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Projected land-use change impacts on ecosystem services in the U.S.

Short title: *Land-use change impacts on ecosystem services*

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Abstract

Providing food, timber, energy, housing, and other goods and services, while maintaining ecosystem functions and biodiversity that underpin their sustainable supply, is one of the great challenges of our time. Understanding the drivers of land-use change and how policies can alter land-use change will be critical to meeting this challenge. Here we project land-use change in the contiguous U.S. to 2051 under two plausible baseline trajectories of economic conditions to illustrate how differences in underlying market forces can have large impacts on land-use with cascading effects on ecosystem services and wildlife habitat. We project a large increase in croplands (28.2 million ha) under a scenario with high crop demand mirroring conditions starting in 2007, compared to a loss of cropland (11.2 million ha) mirroring conditions in the 1990s. Projected land-use changes result in increases in carbon storage, timber production, food production from increased yields, and >10% decreases in habitat for 25% of modeled species. We also analyze policy alternatives designed to encourage forest cover, natural landscapes, and reduce urban expansion. Although these policy scenarios modify baseline land-use patterns, they do not reverse powerful underlying trends. Policy interventions need to be aggressive to significantly alter underlying land-use change trends and shift the trajectory of ecosystem service provision.

Significance Statement

Land-use change affects the provision of ecosystem services and wildlife habitat. We project land-use change from 2001 to 2051 for the contiguous U.S. under two scenarios reflecting continuation of 1990s trends and high crop demand more reflective of the recent past. These scenarios result in large differences in land-use trajectories that generate increases in carbon storage, timber production, food production from increased yields (even with declines in cropland area), and >10% decreases in habitat for one-quarter of modeled species. We analyzed three policy alternatives that provide incentives to maintain and expand forest cover, conserve natural habitats, and limit urban sprawl. Policy interventions need to be aggressive to significantly alter underlying land-use trends and shift the trajectory of ecosystem service provision.

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Introduction

Land-use change can greatly alter the provision of ecosystem services. Globally, the conversion of native grasslands, forests, and wetlands into croplands, tree plantations, and developed areas has led to vast increases in production of food, timber, housing, and other commodities but at the cost of reductions in many ecosystem services and biodiversity (1). Although recent land-use change in the U.S. has not been as rapid as in the tropics, it has been significant. The area of croplands has decreased and forests and urban areas have expanded since World War II (2). For example, forest lands in the contiguous U.S. expanded by 5.7 million acres between 1982 and 2007. However, basic estimates of net land-use change often hide more complex dynamics. More than 30 million acres transitioned into or out of forest between 1982 and 2007 (3). Such transitions alter landscape patterns and ecosystem functions, both of which affect the provision of ecosystem services.

We use an econometric model to predict spatially explicit land-use change across the contiguous U.S. from 2001 to 2051. The model estimates the probability of conversion among major land-use categories (cropland, pasture, forest, range, and urban) based on observations of past land-use change, characteristics of land parcels, and economic returns, while accounting for endogenous feedbacks from the policies into commodity prices. A key advantage of this approach is that it allows us to simulate the effects of future policies that modify the relative returns to different land uses.

We integrate land-use change analysis with models of ecosystem service provision: carbon storage, food production, timber production, and the habitats of 194 terrestrial vertebrate species selected for their ecological and cultural importance or sensitivity, including amphibians, influential species (e.g., top predators, keystone species, and ecosystem engineers), game species, and at-risk birds. We use a broad definition of ecosystem services (the goods and services provided by nature that are of value to people) to include both agricultural production, which includes both natural and human-made inputs, and habitat provision for wildlife, which may or may not be directly valued by people. We use the coupled econometric land-use and ecosystem service models to explore the effects of incentive and land-use regulation policies that affect land-use patterns and ecosystem service provision.

We explore the potential impacts of land-use change under two alternative baseline scenarios and three alternative policy scenarios (Table 1). The first baseline scenario (1990s Trend) assumes continuation of exogenous factors driving land use during a five-year period from 1992 to 1997. The second baseline scenario (High Crop Demand) increases the price of agricultural commodities relative to the 1990s Trend with concomitant pressures to expand agricultural lands, which more closely resembles the five-year period from 2007 to 2012. The two scenarios allow us to gauge the sensitivity of our results to different assumptions about the underlying drivers of land-use change. We also analyze how three alternative policy scenarios would shift land use and the provision of ecosystem services relative to the baseline scenarios – see Table 1: i) Forest Incentives—incentives for afforestation and reduced deforestation similar to carbon sequestration incentives (e.g., 4), ii) Natural Habitats—incentives for conservation of forest and range (grasslands/shrublands) to prevent conversion to crop land, pasture, or urban, and iii) Urban Containment—prohibition on urban land expansion in all non-metropolitan counties to concentrate urban expansion in existing metropolitan areas. For all scenarios, land-use changes were only simulated for privately-owned land from 2001 to 2051; land use on public land was held constant.

Results

Our model projects substantial land-use change between 2001 and 2051 under both the 1990s Trend and the High Crop Demand scenarios (Fig. 1, 2, and S1) with rapid urban growth (Fig. 1d, 2a) and loss of rangelands and pasture (Fig. 1b, 1e, 2a). Urban growth is projected to be greatest near existing major metropolitan areas. Not surprisingly, given the current distribution of rangeland and pasture, the losses in these two land-cover types are primarily in the western and eastern U.S., respectively. Forest land showed modest increases overall but had a complex pattern of gains and losses (Fig. 1c, 2a).

Comparing the projections for the two baseline scenarios clearly demonstrates the importance of underlying drivers of land-use change (Fig 1a, 2a). In the High Crop Demand scenario cropland is projected to have a large increase (28.2 million ha) compared to a loss of cropland under the 1990s Trend scenario (-11.2 million ha). The increase in cropland in the High Crop Demand scenario comes at the expense of larger declines in pasture (30.5 million ha versus 15.0 million ha) and range (31.2 million ha versus 19.6 million ha) and smaller increases in forest (7.3 million ha versus 16.3 million ha) and urban land (26.2 million ha versus 29.5 million ha) relative to the 1990s Trend scenario.

We project a large increase in food production under both scenarios—a 50% increase in kilocalories under the 1990s Trend, and a doubling under the High Crop Demand scenario (Fig. 2b). These increases are roughly in line with estimated increases in global food demand between 2000 and 2050 of 70% (5) or doubling (6). Increases in food production are driven by increases in crop yield (which we assume increase by 6% every five years) and changes in agriculture area.

Both land-use change scenarios also result in overall increases in carbon storage and timber production (Fig. 2c, 2d). Carbon stored in biomass increases by 1.1 billion Mg (6%) under the 1990s Trend scenario and 556 million Mg (3%) under the High Crop Demand scenario. Carbon stored in soil increases slightly under the 1990s Trend (121 million Mg) but decreases under the High Crop Demand scenario (-306 million Mg). Both changes are small relative to the total stock of soil carbon (Fig. 2c).

Habitat for the four groups of species we modeled showed overall declines under both land-use change scenarios. Overall, 47 out of 194 species are projected to lose more than 10% of their habitat under the 1990s Trend scenario whereas only 10 experience gains of more than 10%. We see a similar pattern in the High Crop Demand scenario (43 species lose more than 10% and 5 gain more than 10%). On average, species do somewhat better in the High Crop Demand scenario compared to the 1990s Trend (Wilcoxon signed-rank test, $V = 5337$, $N = 194$, $P < 0.001$, median difference = 1.6%). The four groups of species (amphibians, influential species, game species, and at-risk birds) responded in broadly similar ways to the two future scenarios (Fig. 2e, 2f, 2g, 2h). At-risk birds are the most sensitive to land-use change. Roughly 1/3 of these species are projected to lose more than 10% of their habitat (Fig. 2h).

We analyze the impact of alternative policy scenarios on land-use change relative to change under the baseline scenarios and find similar results regardless of which baseline scenario (1990s Trend or High Crop Demand) is used. Therefore, we only present policy results relative to the 1990s Trend scenario (see SI Text for the comparison with the High Crop Demand scenario). Each of the three policy alternatives (Forest Incentives, Natural Habitats, and Urban Containment) result in substantial land-use change relative to the 1990s Trend scenario (Fig. 3, 4, and S2-S4). The Forest Incentives policy produces an additional 30.6 million ha of forest land (a 14% increase relative to baseline), which occurs largely at the expense of rangeland and cropland and to a lesser degree pasture (Fig. 3a, 4a). The largest increases in forest land are east of the 100th meridian in areas with large amounts of land currently in agriculture. Most of the increase in forest area is the result of afforestation and, thus, requires large government expenditures on subsidies to landowners (approximately $7.5 billion per year). The Natural Habitats policy results in an increase in rangeland (12.4 million ha, a 5% increase relative to baseline) at the expense of crops and pasture, but virtually no change in forest land despite there being a tax on land leaving forest (Fig. 3b, 4a). In contrast to the Forest Incentives policy, the Natural Habitats policy generates tax receipts for the government of approximately $1.8 billion per year. The Urban Containment policy reduces the amount of urban growth (from 29.5 million ha to 12.2 million ha) and results in slight increases in the other land-use types (Fig 3c, 4a). The Urban Containment policy is the only one of the three policies that alters the expansion of urban land in a meaningful way.

