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ANALYSIS

Quantifying the impacts on biodiversity of policies for carbon sequestration in forests

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Abstract

There is currently a great deal of interest in the use of afforestation (conversion of non-forest land to forest) to reduce atmospheric concentrations of carbon dioxide. To date, economic analyses have focused on the costs of forest carbon sequestration policies related to foregone profits from agricultural production. No studies have examined additional costs or benefits associated with impacts on biodiversity. The main objective of this paper is to estimate the changes in farmland and forest bird populations that are likely to occur under an afforestation policy. Econometric models of land use are used to simulate the response of private landowners to subsidies for tree planting on agricultural land. We evaluate subsidies that achieve conversion of 10% of the total agricultural land in each of three U.S. states (South Carolina, Maine, and southern Wisconsin). Bird density estimates are derived for 615 species with data from the national Breeding Bird Survey. Percentage changes in agricultural and forest land for each county are applied to county-level estimates of bird densities for farmland and forest birds. Despite considerable spatial variation in agricultural land conversion rates and farmland bird distributions within these states, statewide losses of farmland birds were relatively uniform at 10.8–12.2%. Increases in forest bird populations, however, varied substantially between states: 0.3% in Maine, 2.5% in South Carolina, and 21.8% in southern Wisconsin. Surprisingly, a net loss in total bird populations results in all three states (−2.0% in Maine, −2.3% in South Carolina, and −1.1% in southern Wisconsin), despite the prevailing wisdom as to bird-rich forests. The loss is due to the coincidence of centers of high farmland bird richness and low forest bird richness with areas economically suited to conversion. Additional gains in forest species may result, however, if afforestation within the economically optimal counties is concentrated to fill in existing forest fragments presently suffering avian losses to edge predators. Our results thus show that assessments of the biological consequences of afforestation for carbon sequestration must consider both current land cover and the distributional patterns of organisms as well as the policy's conversion goal.

Keywords: Carbon sequestration; Avian abundance; Econometric models; Wildlife models; Land-use change

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1. Introduction

The Kyoto Protocol to the Framework Convention on Climate Change, adopted by a majority of the world's nations in December, 1997, sets specific targets and timetables for the reduction of greenhouse gas emissions by Annex I (industrialized) countries. There is currently a great deal of interest in converting non-forest to forest land (afforestation) to offset carbon dioxide (CO_2) emissions. Trees and other forest vegetation photosynthesize CO_2 to yield carbon, and since forests generally store more carbon than land in other uses (e.g. agriculture), afforestation can achieve a reduction in net greenhouse gas emissions. Article 3.3 of the Protocol states that carbon sequestered as the result of human-induced afforestation, reforestation, and deforestation is to be included in the emissions inventory used to determine a nation's compliance with its treaty obligations.

The decision to pursue an afforestation strategy depends, in part, on the costs of afforestation relative to costs of alternative approaches such as improving energy efficiency, switching to cleaner fuels, as well as other methods of carbon sequestration (National Academy of Sciences, 1992; Holdren and Lee, 1999). A number of authors have estimated the marginal costs of sequestering carbon in forests.[1] For example, Plantinga et al. (1999) estimate econometric models of land use in which the shares of land allocated to forestry and agriculture are functions of net returns to alternative uses and other decision variables. The fitted models are then used in a simulation of a subsidy program for afforestation. The subsidies increase the relative net returns to forestry, which increases the area of land allocated to forest and the amount of carbon sequestered. Marginal cost schedules are constructed by arraying subsidies per unit of carbon against total carbon sequestered.

In general, previous studies find that the costs of carbon sequestration in forests are comparable to, and in some cases lower than, costs of alternative mitigation and abatement approaches. However, these analyses are focused solely on the opportunity costs of agricultural production. An important issue not considered in these studies is the impact of the resulting land use changes on biodiversity.[2] Although agricultural land is generally regarded as purely an anthropogenic habitat, it is, in fact, a significant resource for a variety of species of conservation interest (e.g. for grassland birds) (Herkert, 1994; Vickery et al., 1994). Similarly, any advantages in the form of enhanced populations of forest species that might result from afforestation are of relevance to conservation efforts, particularly in the case of neotropical migrant birds, many species of which are markedly declining in numbers (Robbins et al., 1989b; Robinson et al., 1995).

Thus, a more comprehensive analysis of carbon sequestration costs would consider not only foregone profits from agriculture, but the additional environmental benefits and costs associated with afforestation. In the present paper, we estimate the changes in bird populations likely to arise under the carbon sequestration policy modeled in Plantinga et al. (1999). Our specific objective is to determine the percentage changes in farmland and forest birds resulting from a policy that achieves conversion of 10% of the total agricultural land in each of three U.S. states (South Carolina, Maine, and southern Wisconsin). We use birds as a template for other biodiversity calculations in this context because the taxon is so data-rich, but the methods developed here can be extended to other taxa, although with less reliable data. Given current momentum toward the use of carbon management strategies to address global climate change, we assume that carbon sequestration is the primary policy objective. However, our study develops the tools needed for analysis of policies

[1] Among the studies providing marginal cost estimates are Moulton and Richards (1990), Adams et al. (1993), Richards et al. (1993), Parks and Hardie (1995), Adams et al. (1999), Alig et al. (1997), Plantinga et al. (1999), Stavins (1999), Newell and Stavins (2000), Plantinga and Mauldin (2001).

[2] Afforestation of agricultural land may have other environmental impacts. For example, in regions where intensive agriculture is practiced, afforestation typically reduces soil erosion and the contamination of ground and surface water by agricultural chemicals.

with multiple objectives. This is a first step towards the ultimate development of a national carbon sequestration strategy designed to mitigate climate change as well as to achieve other national environmental goals.

2. Materials and methods

2.1. Land-use change in response to subsidies for carbon sequestration

In an earlier study, Plantinga et al. (1999) simulate the response of private landowners to subsidies for carbon sequestration in forests. In the present study, we analyze the biodiversity impacts of the land-use changes associated with the afforestation policies. We provide a summary of the methods used in the Plantinga et al. (1999) study and present the results relevant to the current analysis. Readers are referred to the original study for more details.

Plantinga et al. (1999) simulate carbon sequestration programs in Maine, South Carolina, and Wisconsin. These states were selected because they represent a broad range of current land-use patterns, physiographic conditions, and apparent opportunities for afforestation. Maine is a heavily forested state with little additional land available for conversion to forest. In contrast, South Carolina and Wisconsin have large amounts of agricultural land that potentially can be afforested. Maine and Wisconsin are northern states with short growing seasons relative to South Carolina. Southern pine tree species, valuable for lumber and plywood production, are abundant in South Carolina. Maine and Wisconsin have a mix of hardwood species (e.g. oak, maple, birch) and softwood species (e.g. spruce, fir) used in paper production.

