1% of NC was suitable in 2010 for the Gopher Frog. This species

has undergone large range-wide population declines (Jensen & Richter, 2005) and is listed as near threatened by the IUCN and a species of special concern within NC (Hammerson & Jensen, 2004; North Carolina Wildlife Resources Commission, 2015). The overall losses that emerged suggest increased conventional forestry within its range would challenge conservation of this species. However, the initial increase in habitat over Baseline from additional conventional forestry suggests that forest biomass harvests for bioenergy could benefit the species if closed canopy forests were thinned in ways appropriate for it (Wilson, 1995). Little of the state was potentially suitable for the Prothonotary Warbler in 2010 (3%), and whereas its association with mature bottomland hardwood forests is very different than the Gopher Frog's association with open forests and woodlands on sandy soils, habitat amount for this species also declined (10%) under scenarios with increased conventional forestry. Other species for which habitat was less abundant in 2010 could fare better; the Oak Toad could gain habitat relative to baseline projections by 2050 under all bioenergy scenarios. Mole and Eastern Tiger Salamanders could gain habitat by 2050 under scenarios with increased conventional forestry. These patterns underscore that consideration of the spatial, temporal, and vegetative characteristics (i.e., successional stage and vegetative structure) of landscape changes due to bioenergy demand would benefit conservation assessments of populations of rare species because they have the capacity to emerge as important influences on habitat amounts. Our results suggest that bioenergy policy guidelines should consider the spatial coincidence of biomass production and species that are rare or have small ranges.

Uncertainty and limitations

Some aspects of future bioenergy demand, biomass source characteristics, and species' habitat use were uncertain in NC, even for the well-described species that we examined. We, therefore, had to make some assumptions in our modeling. The utility of novel bioenergy crops, thinned forests, and harvested forests to wildlife cannot be accurately modeled because their composition and structure will be dictated by cultivation and harvest practices that have not yet been fully developed and described. We, therefore, associated species with some land cover classes under assumptions of those classes' characteristics. If those assumptions prove inaccurate, then the accuracy of our projections will suffer. For example, if all snags are generally removed from clear-cuts, then Red-headed Woodpeckers will not gain as much habitat from increased conventional forestry as we indicated. Similarly, if switchgrass crops in

NC turn out to have vegetative composition or structure that is unsuitable for the Yellow-breasted Chats, then the species may not gain as much from bioenergy production as we projected. In light of these necessary assumptions, we caution against applying our results to questions other than those that we have addressed, or treating them as predictions. Future work to thoroughly describe and document the characteristics of bioenergy crops and the wildlife species that inhabit them will greatly benefit wildlife impact assessments.

Our study assessed general changes to the landscape, but the importance of finer-scale habitat factors should not be underestimated. For wildlife, the value of a site depends upon site, patch, and landscape characteristics; local management practices; and a myriad of other 'natural' influences on individual survival and reproduction not captured in our models. These factors could render patches of suitable land cover only marginally suitable or ecological traps (Schlaepfer *et al.*, 2002). Similarly, creation of habitat via bioenergy crops and forest harvests will depend upon patch characteristics (e.g., field or forest patch size and shape), management practices, and landscape composition (Robertson *et al.*, 2011, 2013; Blank *et al.*, 2014). In forests, influences on survival and reproduction could include predation on adults, juveniles, and nests; nest parasitism; and prey densities (Wilcove, 1985; Streby *et al.*, 2011; Zitske *et al.*, 2011). In crops, management practices and field characteristics are still uncertain (Rupp *et al.*, 2012; Blank *et al.*, 2014; Immerzeel *et al.*, 2014), but management practices including tillage; crop residue harvest intensities; and the intensity, frequency, and timing of harvest will be important (Rupp *et al.*, 2012). Furthermore, without plant diversity and heterogeneity within fields, bioenergy crops could be of limited value to wildlife (Bonin & Lal, 2012). While we knew that patch and landscape characteristics influence whether sites are used by some of our focal species, we could not include these parameters in our models because predicting the sizes, shapes, and configurations of future habitat patches was not possible in our modeling framework. Understanding site-scale relationships between feedstocks and habitat quality better, as well as increasing the ability to predict future patch characteristics and landscape trends would improve our ability to understand the impacts of bioenergy on habitat (Immerzeel *et al.*, 2014).