The Forest Incentives policy has the largest positive effect on biomass carbon (1.7 billion Mg increase relative to baseline, 8%), and timber production (235 million cf relative to baseline, 18%). The Forest Incentives policy reduces food production by 10% (1.93×10^{14} kcal) compared to the 1990s Trend scenario. The Urban Containment policy results in modest increases in biomass carbon storage (2%), timber production (5%), and food production (4%), relative to the 1990s Trend values. By contrast, the Natural Habitats policy has relatively small negative effects on all three of these services.

The Natural Habitats policy has the greatest positive effect on habitat of any policy scenario with 31% of the species (61 of 194) gaining at least 10% in habitat area by 2051, compared to 13% of species under the Forest Incentives policy, and 16% under the Urban Containment policy. All groups of species do better under the Natural Habitats policy (Fig. 4e-h). Both Forest Incentives and Urban Containment policies also result in more species gaining than losing at least 10% in habitat area, but the positive changes were not as great as under the Natural Habitats policy.

Discussion

Land-use change is a major driver of change in the spatial pattern and overall provision of ecosystem services. Our results demonstrate that differences in the underlying drivers of land-use change, such as changes in future crop prices, can have large impacts on projected land-use change with cascading effects on the provision of ecosystem services. We find that projected land-use changes by 2051 will likely enhance the provision of some ecosystem services, carbon sequestration and timber harvests, due to expansion of forest land under our baseline scenarios. On the other hand, almost one-quarter of modeled species (47 out of 194 species in the 1990s Trend scenario) are projected to lose greater than 10% of their habitat by 2051; only a few species are projected to gain more than 10% of their habitat.

During the 1990s, low agricultural prices generated low returns to agriculture relative to returns to other land uses driving land out of agriculture and into forest and urban land. The shift toward forest land increases the amount of carbon storage in biomass and timber production, and generates a modest gain in carbon stored in soil. Despite land moving out of agriculture, food production increases under the 1990s Trend scenario due to increases in crop yields. We assume a 6% increase in yield every 5 years, which generates a 79% increase in yields between 2001 and 2051. This productivity gain is below the increase in major crops during the previous 50-year period (7) but consistent with projections showing positive but declining growth in U.S. agricultural productivity (8). This predicted increase in yields could be overly optimistic if yield growth is linear rather than exponential (9) or if climate change has significant negative impacts on yields (10). We find that assumptions regarding trends in yields have more impact on food production than do changes in cropland area. Other factors, such as changes in management intensity in response to changes in prices, will also affect productivity. These other factors, however, were not modeled here.

Our results show that the adoption of specific policies can influence land-use changes and increase the expected provision of some ecosystem services but at the expense of others; there appear to be inevitable tradeoffs among services (11). For example, forest land increases by over

30 million ha under the Forest Incentives policy, the largest change relative to the baseline under any of the three policies. This increase in forest land leads to significant increases in timber production (18%) and biomass carbon (8%), relative to the 1990s Trend scenario. The Forest Incentives policy also leads to some improvement in species conservation (the number of species gaining >10% habitat increases from 10 to 26, whereas the number losing >10% decreases from 47 to 26). One cost of this policy, however, is a decline in food production relative to the 1990s Trend scenario.

Such tradeoffs can make it difficult to provide clear policy advice. Providing evidence of the change in overall net benefits when some ecosystem services increase and others decrease requires taking the analysis a step further by either pricing ecosystem services and applying benefit-cost analysis, or using some form of multi-objective decision analysis (12-14). Pricing ecosystem services would allow comparison of the value of changes to each ecosystem service in a common monetary metric and a summary statement of overall change in net value. Methods to value ecosystem services have been outlined elsewhere (e.g., 13, 15) and applied to at least some services to illustrate how to rank alternatives (16). Although some ecosystem services are readily expressed in a common monetary metric of value (e.g., crop and timber production values), other ecosystem services are not (e.g., existence value of wildlife).

Even without valuing all services in a common monetary metric, several lessons emerge from our analysis. Whether positive incentives (a subsidy) are more effective than penalties (a tax) in affecting land-use change depends on trends in baseline conditions. For example deforestation taxes in the Forest Incentives and Natural Habitats policies have little impact because there is a limited amount of baseline deforestation. By contrast, the payments provided under the Forest Incentives policy for establishing new forests has a large effect because there is a large amount of agricultural land that can be converted to forest.

Though policies clearly have some effect, we find it difficult for them to overcome powerful trends originating from market fundamentals or the overall structure of government programs that shape land-use change. For example, urban land is projected to increase by 26.2 or 29.5 million ha (63 or 71%) from 2001 to 2051 under baseline conditions. Under the Urban Containment policy, a policy that is probably stronger than could realistically be put into practice, we still see a gain of 12.2 million hectare in urban area. One reason that policy effects are limited is because of market price feedbacks. A policy that subsidizes one land use indirectly raises the returns to other uses. For example, a subsidy to forests reduces the supply of cropland. Increases in forest land lead to larger timber supply and lower timber prices whereas a reduction in cropland leads to reductions in crop production and increases in crop prices. These price effects tend to limit how much land shifts from cropland to forest. Further, increases in crop prices can lead to conversion of pasture or range into crops. The gains in total carbon storage resulting from forest expansion are then partially offset by decreases in soil and biomass carbon from the conversion of pasture and range to cropland.

Our research contributes to a large existing literature on land-use change and ecosystem services (1) in two significant ways. First, we build from empirical analysis of landowner decisions based on relative returns (4) to predict land-use change and its impact on ecosystem services and habitat provision with illustrative and implementable policies. Previous simulations of grid cell-level land-use change over large landscapes have used a combination of basic economic theory, agent-based models, and *ad hoc* rules to predict land-use change (17-19). Other ecosystem service assessments have used experts to envision land-use changes (e.g., 16, 20). Our model results can be compared to other relatively fine-grained model projections of regional land-use change scenarios (e.g., 21), agent-based modeling approaches, and deterministic housing growth models (e.g., 19).

Second, we combine an endogenous price modeling approach that captures the effect of changes in major land uses (agriculture, forestry, urban development) with detailed local-scale analysis of land-use change important for determining the provision of ecosystem services. Our approach is not a true general equilibrium model because we do not simultaneously balance supply and demand in all markets or account for all market feedbacks. However, we do account for what is arguably one of the most critical market feedbacks, the influence of aggregate land-use change on commodity prices. Most endogenous price modeling approaches generate results at aggregate regional scales (e.g., 9, 22). On the other hand, many of the most spatially detailed local-scale land-use analyses suitable for ecosystem service analysis do not incorporate price feedbacks resulting from induced changes in land use (e.g., 20).

Although our analyses address several of the main forces that drive land-use change and their impacts on ecosystem services, there are additional aspects of these relationships that our models do not address. For example, we do not include analyses of changes in land management. Land management is likely to respond to changes in relative prices and to biophysical restrictions. We would, for instance, expect more intensive farming practices in response to higher agricultural prices (23). Similarly, although we only allowed conversion to forest in areas where Holdridge Life Zones indicate forests can grow (SI Text), conversion in some arid rangelands will likely require intensive management. Our conclusions regarding trends in wildlife habitat are also a function of the species we have chosen to evaluate and not just patterns in land-use change. For example, few of these species we have modeled are threatened or endangered. These somewhat common species generally have relatively large ranges and are less likely to experience large percentage changes in habitat area than are more area-restricted species.

Clearly, we cannot anticipate all of the market and biophysical forces that will influence land use over the next four decades, such as the emergence of new technologies, shifts in societal preferences, and climate change. Our primary goal is to explore the effects of land-use policies relative to a given baseline rather than to predict future land use. Unanticipated market and societal preference events that affect relative returns will influence future land use under both the baseline and policy scenarios, making predictions about the difference between scenarios less uncertain than prediction of future land use itself. And although climate change could impact certain scenarios and policies more than others we have left that analysis for further research (see the SI Text for discussion).

Despite these modeling caveats, our results provide an empirically-based estimate of the ability of relatively strong land-use based policies to deliver ecosystem services. Perhaps the most important lesson that emerges from our analyses is that there are powerful underlying trends that will drive land-use change, as illustrated by the two baseline scenarios that we examined. Land-use patterns can be affected by policy interventions, but such interventions will need to be aggressive to significantly alter underlying land-use change trends.

Materials and Methods

Our analysis consists of two major parts: projections of future land use based on an econometric model, and an assessment of the implications of future land-use change on select ecosystem services. We discuss both parts briefly here. Details are provided in the SI Text.

Econometric land use model and policy simulations

The land-use change model was parameterized using observed land-use changes between 1992 and 1997 at 844,000 sample points of the USDA NRI National Resource Inventory (3). Plot-level land-use change is explained by county-specific net returns to each land use and each plot's soil type and starting land use (4). As such, our land-use model accounts for spatial heterogeneity in the factors driving land-use decisions (e.g., differences among plots in soil type), but does not explicitly model spatial processes such as the effect that the land use of one plot might have on land-use decisions made for neighboring plots. From the estimated econometric model, we generated a land-use transition probability matrix for the period 2001 to 2051 for each county-soil type combination. The transition matrices account for movements of land among five NRI categories: crops, pasture, forest, urban, and range (see Table S1), where range includes grasslands and shrublands, and urban includes developed open space and low- to high-intensity urban lands. The econometric model also includes endogenous feedbacks from land-use changes to net returns. By using endogenous price feedbacks in our model we control for the impact that changes in the supply of a good can have on market prices and net returns to land. The econometric model represents changes among land uses (the extensive margin) but does not model changes in the intensity of uses (the intensive margin). As a way to partially remedy this shortcoming, we assume an exogenous 6% increase in crop yields every five years. Allowing land-use intensity to change endogenously would be an important extension of the current approach. Further, many spatial variables that plausibly affect land use, such as distance to cities and the land use choices of neighboring parcels, cannot be included in our land-use change model due to limitations in our 1992 – 1997 land use data (SI Text).