Econometric land-use models were estimated using standard methods developed in Lichtenberg (1989), Wu and Segerson (1995), Hardie and Parks (1997). The county shares of land in private forest (s_{it}^{f}), agricultural uses (s_{it}^{a}), and urban and other uses (s_{it}^{u}) are specified as logistic functions of exogenous variables (X_{it}):

$$s_{it}^{\mathrm{f}} = \frac{e^{\beta_{\mathrm{f}}' X_{it}}}{1 + e^{\beta_{\mathrm{f}}' X_{it}} + e^{\beta_{\mathrm{a}}' X_{it}}}, \quad s_{it}^{\mathrm{a}} = \frac{e^{\beta_{\mathrm{a}}' X_{it}}}{1 + e^{\beta_{\mathrm{f}}' X_{it}} + e^{\beta_{\mathrm{a}}' X_{it}}},$$

$$s_{it}^{\mathrm{u}} = \frac{1}{1 + e^{\beta_{\mathrm{f}}' X_{it}} + e^{\beta_{\mathrm{a}}' X_{it}}}, \tag{1}$$

where i indexes counties, t indexes time, and β_{f} and β_{a} are vectors of parameters to be estimated. The three land-use shares account for all land in the county, implying $s_{it}^{\mathrm{f}} + s_{it}^{\mathrm{a}} + s_{it}^{\mathrm{u}} = 1$ and that one of the shares is redundant. The additivity constraint is incorporated into Eq. (1) by expressing s_{it}^{u} in terms of the remaining shares (i.e. $s_{it}^{\mathrm{u}} = 1 - s_{it}^{\mathrm{f}} - s_{it}^{\mathrm{a}}$). The exogenous variables include the county average per-acre net return to forestry; the county average per-acre net return to agriculture; county population density, which controls for the diversion of land to urban and other uses; composite land quality measures, including the average quality of land in the county and the proportion of the county's land in the highest land quality classes; and a constant term and time dummies.

Separate models were estimated for each state using pooled time-series and cross-sectional data. Data were collected for all 16 counties in Maine for the years 1971, 1982, and 1995, all 46 counties in South Carolina for the years 1986 and 1993, and 49 counties in the southern two-thirds of Wisconsin for the years 1983 and 1996. Only the southern counties of Wisconsin were included because much of the land in northern Wisconsin is publicly-owned and already forested. See Ref. Plantinga et al. (1999) for details on the econometric procedures used to estimate Eq. (1) and the estimation results.

The land-use models were then used in a simulation of carbon sequestration programs. The basic approach was to simulate per-acre subsidies to forestry by augmenting the corresponding net return measure in the econometric model. This implied increases in forest area and declines in agricultural area relative to land use in the baseline. Simulations were conducted for different levels of a per-acre subsidy and the corresponding land-use changes were converted to carbon units using yield functions developed by Birdsey (1992). A marginal cost schedule was constructed by arraying the subsidies—expressed in dollars per

unit of carbon—against total carbon sequestered. The highest marginal costs were estimated for Maine, followed by South Carolina and, lastly, Wisconsin. The low costs in Wisconsin were due to the relative abundance of marginal lands with low opportunity costs for agricultural production. Opportunity costs were sufficiently low in Wisconsin to more than offset the somewhat higher carbon sequestration rates in South Carolina.

In the current study, we focus on land-use changes under scenario 1 in Plantinga et al. (1999). This scenario runs for 60 years beginning in 2000. In the baseline, all of the exogenous variables in the econometric model (net returns, population, etc.) are held constant at mid-1990s values and no timber harvesting is permitted on land enrolled in the program. Only agricultural lands are eligible and land must remain in the program for 10 years. In exchange, participating landowners receive a per-acre payment plus the costs of tree establishment. Since all the exogenous variables are constant in the baseline and subsidy levels remain constant over time, land enrolled in the 1st year of the program remains enrolled for the duration of the program. In scenario 1, the subsidy is uniformly applied across counties. Accordingly, marginal enrollment costs are equated across counties and the total cost of achieving a given amount of land conversion is minimized.[3]

2.2. Bird data

Maine, South Carolina, and Wisconsin provide diverse settings in which to study impacts on birds. Bird populations differ substantially between the three states, with South Carolina having a large component of year-round resident species while the avifauna of Wisconsin and Maine have a much higher proportion of migrant species. Bird populations in the three states differ markedly as to the environmental and land cover variables associated with their prevailing levels of species richness (O'Connor et al., 1996), providing an ecological diversity paralleling the economic diversity described above.

The bird data for the three states were derived from the national breeding bird survey (BBS), a bird population monitoring program conducted annually since 1966 in the United States and Canada, currently by the Biological Resources Division of the U.S. Geological Survey and by the Canadian Wildlife Service. The scheme is administered by USGS staff at the Patuxent Wildlife Research Center in Laurel, Maryland, and currently acquires bird data from some 4000 routes across the continent, though not all routes are surveyed annually (Robbins et al., 1989a). The survey focuses on diurnal birds that can be counted along a pre-determined route on secondary roads (3 min counts of all birds detected at 50 stops along a 25 mile route). Crepuscular and nocturnal species, and species restricted to off-road habitats, are, therefore, not surveyed. The survey was designed to obtain representative results across North America, within the constraints of this survey protocol, and a recent peer review concluded that the scheme results in data that, with only minor biases, largely meet its goals (O'Connor et al., 2000). O'Connor et al. (1996) extracted a set of 1200 representative BBS routes that had frequent and high quality surveys over the period 1981–90, and determined for each route the incidence of each species (the proportion of surveys along the route that had recorded the species). Yang et al. (1995) interpolated these incidence data for each of the 615 individual species to obtain an abundance surface over the conterminous U.S. for each individual species. The grid used was the hexagonal grid of White et al. (1992), with some 12,600 points over the conterminous U.S.

For the present project we have estimates of land use changes for each county. We therefore overlaid the hexagonal grid on a county boundary layer and determined the polygons generated by intersections of county and hexagon borders. Each polygon received the incidence value of its

[3] One could argue that this does not represent the true least-cost solution since the program targets acres rather than carbon (see Ref. Parks and Hardie, 1995). However, within each state, there is little variation in carbon sequestration rates across counties, and the gains in efficiency from targeting carbon would likely be outweighed by the additional costs of administering such a program.

source hexagon for that species and a county-wide incidence estimate was obtained by area-weighting the incidence values for the polygons. To determine how many birds would be lost from agricultural land or gained by new forests we consulted the species lists of Lauber (1991), Peterjohn et al. (1993), Rodenhouse et al. (1995) to determine which species should be assigned to each of these habitats.[4] Species not in either list were omitted, examples being shorebirds and wetland species. We then assembled the incidence data for all of the forest species and added the incidence values for each county to get an index of abundance of forest birds since incidence measures are normally proportional to absolute abundance (Hanski, 1992). We repeated this for the species in the list of farmland species. Note that the resulting abundance measures for forest and farmland birds assign equal weights to each species and, thus, assume that all species have the same conservation value. Below, we discuss alternative approaches that recognize differences in conservation importance.