Our baseline projections of habitat availability changed more during the first time step than for subsequent time steps for some species (Fig. 1). For example, our model for the Gopher Frog projected large initial gains (120% by 2020) that were lost by 2050, with the amount of habitat in 2050 similar to 2010. We suspect that this was in part an amplification of landscape changes in the first few time steps of the STSM during which new

thinned and harvested land cover classes increased on the landscape (Costanza *et al.*, 2015) in combination with the species' distribution and the aging of recently harvested longleaf pine woodlands in the modeled landscape. These modeled landscape changes had the capacity to affect baseline trends for our species that use intact, thinned, or harvested forests and may have amplified early losses under Baseline for the Mole Salamander, Hairy Woodpecker, Eastern Tiger Salamander, Cerulean Warbler, Prothonotary Warbler, Swainson's Warbler, and Wood Thrush. There may have been a greater effect on gains and losses by 2050 for other species. For example, habitat was more abundant in 2050 than 2010 under Baseline for the Oak Toad, Gopher Frog, Field Sparrow, and Yellow-breasted Chat, but this overall gain in 2050 may have existed because of the large increase from 2010 to 2020 for those species (Fig. 1).

General insights for research, conservation, and policy

Biomass harvests and land conversions have the capacity to add or remove large amounts of habitat from a landscape under realistic levels of demand for bioenergy. Such changes will occur against a backdrop of other drivers of landscape change, and there may be cases where bioenergy will amplify or mediate changes in habitat amount from other change agents. Future increases in bioenergy production will likely have a variety of implications for wildlife because of the diversity of species' habitat requirements and biomass sources, along with the importance of how biomass harvests and cultivation will overlap with species' distributions. Each species that we examined experienced some variation among the bioenergy feedstock portfolios that we explored, but one species benefitted under all bioenergy scenarios. None of the scenarios that we explored were 'best' or 'worst' for all of our focal species, and the impacts of scenarios varied over time for some.

Our models indicated that there will be both 'winners' and 'losers' under a given bioenergy portfolio, and how a species fares may vary over time. Consequently, bioenergy policies will likely involve trade-offs for wildlife, and choices about which species to include in assessments of wildlife impacts will influence the assessments' conclusions. When evaluating the sustainability implications of bioenergy production, researchers and policymakers should therefore carefully consider whether to focus on all species, or subgroups of species occurring in an area of interest. If subgroups are examined, then the groups should be carefully defined to match the objectives. We present four general principles to consider when undertaking such a task, although there are likely others. One, species that inhabit regenerating forests may benefit from bioenergy demand. Two, species that

are largely dependent upon the mature state of a single land cover type that could be harvested for biomass are at risk to be negatively impacted by bioenergy demand. Three, bioenergy demand could exacerbate habitat loss for species that primarily inhabit land cover types that are also being lost to other drivers of landscape change (e.g., urbanization). Four, species with small ranges can be more sensitive to landscape changes from bioenergy and deserve special consideration.

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Supporting Information

Additional Supporting Information may be found online in the supporting information tab for this article:

Table S1. Descriptions of the planted pine vegetation classes ('planted' or 'public') used in our vegetation and species-habitat models.

Table S2. Species-land cover class associations used to model available habitat for 16 wildlife species in NC, USA.

Table S3. A general representation of species-land cover class associations.

Table S4. The conservation listing status of focal species.

Table S5. Ancillary parameters used in species models.

Table S6. The predicted area (thousand ha) of suitable land cover within NC, USA for species from 2010 until 2050 under Costanza *et al.*'s (2016) bioenergy production scenarios.

Table S7. Definitions of species codes used in Tables S2, S3, and S5.

Figure S1. The predicted distribution of Cerulean Warbler habitat within NC, USA in 2010.

Figure S2. The predicted distribution of Eastern Tiger Salamander habitat within NC, USA in 2010.

Figure S3. The predicted distribution of Gopher Frog habitat within NC, USA in 2010.

Figure S4. The predicted distribution of Mole Salamander habitat within NC, USA in 2010.

Figure S5. The predicted distribution of Oak Toad habitat within NC, USA in 2010.

Figure S6. The predicted distribution of Brown-headed Nuthatch habitat within NC, USA in 2010.

Figure S7. The predicted distribution of Field Sparrow habitat within NC, USA in 2010.

Figure S8. The predicted distribution of Grasshopper Sparrow habitat within NC, USA in 2010.

Figure S9. The predicted distribution of Hairy Woodpecker habitat within NC, USA in 2010.

Figure S10. The predicted distribution of Kentucky Warbler habitat within NC, USA in 2010.

Figure S11. The predicted distribution of Loggerhead Shrike habitat within NC, USA in 2010.

Figure S12. The predicted distribution of Prothonotary Warbler habitat within NC, USA in 2010.

Figure S13. The predicted distribution of Red-headed Woodpecker habitat within NC, USA in 2010.

Figure S14. The predicted distribution of Swainson's Warbler habitat within NC, USA in 2010.

Figure S15. The predicted distribution of Wood Thrush habitat within NC, USA in 2010.

Figure S16. The predicted distribution of Yellow-breasted Chat habitat within NC, USA in 2010.

Figure S17. Projected trends in habitat availability in NC, USA from 2010 to 2050 under a baseline scenario and five bioenergy scenarios.

Appendix S1. Sources used during the literature review of species' habitat associations.

Appendix S2. Outline of the geoprocessing steps of habitat map production.