The initial 2001 land-use map in our simulations comes from the National Land Cover Database (NLCD) (24). We re-sampled the original 30-m resolution NLCD grid map to a 100-m resolution to give a more realistic size for average land-use change plots (25). We then used the 50-year land-use transition matrices with the re-sampled 2001 map to generate an expected plot-level 2051 land-use map for the contiguous U.S. The spatial grain mismatch between the net returns data (county-level resolution) and land use map means the interpretation of our results is constrained by the coarser county-level data.

Ecosystem service models

We modeled soil carbon storage for all land uses. Additionally, for forest and urban areas, we accounted for above- and below-ground biomass carbon storage, but not for other land use types. To estimate forest biomass carbon, we made several simplifying assumptions. We assumed that all privately-owned forests would be managed with even-aged rotations, that the rotation length was determined by the Faustmann formula, and that all age classes were evenly represented in the landscape. Forest biomass carbon was then assessed based on the Forest Inventory and Analysis (FIA) estimates for forest types in each county and allometric curves of tree growth (26). Soil carbon estimates to a soil depth of 30 cm for each land-use type in a county were based on carbon stock estimates from Bliss *et al*. (27). See the SI Text for details.

We estimated kilocalorie production on private croplands in 2051 as a function of observed 2001 yields and observed 2001 crop-planting patterns on the landscape (28, 29). We assumed a 6 % increase in yield every 5 years across the entire nation and all crops (28). In addition, we modeled time-invariant timber yield from private forests based on average yield data from FIA and the rotation length that was estimated as part of the biomass carbon assessment.

To assess species responses to land-use change, we quantified the amount of change in habitat area individually for 194 terrestrial vertebrate species, which were chosen for their ecological or social importance: amphibians (because of their sensitivity to environmental change), influential species (in terms of their ecological role, e.g. top predators, keystone species, and ecosystem engineers), game species (because of their importance to hunters and land managers), and at-risk birds (categorized by the American Bird Conservancy (30) as ‘vulnerable’ or ‘potential concern’). We quantified habitat area for each species under current and future land-use conditions, based on species’ geographic range and habitat associations. For birds, we used only portions of the range that were used for breeding or year-round residency. Our species-habitat associations were based on a land-cover classification of ecological systems (31), cross-walked to the land-use categories used in the econometric model. Across the contiguous U.S., for each species, areas of current (2001) land use/land cover (LULC) were given a score of 1 if they were prime habitat and a score of 0 otherwise. For simulated future LULC, we used the land-use transition probability matrices generated by the econometric land-use model under each of our scenarios. The summation of the potential habitat values within a species' range in 2051, compared to the summed habitat value of current land cover, quantified the impact of future land-use change on a given species. For each species, we compared the projected change in habitat area resulting from each policy scenario, and summarized results by our four species groups. See SI text, Tables S2-S4, and Dataset S1 for more details.

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References

1. Millennium Ecosystem Assessment (2005) *Ecosystems and Human Well-being Biodiversity Synthesis* (World Resources Institute, Washington, D.C.).

2. Nickerson C, Ebel R, Borchers A, & Carriazo F (2011) Major Land Uses in the United States, 2007. Economic Research Service, Economic Information Bulletin 89. (U.S. Department of Agriculture, Washington, DC).

3. U.S. Department of Agriculture (2009) Summary Report: 2007 National Resources Inventory. (Natural Resources Conservation Service, Washington, DC).

4. Lubowski RN, Plantinga AJ, & Stavins RN (2006) Land-use change and carbon sinks: Econometric estimation of the carbon sequestration supply function. *Journal of Environmental Economics and Management* 51:135-152.

5. Alexandratos N & Bruinsma J (2012) World agriculture towards 2030/2050: The 2012 Revision. (Food and Agriculture Organization, Rome).

6. Tilman D, Balzer C, Hill J, & Befort BL (2011) Global food demand and the sustainable intensification of agriculture. *P. Natl. Acad. Sci. USA* 108:20260-20264.

7. US Department of Agriculture-National Agricultural Statistics Service (2012) Quick Stats 2.0. Last accessed: December 20, 2013. http://nass.usda.gov/Quick_Stats/.

8. Alston J, Anderson M, James J, & Pardey P (2010) *Persistence Pays: U.S. Agricultural Productivity growth and the Benefits from R&D Spendin* (Springer, New York).

9. Baker JS, Murray BC, McCarl BA, Feng S, & Johansson R (2013) Implications of alternative agricultural productivity growth assumptions on land management, greenhouse gas emissions, and mitigation potential. *American Journal of Agricultural Economics* 95:435-441.

10. Lobell D, Schlenker W, & Costa-Roberts J (2011) Climate trends and global crop production since 1980. *Science* 333:616-620.

11. Polasky S, *et al.* (2008) Where to put things? Spatial land management to sustain biodiversity and economic returns. *Biol. Conserv.* 141:1505-1524.

12. Bateman IJ, Mace GM, Fezzi C, Atkinson G, & Turner K (2011) Economic analysis for ecosystem service assessments. *Environmental and Resource Economics* 48:177-218.

13. National Research Council (2005) Valuing Ecosystem Service: Towards Better Environmental Decision-making. (National Academies Press, Washington, DC).

14. Newton AC, *et al.* (2012) Cost–benefit analysis of ecological networks assessed through spatial analysis of ecosystem services. *J. Appl. Ecol.* 49:571-580.

15. Polasky S & Segerson K (2009) Integrating ecology and economics in the study of ecosystem services: some lessons learned. *Annual Review of Resource Economics* 1:409-434.

16. Nelson E, *et al.* (2009) Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Front. Ecol. Environ.* 7:4-11.

17. Alcamo J, Leemans R, & Kreileman E (1998) *Global Change Scenarios of the 21st Century: Results from the IMAGE 2.1 Model* (Pergamon, Oxford).

18. Swetnam RD, *et al.* (2011) Mapping socio-economic scenarios of land cover change: A GIS method to enable ecosystem service modelling. *Journal of Environmental Management* 92:563-574.

19. Bierwagen BG, *et al.* (2010) National housing and impervious surface scenarios for integrated climate impact assessments. *P. Natl. Acad. Sci. USA* 107:20887-20892.

20. Goldstein JH, *et al.* (2012) Integrating ecosystem-service tradeoffs into land-use decisions. *P. Natl. Acad. Sci. USA* 109:7565-7570.

21. Sohl T, Loveland TR, Sleeter BM, Sayler KL, & Barnes CA (2010) Addressing foundational elements of regional land-use change modeling. *Landscape Ecology* 25:233-247.

22. Beach RH, Zhang YW, & McCarl BA (2012) Modeling bioenergy, land use, and GHG Emissions with FASOM GHG: Model overview and analysis of storage cost implications. *Climate Change Economics* 2:1-34.

23. Golub AA & Hertel TW (2012) Modeling land-use change impacts of biofuels in the GTAP-BIO framework. *Climate Change Economics* 3:1250015.

24. Homer C, *et al.* (2007) Completion of the 2001 National Land Cover Database for the Conterminous United States. *Photo. Eng. Rem. Sens.* 73:337-341.

25. Radeloff VC, *et al.* (2012) Economic-based projections of future land use in the conterminous United States under alternative policy scenarios. *Ecol. Appl.* 22:1036-1049.

26. Smith JE, Heath LS, Skog KE, & Birdsey RA (2006) Methods for Calculating Forest Ecosystem and Harvested Carbon with Standard Estimates for Forest Types of the United States. (U.S. Department of Agriculture, Newtown Square, PA).

27. Bliss NB, Waltman SW, & West L (2009) Detailed mapping of soil organic carbon stocks in the United States using SSURGO [abs.], in Fall Meeting, San Francisco, CA, Dec. 14-18, 2009, Eos Transactions, Suppl., v. 90, no. 52: Washington, D.C., American Geophysical Union, p. B51F-0367.

28. Haim DA, Alig RJ, Plantinga AJ, & Sohngen B (2011) Climate change and future land use in the United States: an economic approach. *Climate Change Economics* 2:27-51.

29. U.S. Department of Agriculture-Agricultural Research Service (2011) USDA National Nutrient Database for Standard Reference, Release 24. Last accessed: December 20, 2013. http://www.ars.usda.gov/ba/bhnrc/ndl.

30. American Bird Conservancy (2012) List of the Birds of the United States with Conservation Rankings. Last accessed: December 19, 2013. http://www.abcbirds.org/abcprograms/science/conservationchecklist/index.html.

31. Comer P, *et al.* (2003) Ecological Systems of the United States: A Working Classification of U.S. Terrestrial Systems. (NatureServe., Arlington, VA).