In our calculations below we assume that bird densities remain constant, which is equivalent to assuming linear relationships between relative changes in bird populations and percentage changes in land use area. With rare exceptions (e.g. the house sparrow *Passer domesticus*) agricultural species do not display strong curvilinear relationships with local habitat abundance (O'Connor et al., 1999). For many widespread forest species, on the other hand, incidence falls off rapidly as forest stands break up (Askins, 1993) and we, therefore, explicitly consider below the possible effects of this on our results.

To compute statewide estimates of bird population changes, we weighted each county's results to account for the differential distribution of farmland and forest and its birds across the state. Proportional changes in the habitat were multiplied by the bird density in the habitat and then by the area of habitat to arrive at a county change in bird population. To compute the proportional change in birds at the state level, we summed the county changes in bird populations and divided by the sum of the total populations of the counties calculated in an analogous fashion.

3. Results

3.1. Land-use changes

Fig. 1 shows for each state the mid-1990s distribution of agricultural land and the distribution of land that would convert to forestry under the state-wide scenario of conversion of 10% of agricultural land for carbon sequestration.[5] Current land use and changes in land use are reported in percentage terms to control for differences in county land areas.[6] In Wisconsin, agriculture is prominent in the southeastern parts of the state, particularly along a belt of counties between Green Bay and Madison, and in a southern belt of counties bordering Illinois. However, the decrease in agricultural land under the carbon sequestration policy is concentrated into the counties with less intensive and profitable agriculture, being greatest in Jackson, Juneau, and Adams counties, and in a group of counties surrounding them in west central Wisconsin. Similarly, in South Carolina, agriculture is concentrated in a broad band of counties across the Atlantic Flatlands, with a scattering of more productive counties such as Anderson and Abbeville to the northwest and York in the north. However, the counties that would experience the greatest relative loss of agricultural land—Georgetown and Berkeley on the Coastal Plain and Fairfield and McCormack inland—are outside these areas, and most (though not all) of the

[4] Interested readers may contact the authors for a list of scientific names and habitat classifications for the 159 species used in this analysis.

[5] Plantinga et al. (1999) consider conversion rates ranging from 0 to 25% of state-wide agricultural land. The distribution of the estimated land-use changes across counties does not vary significantly with the state-level rate of agricultural land conversion.

[6] For the same reason, we emphasize percentage changes in bird populations below. This makes our results easier to compare across counties and states, though at the expense of obscuring information on absolute changes.

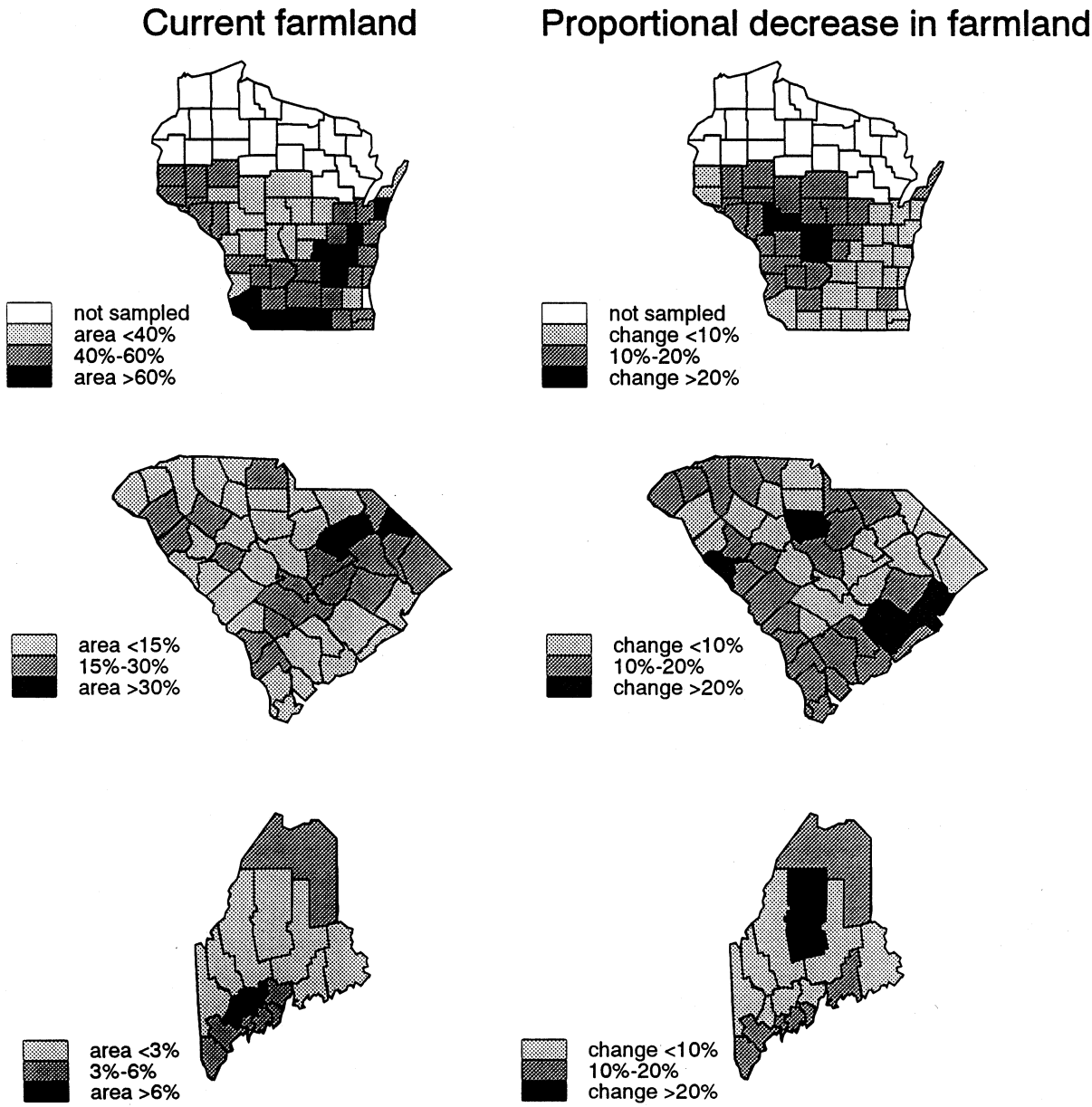

Fig. 1. Current distribution of agricultural land (left) and simulated change in agricultural land under a carbon sequestration policy (right) in southern Wisconsin, South Carolina, and Maine.

counties that would experience conversion rates of 10–20% currently have less than 15% of their land in agriculture. Finally, in Maine, agriculture is largely concentrated in the southern coastal counties, except for the potato lands of Aroostook county in the north, but with so little agriculture in Maine most of these counties would also experience conversion of land in pursuing the carbon sequestration policy. Land values in the two most intensively agricultural counties—Androscoggin and Kennebec—are such that conversion there would be low, but all the other moderately farmed