Table 1: Description of Alternative Reference and Policy Scenarios

Scenario	Description	Targeted Services
Alternative Reference Scenarios		
1990s Trend	Continuation of land-use change trends from 1992 to 1997	NA
High Crop Demand	Land-use changes accounting for 10% increase in crop prices every five years relative to the 1990s Trend scenario	NA
Alternative Policy Scenarios		
Forest Incentives	$100/acre payment per year for land converted to forest; $100/acre tax per year for land taken out of forest	Timber production, carbon storage, habitat
Natural Habitats	$100/acre tax per year on land converted from forest or range to crop land, pasture, or urban	Habitat
Urban Containment	Prohibition on land conversion to urban in non-metropolitan counties.	Habitat, timber production, carbon storage, food production

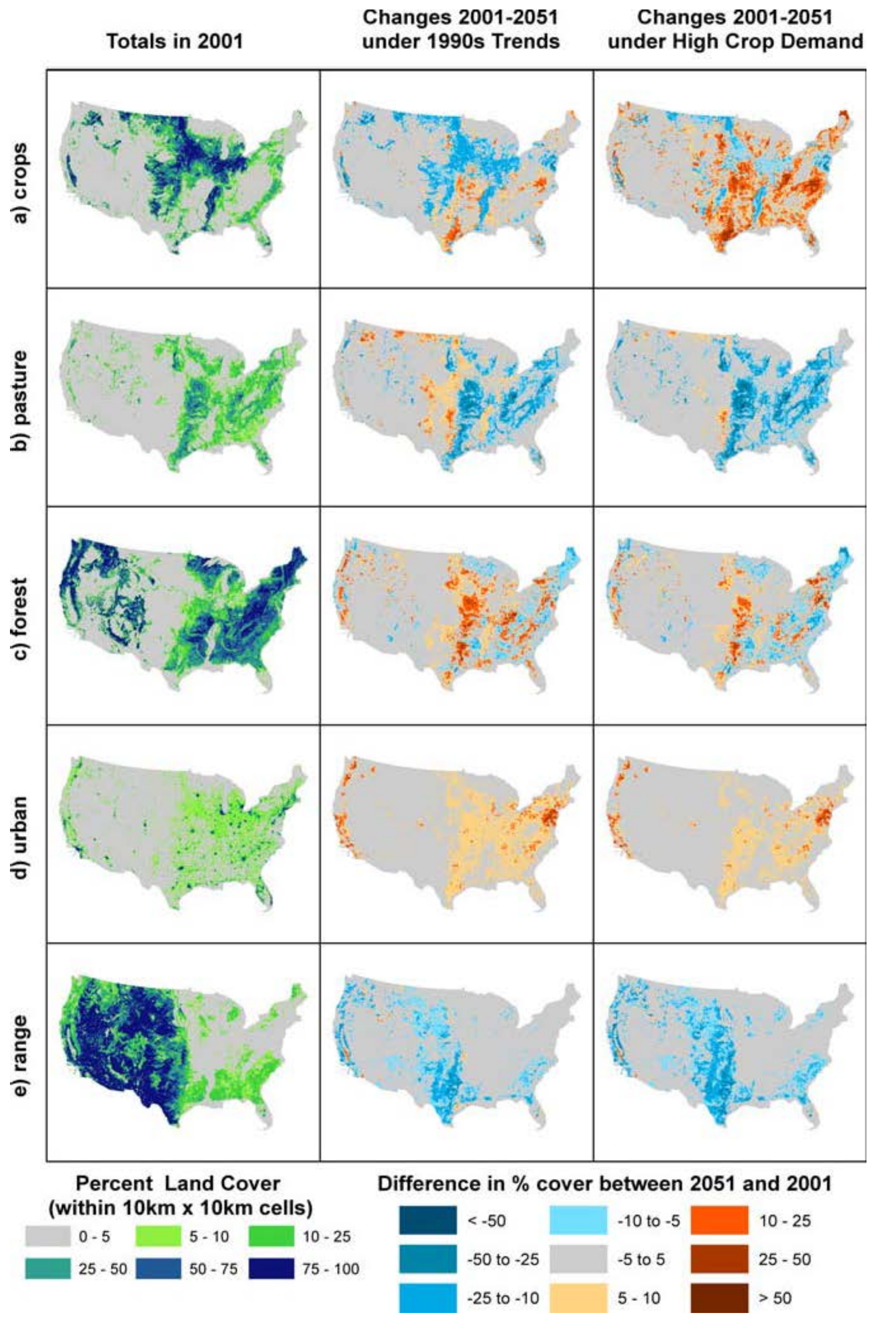
Figure Legends

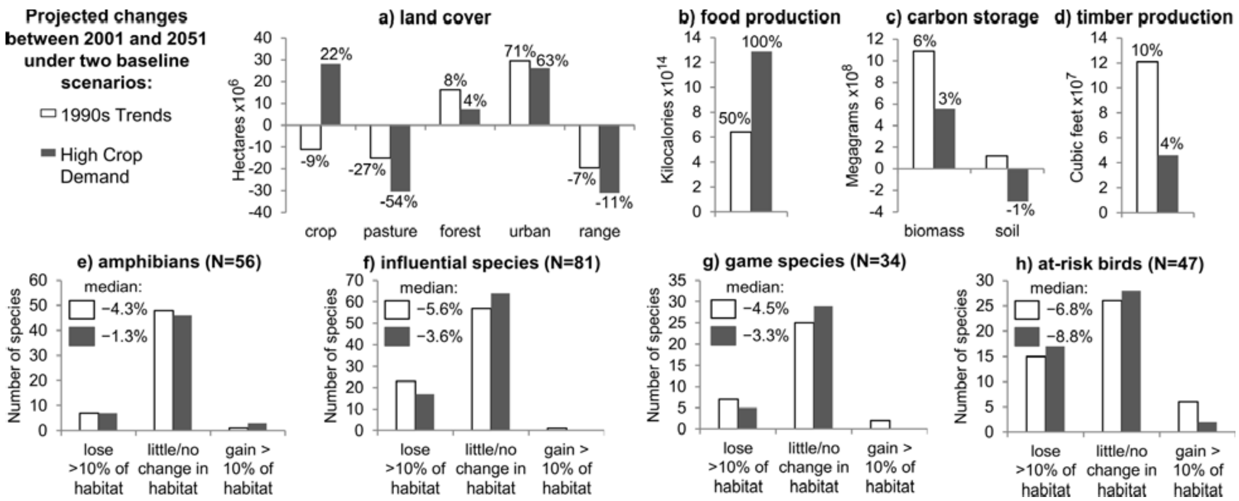
Figure 1. Spatial patterns in land cover in 2001 and changes between 2001 and 2051 under two baseline scenarios, 1990s Trends and High Crop Demand.

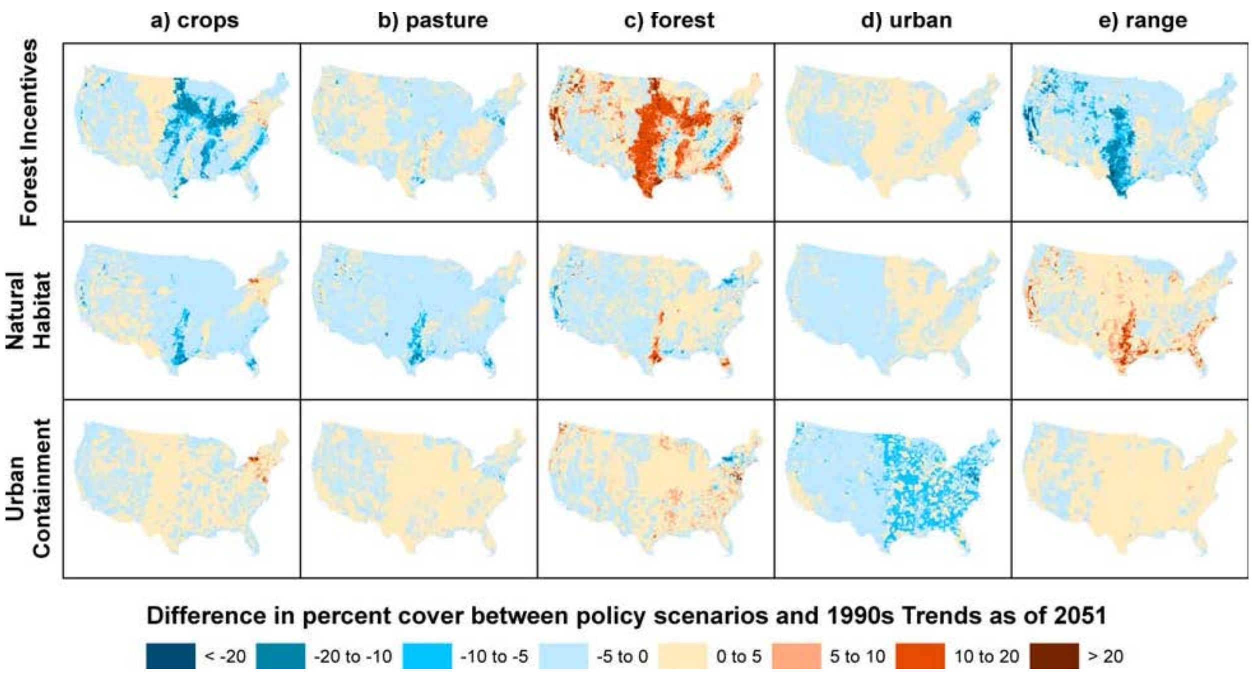
Figure 2. Projected changes between 2001 and 2051 under the two baseline scenarios for: a) land cover, b) food production, c) carbon storage, d) timber production, and area of prime habitat for different groups of wildlife species (e-h). The bars in figures (a) through (d) display the difference between 2051 and 2001 with labels for changes greater than 1%. Bars in figures (e) through (h) show the number of species in each of three categories: lose >10% of prime habitat area, little/no change in prime habitat area (-10% to +10%), and gain >10% in prime habitat area. In addition the median % change across species in each group, by baseline scenario, is shown in figures (e) through (h).