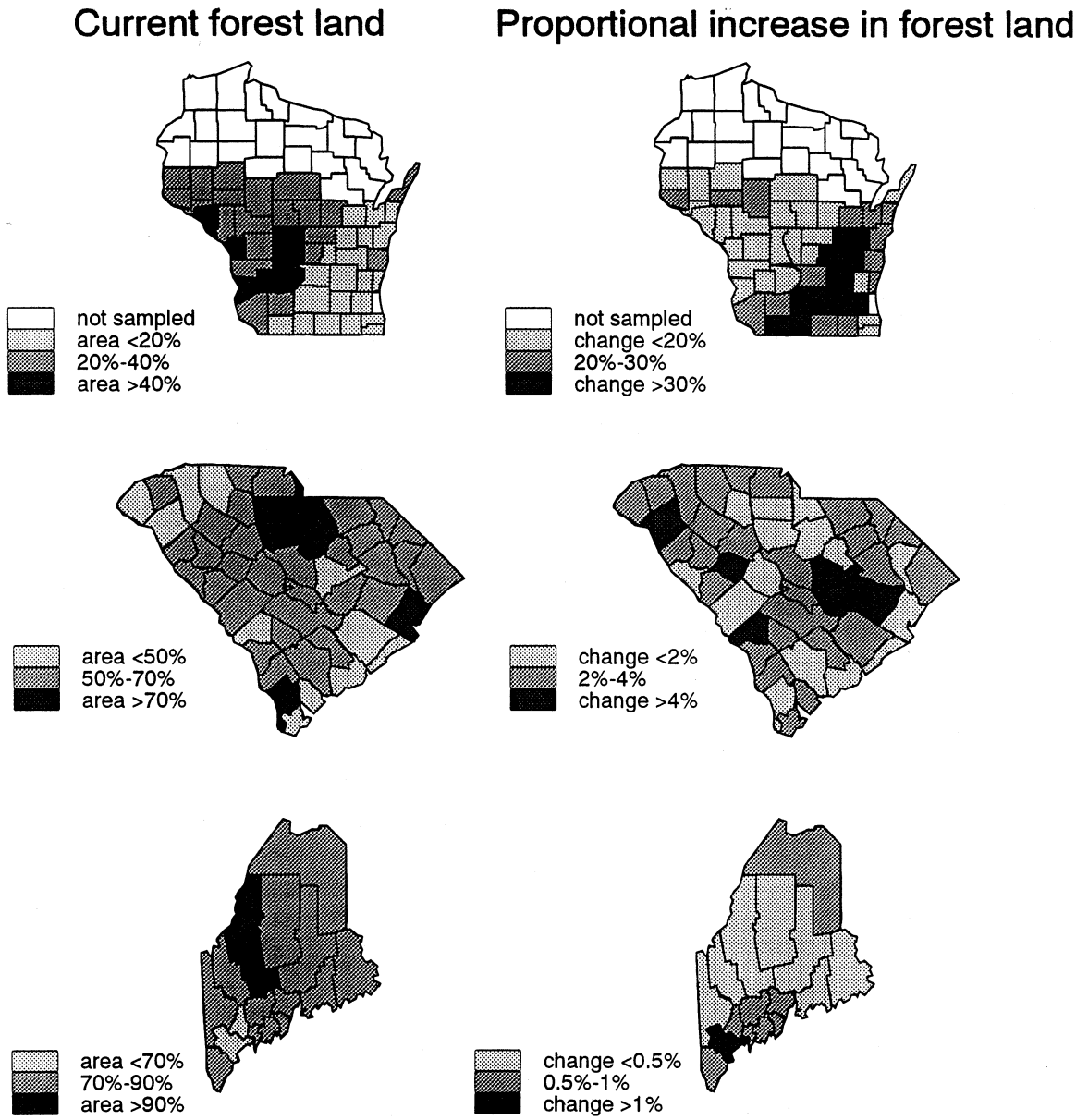

Fig. 2. Current distribution of forests (left) and simulated change in forest area under a carbon sequestration policy (right) in southern Wisconsin, South Carolina, and Maine.

counties except Waldo in the east would join inland Piscataquis and coastal Hancock counties in disproportionate conversion.

Conversion of a given area of land to forestry can have a small or a large relative effect on the extent of forests in a county, depending on the existing forest base. This is shown in Fig. 2 which maps both current forestlands and the relative increases brought about by the 10% conversion policy. The distribution of forests in Wisconsin is

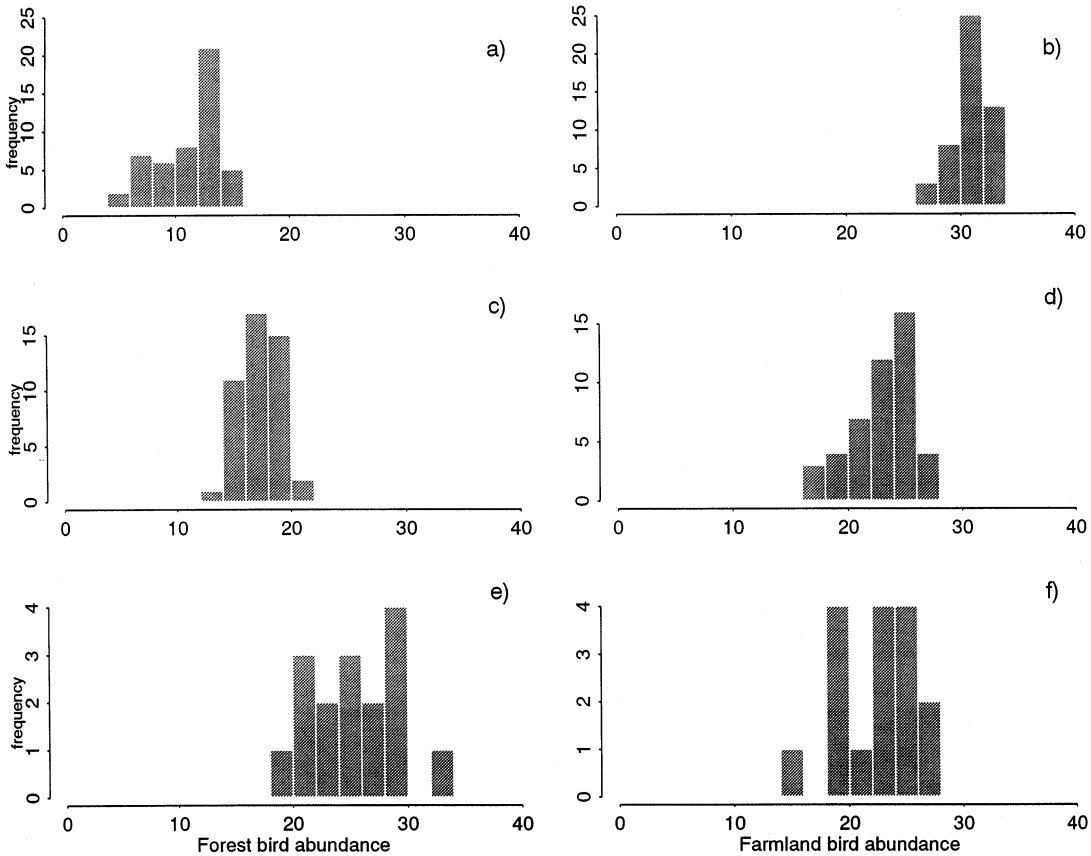

Fig. 3. Frequency distributions of forest birds (left) and farmland birds (right) across counties in southern Wisconsin (a and b), South Carolina (c and d), and Maine (e and f).

largely complementary to that of agricultural land and concentrated in the northwest of the area modeled here. As a result, the greatest relative increases in forest are not where the relative loss of agricultural land would be greatest (Fig. 1), but along its southeastern fringe: the largest relative increases in forest lands occur in a belt of counties extending northeastward and southwestward from Madison. In South Carolina, on the other hand, very few counties have less than 50% of their land already in forest, and the relative increases in forest would be only a few percent (i.e. an order of magnitude smaller than in Wisconsin) and patchy in distribution. Similarly, with so much of Maine heavily forested the extra land in forest would result in increases of only 1% or less for most counties (Fig. 2).

3.2. Bird distribution changes

The densities of birds within a county differed markedly between states, between habitats, and among counties (Fig. 3). Median densities in Wisconsin were low in forests (median 12.2, range 5.7–15.9), but much higher on agricultural land (median 31.5, range 27.5–33.3), but in South Carolina were closer together (forest median density 16.9, range 13.9–21.7; agricultural land density 23.8, range 16.5–26.5). In Maine there was considerable overlap in densities in the two habitats (forest median 25.5, range 19.6–32.2; agricultural land median 23.4, range 14.4–26.3). Examination of Fig. 3 shows that forest densities were about equally variable in all three states, though with different median densities, but agricultural bird

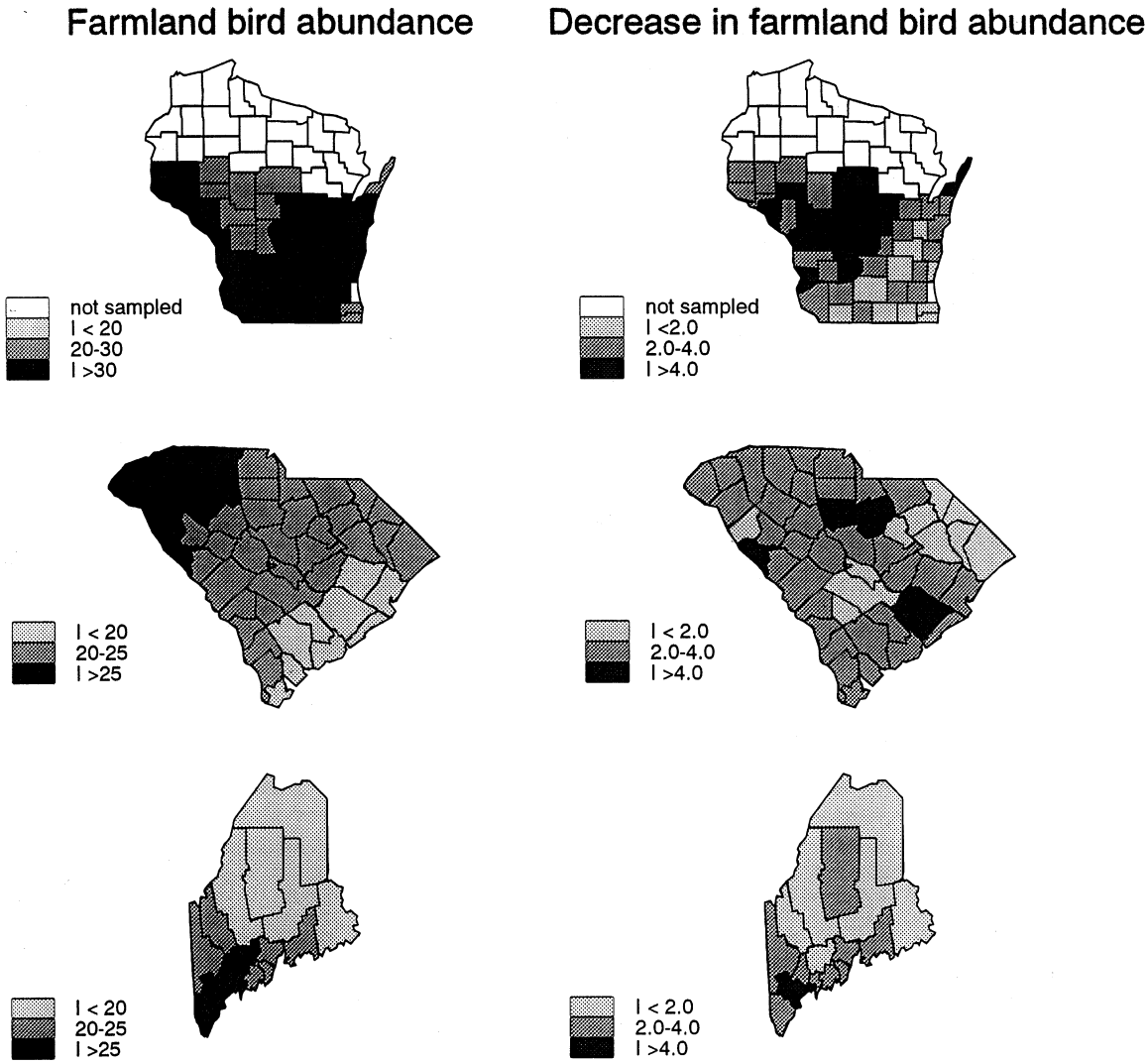

Fig. 4. Current distribution of farmland bird abundance (left) and projected change in farmland bird abundance under a carbon sequestration policy (right) in southern Wisconsin, South Carolina, and Maine. The metric I is Yang et al.'s (1995) index of total bird abundance across multiple species (see text for details).

densities became more variable from Wisconsin to Maine, probably reflecting the increase in the variability of agricultural conditions in the less agricultural states.

Since bird densities are not uniform across each state, the consequences of land conversion for bird populations depend on the product of land use changes and local bird densities. Although locally the *relative* change in farmland bird numbers must exactly mirror the relative changes in agricultural land shown in Fig. 1, a 10% (say) change in the farmland bird population can involve a large absolute change or a small absolute change, depending on the prevailing local density of birds. Fig. 4 shows the spatial distribution of farmland birds within each state. In Wisconsin the distribution of farm bird density largely resembles the distribution of agricultural land, being generally high except within a cluster of eight counties in the middle of the state. However, the two

distributions are not completely parallel and the area of largest population decrease under the carbon sequestration policy extends well beyond these less intensively farmed counties (Fig. 4). Since these changes occurred over counties different in area and with different farmland bird densities, a state-wide estimate of farmland bird loss required appropriate weighting of these effects, yielding a net reduction of 11.7% for Wisconsin (Table 1).