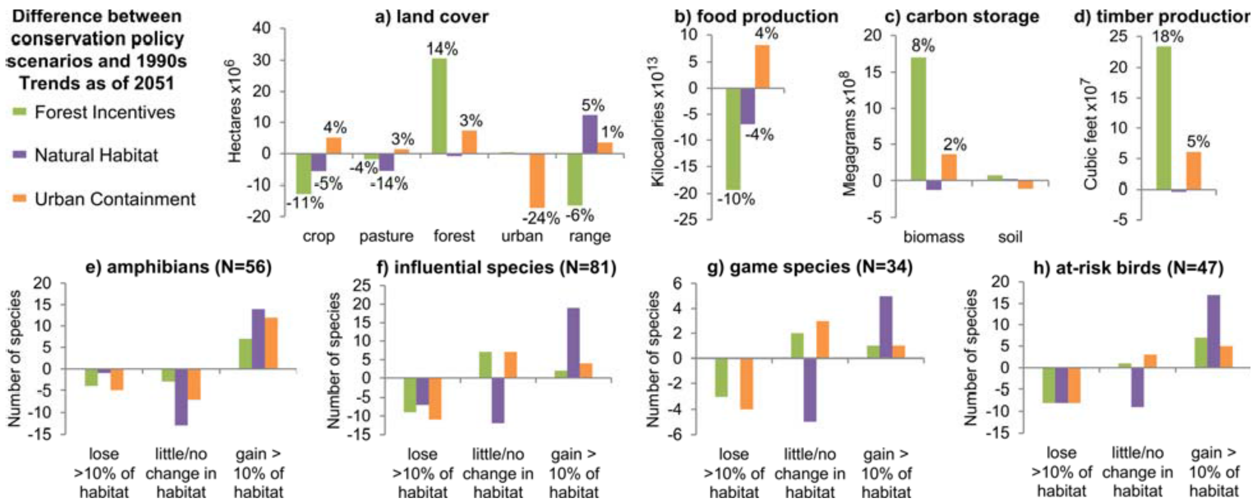
Figure 3. Spatial patterns in land cover changes under the three conservation policy scenarios (Forest Incentives, Natural Habitats, and Urban Containment) relative to projections based on the 1990s Trends baseline scenario.

Figure 4. Projected changes under the three conservation policy scenarios (Forest Incentives, Natural Habitats, and Urban Containment) relative to projections based on the1990s Trends scenario for: a) land cover, b) food production, c) carbon storage, d) timber production, and area of prime habitat for different groups of wildlife species (e-h). The bars in figures (a) through (d) display the difference between the policy scenarios and 1990s Trends projection as of 2051, with labels for changes greater than 1%. Bars in figures (e) through (h) show the increase or decrease in the number of species in the categories (defined in Figure 2) under each policy scenario compared to 1990s Trends baseline scenario.


Totals in 2001
Changes 2001-2051 under 1990s Trends
Changes 2001-2051 under High Crop Demand
a) crops
b) pasture
c) forest
d) urban
e) range
Percent Land Cover (within 10km x 10km cells)
0 - 5
5 - 10
10 - 25
25 - 50
50 - 75
75 - 100
Difference in % cover between 2051 and 2001
< -50
-50 to -25
-25 to -10
-10 to -5
-5 to 5
5 - 10
10 - 25
25 - 50
> 50


Projected changes between 2001 and 2051 under two baseline scenarios:
1990s Trends
High Crop Demand
a) land cover
Hectares x10^6
crop
pasture
forest
urban
range
22%
-9%
-27%
-54%
8%
4%
71%
63%
-7%
-11%
b) food production
Kilocalories x10^14
50%
100%
c) carbon storage
Megagrams x10^8
biomass
soil
6%
3%
-1%
d) timber production
Cubic feet x10^7
10%
4%
e) amphibians (N=56)
f) influential species (N=81)
g) game species (N=34)
h) at-risk birds (N=47)
Number of species
median:
−4.3%
−1.3%
−5.6%
−3.6%
−4.5%
−3.3%
−6.8%
−8.8%
lose >10% of habitat
little/no change in habitat
gain > 10% of habitat


a) crops
b) pasture
c) forest
d) urban
e) range
Forest Incentives
Natural Habitat
Urban Containment
Difference in percent cover between policy scenarios and 1990s Trends as of 2051
< -20
-20 to -10
-10 to -5
-5 to 0
0 to 5
5 to 10
10 to 20
> 20


Difference between conservation policy scenarios and 1990s Trends as of 2051
Forest Incentives
Natural Habitat
Urban Containment
a) land cover
Hectares x10^6
crop
pasture
forest
urban
range
b) food production
Kilocalories x10^13
c) carbon storage
Megagrams x10^8
biomass
soil
d) timber productior
Cubic feet x10^7
e) amphibians (N=56)
f) influential species (N=81)
g) game species (N=34)
h) at-risk birds (N=47)
Number of species
lose >10% of habitat
little/no change in habitat
gain > 10% of habitat

Supporting Information

SI Text

National-scale econometric land-use model

A national-level econometric land-use model (1) is used to project land-use changes on all privately owned parcels in the contiguous U.S. The model is based on the assumption that landowners will choose the land use that maximizes the present discounted value of the stream of expected net revenues from the land. Furthermore, it assumes that landowners base their expectations of future net revenues on current and historic values of relevant variables. Net revenues are defined as the quantity of the good produced from land multiplied by its price less the opportunity cost of all variable inputs to production. Given these assumptions, a simple decision rule emerges from the related dynamic optimization problem (2). In time *t*, the landowner chooses the use with the highest expected one-period net revenues at time *t* minus the current one-period expected opportunity cost of undertaking land-use conversion. Formally, the owner of a parcel in use *i* will transition from land-use *i* to *k* at time *t* if,

$$R_{kt} - C_{ikt} \geq R_{lt} - C_{ilt} \qquad \text{SI eq. (1)}$$

for all uses $l \neq k$, where R_{kt} and R_{lt} represent the expected net revenues at time *t* from a parcel of land in uses *k* and *l*, respectively, and C_{ilt} is the expected annualized cost of converting from use *i* to use *l* at time *t* where $C_{iit} = 0$. If the current use *i* satisfies SI eq. (1), then the parcel remains in that use at time *t*; otherwise, the landowner will reallocate the land to the use $k \neq i$ that maximizes expected net revenues minus conversion costs.

In practice, private land-use decisions can be influenced by factors other than market returns. For example, landowners may derive non-market benefits from their land (e.g., from recreation or aesthetics) or have historical ties to the land in particular uses (e.g., family-owned farms). Data are available to measure the net revenue variables (or construct suitable proxies) in SI eq. (1); however, these additional non-market factors are unobservable. As such, we model them as random disturbances and modify equation SI eq. (1) as follows,

$$R_{kt} - C_{ikt} + \varepsilon_{kt} \geq R_{lt} - C_{ilt} + \varepsilon_{lt} \qquad \text{SI eq.(2)}$$

where ε_{kt} and ε_{lt} are random variables associated with uses *k* and *l*, respectively. Because of the unobserved components, we can now make only probabilistic statements about land-use decisions. By imposing distributional assumptions on the random variables ε_{kt} and ε_{lt} (3), we obtain a parametric expression for the probability that a parcel in use *i* will be allocated to use *k* conditional on the net revenue and transition cost variables. The goal of the econometric modeling is to estimate the parameters of the land-use transition probabilities that best fit observed land-use change data. The estimation yields response functions indicating the probability of land-use changes conditional on economic variables.

The National Resources Inventory (NRI) is the primary data set we use to estimate a national land-use model (4). The NRI is a panel survey of land use and land characteristics on non-federal lands conducted across the periods 1982-87, 1987-92, and 1992-97 over the entire United States, excluding Alaska. Data include approximately 844,000 plot-level observations, each representing a land area indicated by a sampling weight. The econometric analysis focuses on land use change during the final period, 1992-1997, and on the contiguous U.S. and six major land uses as defined by the NRI: crops, pasture, forest, urban and built-up, range, and land enrolled in the federal Conservation Reserve Program. This land base comprises 1.4 billion acres, representing about 74% of the total land area and 91% of non-federal land in the contiguous U.S. Definitions of the land use categories are provided in (4). For simplicity, we refer to urban and built-up land as urban.

Distributional assumptions imposed on the random terms in SI eq. (3) yield a nested logit model for estimation (3). The dependent variable is the land-use choice in year *t* = 1997 at each NRI plot. The land-use transition probabilities are given by,

$$P_{jikt} = f\left(\boldsymbol{\beta}_{ik}, \mathbf{NR}_{jt}, \mathbf{LQ}_{j}\right) \qquad \text{SI eq. (3)}$$

for all j, i, k, and t, where j indexes the parcel, $\boldsymbol{\beta}_{ik}$ is a vector of parameters associated with the transition from use i to k from 1992 to 1997, $\mathbf{NR}_{jt}$ is a vector of net revenue variables for plot j in time t = 1997, and $\mathbf{LQ}_j$ is a vector of plot-level variables measuring land quality. By assembling data from a variety of private and public sources, Lubowski constructed county-level estimates of annual per-acre net revenues for crops, pasture, forest, range, and urban uses for all 3,014 counties in the contiguous U.S. Conversion costs are measured implicitly with constant terms specific to each i to k transition (5). The land-quality measure is an indicator variable for the land capability class rating (I to VIII) of NRI plots (6). We combine classes I and II, III and IV, V and VI, and VII and VIII to form four land quality categories. Land capability class rating variables indicate the productivity of the land for agriculture and are interacted with the net revenue and conversion cost variables to allow for plot-level deviations from the county average net revenue. Details on the estimation procedure and results are provided in (1,5).