In South Carolina, farmland bird densities increased from the coast to the mountains rather than varying from county to county in direct proportion to the intensity of agriculture in each (Fig. 4). The gradient was relatively shallow, however, and as a result the changes in farmland bird distribution were largely determined by the relative change in agricultural land: three of the four counties with greatest change in farmland birds—McCormack, Fairfield, and Berkeley—were also among the top four for relative loss of agricultural land, and the patterns of change in agricultural land and in farmland bird distribution were generally similar (compare Figs. 1 and 4). Weighting these changes by area and size of current bird population for each county gave a state-wide reduction of 12.2% for South Carolina. In Maine, bird abundance generally decreases from south to north (Allen and O'Connor, 2000) and farmland birds, although largely paralleling the distribution of agricultural land, were correspondingly more abundant in southern farming counties than in northern ones (Fig. 4). As with South Carolina, the gradient was relatively shallow and the changes in bird numbers largely reflected the changes in agricultural land area. When weighted for county area and distribution, the estimate of the state-wide decline was 10.8%.

Table 1
Summary of percentage changes in forest and agricultural land area and in population size of forest and farmland birds

Category	Percentage changes		
	Maine	South Carolina	Wisconsin
Forestland area	0.3	2.8	18.0
Agricultural land area	−10.0	−10.0	−10.0
Forest birds	3.2	2.5	21.8
Farmland birds	−10.8	−12.2	−11.7
Forest and farmland birds	−2.0	−2.3	−1.1

Fig. 5 shows how the forest bird distribution would change under a carbon sequestration policy. In Wisconsin, the forest bird distribution largely matches that of forests and the largest increases are in counties with proportionately large increases in forest land. However, with dense populations of forest birds across the northwest of the state, even modest increases in forest lands there result in large numerical increase (Fig. 5). When this is coupled with the large amount of agricultural land available for conversion under the 10% scenario, the state-wide increase in forest birds is very substantial, constituting a net increase of 22% (Table 1). In South Carolina, the distribution of forest birds is strongly regional with highest densities in the Coastal Plain and in the Piedmont, but as there would be relatively little increase in forest area in these parts of the state under the carbon policy scenario, these areas would contribute little to the state-wide population change. Instead, the largest change in forest bird abundance would be across the Atlantic Flatlands, but as forest bird densities there are currently rather low, the increases are also low. As a result, when weighted for area and bird abundance, the state-wide change in forest bird populations would be only 2.5% under the carbon sequestration policy. Similarly, in Maine, despite high densities of forest birds in the North Woods, the planting of additional forest within the extant farming areas results in only minor increases among forest birds (Fig. 5), with a negligible state-wide increase of 0.3%.

3.3. Overall bird population changes

Since the loss of agricultural land as habitat leads to a reduction in the farmland bird population in each county while the planting of new forest leads to a gain in forest birds, the net change in bird numbers is a weighted function of the farmland and forest bird densities. Since the

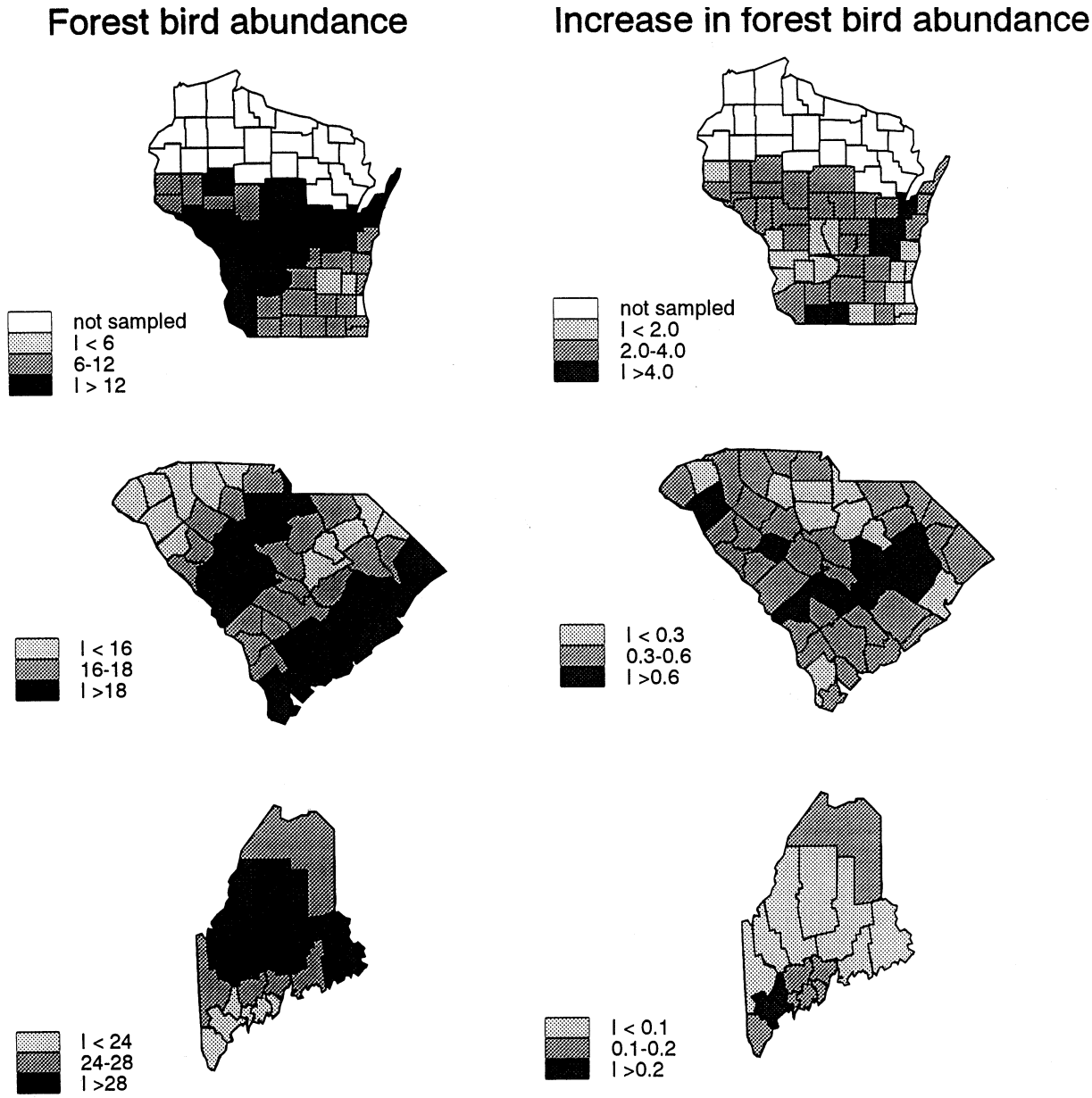

Fig. 5. Current distribution of forest bird abundance (left) and projected change in forest bird abundance under a carbon sequestration policy (right) in southern Wisconsin, South Carolina, and Maine. The metric I is Yang et al.'s (1995) index of total bird abundance across multiple species (see text for details).

area changing in land use and farmland and forest bird densities all vary from county to county, the figures have to be computed within counties and summed to a state-wide total (Table 1). In fact, all three states experience a net loss of birds: Wisconsin loses 1.5%, South Carolina 2.3%, and Maine 2.0%. That there should be a net loss even in Wisconsin where the forest bird populations increased disproportionately largely reflects the higher densities of farmland than of forest species.

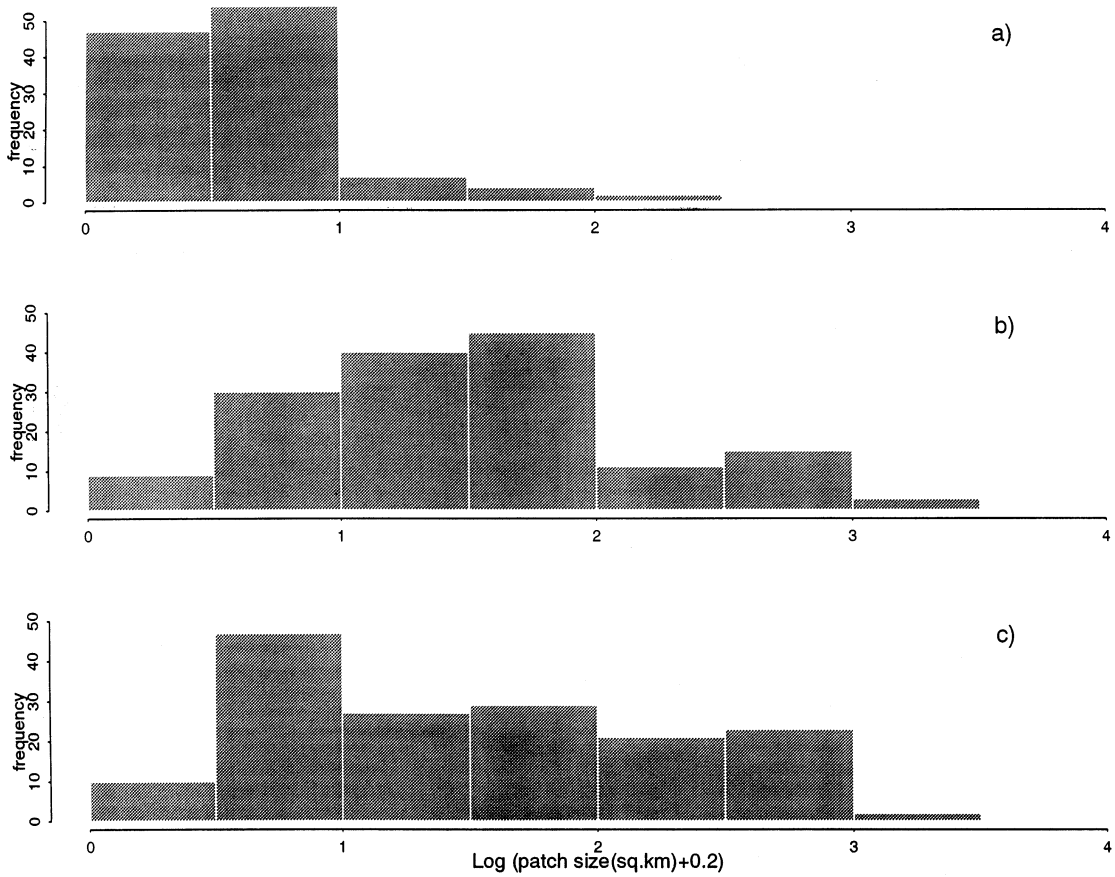

Fig. 6. Frequency distributions of forest patch sizes for mixed coniferous–deciduous forests within cells of a 640 km^2 hexagonal grid across: (a) southern Wisconsin; (b) South Carolina; and (c) Maine.

3.4. Influence of forest patch size

In calculating the likely effects of afforestation on the populations of forest birds, it was assumed above that new planting within any county would induce only a *pro rata* increase in the local population of forest birds. Where existing forest is distributed as a mosaic of small woodland blocks, however, new planting may coalesce these patches into larger stands of forest. Substantial evidence (Ambuel and Temple, 1983; Lynch and Whigham, 1984; Askins, 1993; Hoover et al., 1995) exists to indicate that bird densities are often very much lower in small patches of forest than in large ones (principally because predators and brood parasites from the surrounding matrix can penetrate a greater proportion of small than of large patches). Whilst much of the evidence derives from small patches some tens of hectares in size, we (R.J. O'Connor and L. Hayes in preparation) have estimated the population losses associated with breeding in forest stands of even some square kilometers (rather than larger ones) to range from 10 to 30% for several neotropical migrant bird species. In Fig. 6, therefore, we show the size distribution of the most common forest patch types in the three states, as derived from the remotely sensed data used by O'Connor et al. (1996). These data suggest that forest patch sizes in Maine and in South Carolina are generally so large that patch size is unlikely to be a major source of further gain in forest bird numbers. In Wisconsin, forest patches are much smaller and the gains in forest bird populations estimated here

for Wisconsin are, therefore, likely to be minima, possibly to be increased by as much as 30% by contiguous planting if the forest species there are generally area-sensitive. At present this is not known.

4. Discussion and conclusions

To the extent possible, all relevant costs and benefits should be considered in developing a national carbon sequestration strategy. Earlier economic analyses of carbon sequestration programs have focused on the opportunity costs of agricultural production, but fail to account for potential environmental effects of afforestation. In this study, we take a first step towards integrating biodiversity impacts into the analysis. The principal value of the results presented here is in their quantification of factors readily identifiable *a priori* as potentially influencing the biodiversity consequences of carbon sequestration policy. These included spatial variation in the density of forest and farmland birds, the relative extent of agricultural land and forest cover within each county, and the relative abundances of the farmland species lost by land conversion to the forest species gained in the newly afforested habitat.

Just as one could anticipate on economic grounds that the conversion of 10% of a state's agricultural land to forest is unlikely to result in a constant conversion rate across counties, one could anticipate on biological grounds that bird densities were likely to differ across counties, introducing a spatial component to the pattern of change. The magnitude of this variation across counties proved to be quite substantial (Fig. 3) and also to have marked spatial patterning (Figs. 4 and 5). These results thus suggest that the biodiversity consequences of afforestation as a carbon sequestration policy are unlikely ever to be captured by simple pro-rating of constant densities to the projected land changes. Moreover, particularly striking in Fig. 3 is the variation in relative abundance of forest and farmland birds within the three states. The densities of forest and farmland birds clearly respond differentially to variation in environment between regions, and do so by quite significant amounts.

The relative abundance of forest and agricultural land cover within a county influences the relative impact of population changes among forest birds. In each county a particular area of land would convert from crops or pasture to forestry, and with a constant density of farmland birds within a county (assumed here) the relative change in farmland bird populations is then necessarily *pro rata*. However, the relative effect on the forest bird population in the county is then weighted by the ratio of agricultural land to forest in the county. With twice as much agricultural land as forest, a 10% loss of agricultural land yields a 20% increase in forest, with associated increase in the local forest bird population. And the converse applies.