We use an econometric model of land use change versus sectoral optimization models (e.g., the Forest and Agricultural Sector Model described in (7)) because the econometric approach can capture actual landowner behavior (e.g., the continuation of land-use over time that does not maximize expected net revenue) that cannot be represented in optimization models with perfectly rational decision-makers.

The advantage of the NRI data for econometric modeling is that it provides comprehensive and consistent data on private land use and plot attributes for the lower 48 states at multiple points in time. The main disadvantage of the NRI is that it does not provide exact information on the location of plots. Therefore, many spatial variables that plausibly affect land use, such as distance to cities and the land use choices of neighboring parcels, cannot be included in the model. This problem is mitigated somewhat by the fact that plots close to large and growing cities are also likely to be in a county with high urban net revenues, an included variable in our model. Nevertheless, an important extension of our model would be to use national spatial-temporal data on land-use change as a means to model explicit spatial processes.

Land use transition matrices

The parameter vector $\boldsymbol{\beta}_{ik}$ is estimated with a nested logit model and then is substituted into SI eq. (3) to yield probabilities of land-use transitions among potential land uses as a function of net revenue and land-quality variables. Therefore, when the response functions are evaluated at the economic and plot-level variables, we obtain a transition probability matrix ($\mathbf{P}_{jt}$) for each NRI plot j and time t. Define the vector $\mathbf{A}_{jt}$ as the number of acres in each of the six land uses in NRI plot j in time t = 1997. Then, the number of acres in each use N 5-year time steps hence is given by,

$$\mathbf{A}_{jt+N} = \mathbf{A}_{jt} \times \mathbf{P}_{jt}^{N} \qquad \text{SI eq. (4)}$$

Each period is five years in length to correspond to the time-step in the NRI data. Because we do not observe land moving out of urban uses, the associated transition probabilities for land beginning in urban use always equal 0. Further, all publicly-owned land is assumed not to change use over time.
The changes in land use over time affect the supply of commodities, commodity prices, and thus the net revenues from each use of the land. Therefore, $\mathbf{P}_{jt}$ for t = 2002 does not equal $\mathbf{P}_{jt}$ for t = 1997, $\mathbf{P}_{jt}$ for t = 2007 does not equal $\mathbf{P}_{jt}$ for t = 1992, etc. Our mechanism for modeling these price feedbacks works as follows. Landowners with static expectations observe the level of net revenues to all alternative uses in period t. With static expectations, the optimal decision at each point in time is to convert to (or remain in) the use with the highest annual net revenue less conversion costs (2). Therefore, landowners are not assumed to anticipate any changes in net revenues induced by future conversions. In the next decision period (t+5), net revenues are updated to account for price and yield changes that occurred between t and t+5. Price changes occur because of shifts in land use, which change the supply of outputs from the land and, therefore, prices for these outputs. Once net revenues are updated to account for these supply shifts, landowners repeat the same decision process that began in period t. This price feedback approach is not a true general equilibrium model since we have no mechanism that simultaneously clears all markets. Rather, our model represents a disequilibrium in land markets together with a price-feedback mechanism that pushes the landscape towards equilibrium. To better understand this feature of our model, consider that landowners respond to the relative levels of net revenues. For example, if there is a large difference between the levels of crop and forest net revenues, landowners may be induced to move more land into

crops because this difference signals a more profitable opportunity. Such changes in land uses affect output and prices and, thus, the levels of net revenues, signaling new opportunities for profitable changes in land use. This process continues until such opportunities are exhausted, which implies no further changes in land use or the levels of net revenues. At this point, markets would be in equilibrium. In our simulations, equilibrium is never actually achieved, though the price-feedback mechanism moves the landscape toward equilibrium.

The endogenous adjustments in net revenues are made using econometrically estimated demand elasticities selected after an exhaustive review of the literature. Lubowski et al. discuss the elasticities used for forest and crop prices (1). For pasture and rangeland, we were unable to find any estimates of forage demand elasticities and assumed that the crop price elasticity used in Lubowski et al. applied to pasture and rangeland revenues. Thus, a 1% increase in a county's pasture or range acreage was assumed to result in a 1.51% decrease in pasture or range revenues in that county. The adjustments to urban net revenues are made using elasticity estimates from Haim (8). Unlike existing studies of urban land markets that used metropolitan area data or focused on single cities, Haim's analysis was conducted using data on all US counties. Because he used a log-level specification, elasticity estimates are proportional to net revenues in each county. For example, in a county with an average urban net revenue of $10000/acre, a 1% increase in urban acreage results in 0.47% decrease in the urban net revenue; i.e., $0.47=(0.000212\times10000)^{-1}$. With the exception of the elasticity for urban land, all demand elasticities in the model are assumed to remain constant over time. In practice, factors such as income and population growth, changes in substitute goods, and shifting demographics may cause elasticities to change over time. Econometric analysis outside the scope of this study would be needed to accommodate time-varying elasticities. See Table SI 1 for all elasticity measures and information on how they are used to update prices and net revenues as a function of shifts in the supply of land use.

Two additional modifications to the modeling approach described above are that crop yields increase by 6% each period (which directly impact agricultural rents and create a feedback on prices), reflecting historical patterns (9), and increases in forest are prevented in regions where climatic conditions severely restrict forest growth. To define these regions, we use maps of Holdridge Life Zones (10) to distinguish between forest zones (e.g., cool temperate moist forest) and non-forest zones (e.g., warm temperate montane steppe). Ideally we would model crop yield as a function of farmer managerial response to crop prices rather than our approach of assuming an exogenous increase in yields. However, adding a model of land use intensity (e.g., how much fertilizer is applied, is irrigation water used, etc.) as a function of crop prices is a non-trivial extension that we leave to future work.

After SI eq. (4) has been used to estimate area of land by use and land quality class in each NRI plot *j* at each five-year time step ($\mathbf{A}_{jt+1}, \mathbf{A}_{jt+2}$, etc.), the area in each land use and land quality category combination are aggregated to the county level for each time step. Land in crops and the CRP are combined into one category at this stage, cropland, because the land use map used for grid-cell level projections of land use (see below) does not separately identify these categories. Finally, all the modeled changes in land use from 1997 to 2047, now aggregated at the county-level, are used to construct 50-year transition probability matrices for each county, land quality combination.

The NRI dataset does not give the exact location of land use. Therefore, to create spatially explicit maps of future land use at the grid cell level, a level of detail necessary for many ecosystem models including our habitat model, we apply the transition probability matrices to a baseline map that defines land use and land quality for every grid cell in the US. Unfortunately, a contiguous US grid cell map of land use is not available for *t* = 1997, the beginning point of our estimated transition matrices. Grid cell maps of national land cover data (NLCD) for the contiguous US are only available for the years 1992, 2001, and 2006 (a time invariant grid cell level map of land quality can be used in combination with any of these maps). We use the 2001 NLCD (11) map as our base year for two reasons: 1) the 2006 map is temporally more distant from the land use change data (1992-1997) used to estimate the econometric model than the 2001 map and 2) the US public land map we use is available for the early 2000s but not 1992 (http://protectedareas.databasin.org/).

To summarize, we use the two different data sets (NRI 1992-1997 and NLCD 2001) for two different purposes. We use the NRI to estimate the econometric model of land use change (equations SI eqs. (3) and (4)) and then to construct the 50-year transition matrices. We then apply these transition matrices to the 2001 NCLD to define a grid cell level map of US land use in 2051.

Policy scenarios

We use the projection model described above to simulate several land-use policies. Market-based incentives, including subsidies and taxes, are introduced by modifying the net revenue measures in SI eq. (3). For example, to simulate a per-acre subsidy *S* for afforestation, we add *S* to the net revenue from forests in the case of all transitions from non-forest uses (crop, pasture, CRP, and range) to forest. An afforestation subsidy increases the amount of land in forest. However, this increase in timber supply also depresses timber prices slightly. In addition, this subsidy engenders reductions in the supply of commodities from cropland, pasture, and ranges, raising the net revenues from these uses. Thus, the model captures policy feedbacks on market prices, and therefore, the incentives for changes in land use.

We evaluate two alternative reference scenarios and three policy scenarios. The first reference scenario (1990 Trends) reflects a continuation of economic conditions during the 1990 decade, the period of the data used to estimate the econometric model. The alternative reference scenario (High Crop Demand) assumes an exogenous increase in crop prices of 10% every five years for every crop type. The scenario also assumes continued support for the CRP at current levels, and so no land is allowed to enter or exit this program. In the Forest Incentives policy scenario, we provide a \$100 per acre per year subsidy for afforestation and levy a \$100 per acre tax per year on land leaving forest. Such a policy can be motivated by a policy goal of increasing carbon sequestration in forests. Based on results in (1), this translates into a carbon tax/subsidy of about \$50/ton of carbon. The Native Habitats policy is designed to retain land in less intensive uses (forest and range). A \$100 per acre tax per year is levied on land leaving forest and range, including transitions between these two uses. The tax remains with the parcel that leaves forest or range until it afforests or becomes rangeland again. Finally, the Urban Containment scenario limits urban expansion to metropolitan counties (counties that form a metropolitan statistical area, as defined by the U.S. Office of Management and Budget). This policy mimics zoning regulations that limit urban expansion in rural areas.