Whether these land-use changes yield a net increase or decrease in bird numbers also depends on the ratio of the local densities of farmland and of forest birds in the county. Even if a 10% change in agricultural land yielded a 20% increase in the area of forest, as in the example in the previous paragraph, a net loss of birds results unless the density of forest birds is at least 75% that on agricultural land (since a 20% increase on 75 yields 90, the reduced density of farmland birds). It is this effect that accounts for much of the net loss in bird numbers that our three study states would experience under the carbon sequestration policy modeled here. This is actually quite counter-intuitive, in that common wisdom holds that loss and fragmentation of forest is a major conservation issue for birds, and it is unexpected to find fewer birds present after planting new forest! It appears that the density of birds on agricultural land may be higher in some states than is commonly acknowledged. Thus, both the relative densities of farmland and forest birds and the relative extent of agricultural land and forest in the county determine whether the policy would result in a gain or loss of avian abundance, with the spatial variation between counties then determining both the spatial patterning of the gains and losses and, by virtue of the variation in area among the counties, whether the statewide outcome is a gain or a loss.

No consideration was given above to uncertainties in the measurement of the land cover propor-

tions and bird densities, but we expect the uncertainties in our estimates to be small. The base land-use statistics are measured with very little error. The forest area measures are from U.S. Forest Service plot-level surveys, which have relatively small sampling errors. For example, the sampling error for timberland area in most Wisconsin counties (1996 inventory) is below 5%. The agricultural land area measures are from the Census of Agriculture, which attempts to provide a complete enumeration of the farm population. Response rates for the Census are very high and, thus, measurement errors are expected to be low. A second source of uncertainty arises with the predictions of land use change. We do not expect the prediction error to be large because of the good fit of the econometric models and the fact that we consider modest changes in land use.

For bird densities the effects of uncertainty in incidence could be assessed by bootstrap sampling of the calculations, but a crude estimate suffices to show that the effect is small. Uncertainty in incidence at a single location will be maximal for a species present in only half the surveys, which with our decadal estimate corresponds to a standard deviation of 0.158 ($=(0.5\times 0.5/10)^{1/2}$). If only 40 species (about half of the farmland or forest bird species pools) were present in a location, the standard error for the farmland or forest bird incidence would then be only 0.025, negligible as a fraction of any of the incidences reported in Table 1 and an order of magnitude smaller than the changes in overall populations discussed. In practice, the use by Yang et al. (1995) of data from multiple locations, the smaller standard deviations of uncertainty in both common and rare species, and the generally larger species tallies at each location mean that census uncertainties can be neglected here.

We noted above that the predictions of forest bird changes might require modification to take account of forest patch size distribution, particularly in Wisconsin. Askins (1993) found that previously declining neotropical migrant species increased in numbers as afforestation restored the contiguity of forest in Connecticut. Hence, our Wisconsin estimates of the increase in forest bird populations must be seen as conservative: our unpublished estimates indicate that populations of neotropical migrants may be 30% lower in areas of forest fragments than where the forest is contiguous. If this were true of the Wisconsin forest species, forest populations could potentially increase by an additional 43% ($=100/(100-30)$) if the new forests were planted to maximize contiguity and reduce patch edge predation and parasitism.

A further assumption in our calculations was that there were no threshold effects in the influence of forest density on birds. It is in principle possible that there might need to be some minimum density of forest in an area before a forest species would settle there, and such an effect would mean that newly forested land in counties with little or no previous forest planting might not yield the expected gain in forest birds. However, the density estimates for forest species that we used were estimated from empirical bird distribution data by Yang et al. (1995) using a grid of sufficient resolution to average four points per county. Accordingly, it is likely that any such effects present have already been incorporated into our analysis.

Our estimates of bird abundance were obtained by summing estimates of incidence—the proportion of surveys at a site that recorded the species—over the 1981–90 decade. It is well established that incidence is directly proportional to absolute densities where estimates of both have been available, except for a small number of very common ubiquitous species whose densities may vary within an incidence of 1.0 (O'Connor and Shrubb, 1986; Hanski, 1997). If the constant of proportionality were the same for all species, our sum of incidence values for forest species and for farmland species would be directly proportional to the sum of the corresponding bird densities, and if the value of the constant were known we could re-express our measure as true densities. It is, however, unlikely that all species share the same constant and this leaves us with a possible bias. Our sum of incidence measure would then really be of form $\Sigma\, k_i D_i$, where D_i is the density of the ith species and k_i is the constant of proportionality between incidence and density for that species. Hence, were the constant for some partic-

ular species to be markedly higher than for the other species, then our summed incidence metric would be higher for all points within the range of that species than it should be for proportionality to true total density. However, our metric was summed over many forest and farmland species, so any single error would be relatively small in effect: with 50 species, even a doubling of the constant of proportionality for one species would induce an error of just two parts in a hundred. In addition, with so many species individual errors were likely to cancel each other. We, therefore, consider the likely magnitude of any error from this source to be rather small relative to the effects measured.

One important issue not considered explicitly here is that different species have different conservation significance. Our present calculations treated all species as equally important, and computed the net change in birds under a carbon sequestration policy as the sum of the forest and farmland bird changes in a county. In practice, some of the species lost from the newly forested agricultural land will be of greater conservation value than the forest species gained. In principle, the analysis presented here could be conducted using an index of conservation value instead of the incidence metric. For example, the incidence value for each species in a county could be weighted to reflect its relative conservation value, and the calculations that were here applied to just two groups—total forest birds and total farmland birds—could instead be computed over each of the entries in vectors of individual species incidence values for forest and for agricultural land. Applying the conservation weights to the results with and without the sequestration policy would then yield estimates of the magnitude and distribution of conservation impacts. Several of the conservation value weighting schemes devised for biodiversity complementarity analyses (Polasky and Solow, 1995; Csuti et al., 1997) could readily be adopted for use in the present context. As well, willingness-to-pay estimates from non-market valuation studies could be applied to derive explicit estimates of the benefits or costs associated with changes in bird populations.

As noted above, carbon sequestration is the objective of the policy considered in this study and the scenario we evaluate is designed to achieve a given level of land enrollment at the least cost. There are several ways in which biodiversity impacts—and other environmental effects of afforestation—can be incorporated into the analysis of carbon sequestration policy. If estimates of all the relevant benefits are available, including the benefits of climate change mitigation and biodiversity, then a standard cost-benefit analysis can be performed. In this case, the optimal policy is to enroll land until the marginal cost of enrollment equals the sum of marginal benefits.[7] If, as in the present case, reliable benefits estimates are elusive because of the complexities of the natural phenomenon involved, an alternative is to conduct cost-effectiveness analysis. This might involve minimizing the total cost of enrolling land subject to constraints on the minimum amount of carbon sequestered and the maximum tolerable impacts on biodiversity. Assuming the biodiversity constraint binds, the solution identifies the cost of departing from a least-cost carbon sequestration strategy in order to accommodate biodiversity objectives. The shadow value on the constraint gives the implicit price for biodiversity and, thus, sheds light on the nature of the tradeoffs involved.

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[7] For a given amount of land conversion, the marginal cost of enrollment equals the corresponding per-acre subsidy that is used in the simulations. The marginal cost of enrollment reflects the opportunity cost of agricultural production. See Ref. Plantinga et al. (1999) for more details.

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