Finally, the use of scenarios allows us to gauge the sensitivity of our results to different assumptions about the underlying drivers of land-use change on the provision of ecosystem services. Scenarios are used in place of a formal treatment of parameter and data uncertainty, which is infeasible given the scale of the modeling exercise.

Biomass carbon storage in private forests

For each forest stand type *m* found in county *c* we use the US Forest Inventory Analysis (FIA) datatset to find the Faustmann volume, given by V_{cm}^F. The economic value from harvesting a stand is maximized if it occurs at the stand's Faustmann volume. The function $V_{cm}(t)$ gives the volume in an acre of stand type *m* in county *c* at stand age *t* (12). Let t_{cm} be the stand age that solves $V_{cm}(t) = V_{cm}^F$. If an exact *t* that solves $V_{cm}(t) = V_{cm}^F$ cannot be found we set t_{cm} equal to the *t* that minimizes $V_{cm}(t_{cm}) - V_{cm}^F$ subject to the difference being positive.

Let $B_{cm}(t)$ indicate the metric tons of biomass carbon found in an acre of stand type *m* at age *t* in county *c* (a stand's biomass carbon includes carbon stored in live aboveground, live belowground and dead woody biomass and detritus on the forest floor) (12). Let $\Delta B_{cm} = {(B_{cm}(t_{cm}) - B_{cm}(0))}/{t_{cm}}$ indicate the average annual gain in carbon storage in an acre of *c*, *m* over t_{cm} years of growth.

If a stand of type *m* in county *c* is in even-age rotation then $1/t_{cm}$ of the stand's area has just been harvested, $1/t_{cm}$ of the stand is comprised of one-year old trees, $1/t_{cm}$ of the stand is comprised of two-year old trees, etc. The last $1/t_{cm}$ of the stand is comprised of $1/t_{cm} - 1$ year old trees. Assume each $1/t_{cm}$ portion of the stand is one acre. The total biomass carbon stored over the t_{cm}-acre even-age rotation stand of type *m* in county *c* at any point in time is given by S_{cm},

$$S_{cm} = \sum_{i=0}^{t_{cm}-1} \left((i \times \Delta B_{cm}) + B_{cm}(0) \right)$$

SI eq. (5)

and the average per acre storage in the stand is $\bar{S}_{cm} = S_{cm}/t_{cm}$. If we assume all private forest land in a county is in even-age rotation then the tons of biomass carbon stored in a representative hectare of private forest land in county *c*, given by $\bar{S}_c$, is equal to the weighted average of $\bar{S}_{cm}$ values across all stand types found in county *c*,

$$\bar{S}_c = 2.471 \times \frac{\sum_{m=1}^{M} A_{cm} \times \bar{S}_{cm}}{\sum_{m=1}^{M} A_{cm}} \qquad \text{SI eq. (6)}$$

where A_{cm} is the area of stand type *m* found in county *c* during the period of US FIA dataset compilation and 2.471 is the constant that converts per acre to per hectare measures.

Let $A_{c,f,2001}$ and $A_{c,f,2051}$ indicate the hectares of private forest in county *c* in 2001 and 2051, respectively. We set biomass carbon stored in private forest land in each county in 2001 and 2051 equal to $A_{c,f,2001} \times \bar{S}_c$ and $A_{c,f,2051} \times \bar{S}_c$, respectively. Note that we credit each hectare of private forest in 2001 and 2051 with its county's even-age rotation biomass carbon measure.

Biomass carbon storage in public forests

In public forests we do not assume trees are managed in a rotation system. Instead we assume that every stand of public forest has trees with an average age of *t*. Recall that $B_{cm}(t)$ is the function that gives the biomass carbon expected in an acre of stand type *m* at age *t* in county *c*. Therefore, the biomass carbon stored in a hectare of public forest in county *c*, given by $\bar{E}_c$, is equal to the weighted average of $B_{cm}(t)$ values across all stand types found in county *c*,

$$\bar{E}_c = 2.471 \times \frac{\sum_{m=1}^{M} A_{cm} \times B_{cm}(t)}{\sum_{m=1}^{M} A_{cm}} \qquad \text{SI eq. (7)}$$

where A_{cm} the area of stand type *m* found in county *c* during the period of US FIA dataset compilation. In SI eq. (7) we set *t* = 70 years for all *c* and *m* combinations.

Let $A_{c,pf,2001}$ and $A_{c,pf,2051}$ indicate the hectares of public forest in county *c* in 2001 and 2051, respectively. In our model, $A_{c,pf,2001} = A_{c,pf,2051}$ Therefore, biomass carbon stored in private forest land in each county in 2001 and 2051 is given by $A_{c,pf,2001} \times \bar{E}_c$.

Biomass carbon storage in other land use types

We assume that a hectare of private urban land use in county *c* has biomass carbon equal to 10% of the biomass carbon found in a hectare of the county's even-age rotation private forests,

$$\bar{U}_c = 0.1 \times \bar{S}_c \qquad \text{SI eq. (8)}$$

Let $A_{c,u,2001}$ and $A_{c,u,2051}$ indicate the hectares of urban land in county *c* in 2001 and 2051, respectively. Therefore, biomass carbon stored in urban land in each county in 2001 and 2051 is given by $A_{c,u,2001} \times \bar{U}_c =$ and $A_{c,u,2051} \times \bar{U}_c$, respectively.

We assume cropland and pasture have biomass carbon steady state values of 0. While annual crops sequester carbon over the growing season most of this storage is lost to the atmosphere at or soon after harvest. Some of the crop biomass may be cycled into the soil but this dynamic is accounted for in the soil carbon pool (See below). Perennial crop operations, especially woody perennial crops such as apple or orange farms, could reach a rotational biomass carbon steady state similar to private forests. But we do not have a nation-wide database that describes the carbon dynamics of the various perennial crop operations in the US. Therefore, we do not include perennial crop biomass carbon processes in our model. Similarly pasture will produce grasses that sequester carbon. However, this storage may be temporary as the grass is eaten by animals or converted to hay that is fed to animals soon after harvest. Further, left over grass will die in winter in some areas of the U.S. Some carbon stored in the grass will migrate to the soil carbon pool; however, this process is accounted for in our soil carbon model (see below). We do not account for carbon stored in the root systems of pasture. On some pasture types this storage capacity can be substantial.

We also assume rangeland has a biomass carbon storage steady state value of 0. This assumption is problematic given that rangeland includes land with scrub-shrub covers and other woody biomass features. However, because we do not have a nation-wide database that describes the carbon dynamics of the various covers that make up the rangeland category we do not include rangeland biomass carbon processes in our study. Again we do not account for carbon stored in the root systems of grasslands in the rangeland category.

Soil carbon storage

We overlaid the 2001 NLCD (11) and a U.S. county map on a map of soil carbon data (13) to determine the average mass of soil carbon stored on a hectare of each NLCD land use/land cover (LULC) category in each US county. Soil carbon is measured to a depth of 30 cm. We then cross-walked the NLCD LULC categories with the NRI land use categories to create a county-level dataset of average soil carbon mass stored in a hectare of each of the 5 modeled land uses (see Table SI 2 for crosswalk). Let L_{ck} indicate the average mass of carbon stored in the first 30 cm of soil on a hectare of land use k in county c.

Let $A_{c,k,2001}$ and $A_{c,k,2051}$ indicate the number of hectares (private and public) in county c in land use k in 2001 and 2051, respectively. The carbon stored in 30 cm of soil in county c in 2001 is given by

$$\sum_{k=1}^{5} A_{c,k,2001} \times L_{ck}$$

where k indexes cropland (k = 1), pasture (k = 2), forest (k = 3), urban (k = 4), and range (k = 5). The carbon stored in 30 cm of soil in county c in 2051 is given by $\sum_{k=1}^{5} A_{c,k,2051} \times L_{ck}$. We do not dynamically track the sequestration of carbon in soil. Instead we assign each 2051 hectare of land use type k the average level of carbon storage observed in that land use type circa 2001.

Kilocalorie production

Let the 2001 per acre yield of crop type j in county c be given by Y_{jc} (8). Let the constant that converts a unit of crop yield k into grams be given by G_j (http://www.futures101.ru/wp-content/uploads/2010/03/conversion.pdf). For example, wheat yield is given in bushels. A bushel of wheat has a mass of 27216 grams. Therefore, G_{wheat} = 27216 grams. Let K_j measure the kilocalories in a gram of crop j. For example, each gram of wheat contains 3.39 kilocalories. See Table SI 3 for the list of crop types and G_j and K_j for each crop type (14). Let O_{jc} indicate the millions of kilocalories produced per acre of crop j in county c in 2001,

$$O_{jc,2001} = \frac{Y_{jc} \times G_j \times K_j}{1{,}000{,}000} \qquad \text{SI eq. (9)}$$

Next we calculate an average per hectare kilocalorie production value in 2001 for each county. Let F_{jc} indicate the fraction of county c's private cropland in crop j (9). Therefore, O_c, county c's weighted average kilocalorie production per hectare of private cropland in 2001, is given by,

$$O_{c,2001} = 2.471 \times \sum_{j=1}^{J} F_{jc} \times O_{jc,2001} \qquad \text{SI eq. (10)}$$

where the constant 2.471 converts per acre values to per hectare values and $O_{c,2001}$ is measured in millions of kilocalories.

We assume a 6% increase in crop yield every 5 years from 2001 to 2051 across all crop types and all counties in the US. We assume F_{jc} stays fixed for each j and c combination from 2001 to 2051. Therefore, 2051 kilocalorie production per hectare of private cropland in county c be given by,

$$O_{c,2051} = O_{c,2001} \times 1.06^{10} \qquad \text{SI eq. (11)}$$

where $O_{c,2051}$ is measured in millions of kilocalories.
Let $A_{c,c,2001}$ and $A_{c,c,2051}$ indicate the hectares of private cropland in county *c* in 2001 and 2051, respectively. Therefore, kilocalorie production in each county in 2001 and 2051 is given by $A_{c,c,2001} \times O_{c,2001}$ and $A_{c,c,2051} \times O_{c,2051}$, respectively.

Timber production

Let D_c indicate the expected yield of timber in county *c* where yield is measured in thousand cubic feet (mcf) per hectare (6). We assume that D_c does not change over time. Let R_c indicate the weighted average rotation length of private forest in county *c*. R_c is a function of A_{cm} and t_{cm} over all *m*,

$$R_c = \frac{\sum_{m=1}^{M} A_{cm} \times t_{cm}}{\sum_{m=1}^{M} A_{cm}} \qquad \text{SI eq. (12)}$$

where A_{cm} and t_{cm} are defined in the "Biomass carbon storage in private forests" section of the SI Text. Timber production in county *c* in any given year is equal to $P_c = D_c / R_c$ where we divide by R_c to account for the fact that each year only $1 / R_c$ of forest area in county *c* is harvested (recall our assumption that all private forests are in steady-state rotation at all times).
Let $A_{c,f,2001}$ and $A_{c,f,2051}$ indicate the hectares of private forest in county *c* in 2001 and 2051, respectively. Therefore, total timber production in each county in 2001 and 2051, measured in thousands of cubic feet, is given by $A_{c,f,2001} \times P_c$ and $A_{c,f,2051} \times P_c$, respectively. Note that we credit each hectare of private forest in 2001 and 2051 with its even-age rotation timber harvest.
We do not track timber harvest on public land. By 2002 harvest on public lands had fallen to less than 8% of the country's annual harvest (15). Therefore, we do not feel our estimates of alternative trends in timber harvest across the contiguous U.S. are invalidated by ignoring public land harvest over time. Finally, note that we do not account for the carbon dioxide fertilization effect in our study. If this phenomenon is real than we underestimate timber production.

Ecosystem service maps

In Figure SI 1 we map the expected 2001 to 2051 changes in biomass carbon across the contiguous U.S. under the 1990s Trends and High Crop Demand scenarios. In Figure SI 1 we also map the differences in expected 2051 biomass carbon storage across the 3 policy scenarios.

In Figure SI 2 we map the expected 2001 to 2051 changes in soil carbon across the contiguous U.S. under the 1990s Trends and High Crop Demand scenarios. In Figure SI 2 we also map the differences in expected 2051 soil carbon storage across the 3 policy scenarios.

In Figure SI 3 we map the expected 2001 to 2051 changes in kilocalorie production across the contiguous U.S. under the 1990s Trends and High Crop Demand scenarios. In Figure SI 3 we also map the differences in expected 2051 kilocalories production across the 3 policy scenarios.

Species habitat

To assess species responses to land-use change we quantify changes in the amount of habitat for 194 terrestrial vertebrate species. All of the chosen species have ecological or social importance. The chosen species represent four major groups*: amphibians* (n = 56); *influential species* (including top carnivores and ecosystem engineers, n = 81); *game species* (n = 34), and *at-risk birds* (categorized by the American Bird Conservancy (2012) as 'vulnerable' or 'potential concern,' n = 47; see Dataset SI 1 for the complete species list). We quantify potential habitat area for each species on the 2001 and 2051 maps by using species' geographic range maps and known habitat associations. We assume that species' geographic ranges and habitat associations will remain static despite the potential of climate change to alter both (16). For each species and species group, we compare the projected change in habitat area from 2001 to 2051 under the 1990 Trends and High Crop Demand scenarios and differences in 2051 habitat across policy scenarios.
We obtained digital range maps for 42 mammals (17), 108 birds (18), and 56 amphibians (data available

online at http://www.iucnredlist.org/technical-documents/spatial-data). For bird ranges, we only use the portions that supplied breeding or year-round residency. For amphibians, we use only species that had significant use of uplands (terrestrial) habitat during their life cycle, i.e. not exclusively aquatic species or wetlands specialists.

To derive species-habitat associations we use a land-cover classification of 144 ecological systems from NatureServe (19), generalized from 30-m to 100-m pixels to match our other spatial datasets, in conjunction with expert opinion. Ecological systems represent “recurring groups of biological communities that are found in similar physical environments and are influenced by similar dynamic ecological processes, such as fire or flooding” (19). Descriptions of the ecological systems and data layers for the conterminous U.S. are available online (www.natureserve.org). We do not use the finest-level classification of ecological systems available (n = 544), but rather groupings of those systems at the macrogroup level as provided by NatureServe (Table SI 4).

To model transitions from one land-use type to a new forest or range land-use type based on the coarse NRI classes used in the econometric model, we use a potential vegetation dataset known as “biophysical settings” from NatureServe. Biophysical settings represent “the vegetation that may have been dominant on the landscape prior to Euro-American settlement,” taking into account historical disturbance regimes (see www.landfire.gov). Since the biophysical settings vegetation classification is based on NatureServe’s ecological systems, we used the same groupings of ecological systems at the macrogroup level (Table SI 4) for potential vegetation as we did for current land cover.

To be compatible with the outputs of the econometric model, we had to ensure that each NRI class (water, crops, pasture, forest, urban and built-up, and range) could be mapped as an appropriate ecological system. First we assign each NRI class (0:5) to the appropriate NLCD Anderson Level 1 class (1:8) and NLCD 2001 land cover class (11:95). This is done by examining current land cover under each classification (Table SI 5). The NRI class ‘water’ includes both wetlands classes as well as ‘barren lands’ in the NLCD 2001 classification. Given the crosswalk shown in Table SI 5 it is relatively straightforward to assign each of our 144 land-cover classes to one of the NLCD 2001 classes, and therefore the appropriate NRI class. A potential problem arises with clear-cut forests, which are likely to be classified as forest in the NRI and as shrub/scrub or grassland/herbaceous in the NLCD. As indicated in Table 1, shrub/scrub and grassland/herbaceous are matched to the NRI rangeland category. Thus, in some cases we will apply the transition probabilities for land starting in rangeland to land that is properly categorized as forest. If there is any ambiguity we use the ecological systems definition(s) to determine the appropriate NLCD 2001 class.

We use the NLCD 2001 grid as the basis for our “current” land cover layer, but with one of our 144 land-cover codes assigned to each 30-m resolution grid cell. To be consistent with land-use and carbon modeling layers, we built a 100-m resolution by taking the most-common land-cover code in the 30-m cells that a single 100-m cell overlapped.

We use expert opinion (the authors and NatureServe staff) to identify the ecological systems considered to be “prime habitat”, defined as those ecological systems use for foraging and/or reproduction and expected to support population growth over time on its own. Our estimates of habitat area and habitat change in this paper are based on prime habitat only.

We quantify the species’ habitat area in 2001 using the following calculation,

$$HV_s^p = \sum_{1}^{j} P_j \qquad \text{SI eq. (13)}$$

where HV^p is the current prime habitat area for species *s*, and P_j is the total area (in the range of species *s*) of land-cover type *j* of the list of land cover types considered to be prime habitat for species *s*.

For habitat area in 2051 we use,

$$HV_s^p = \sum_{1}^{y} \left\{ \sum_{i=1}^{5} \sum_{1}^{j} p_i C_j + \sum_{i=1}^{5} \sum_{1}^{j} p_i T_j \right\} \qquad \text{SI eq. (14)}$$

where HV^p is the scenario-specific prime habitat value for species *s*, *y* is the total of all pixels in the range of species s, p_i is the probability of land cover at a pixel being type i 50 years in the future, $C_j = 1$ if i is the *current* land cover type at pixel y and land-cover type j is on the prime habitat list for species s, while $T_j = 1$ if i represents a transition to a new land-cover type and land-cover type j is on the prime habitat list for species s and is assigned to pixel y according to the rule for new land cover.

The 100-m grid with ecological system values described above are used to calculate the current amount of habitat available to each species on our list, based on a simple sum of the area of all macrogroups on the prime habitat list for each species. Because future land cover is based on the probability of any given grid cell staying the same (NRI-based) land cover, or transitioning to another land cover type, if the land cover stayed the same we used the habitat value (if any) assigned to the ecological system in the current land-cover grid. If the land cover represented a transition to a new land cover we use the rules shown in Table SI 6 to assign a habitat value to the grid cell.

For each scenario of projected land-use change, any individual pixel y in the range of species s can have a potential habitat value from 0 to 1 (the term inside the brackets of SI eq. (14)). We create a grid with continuous pixel values of 0 to 1 across a species' range for projected habitat values in order to calculate overall habitat amounts and changes. Changes in habitat are calculated for each species comparing the scenario-specific habitat values to current values, and comparing those changes under each policy scenario to the amount of change under the baseline scenario.

A note on climate change

In our analyses, changes in land use and land use productivity (e.g., crop yield, timber yield, biomass carbon sequestration rates) are not a function of expected climate change. Haim et al. (9) found that land-use choices will not be greatly affected by climate change between 2001 and 2051 across the US. However, it is fairly clear that land-use productivity and vegetative cover will be affected by climate change (20-21). There is also evidence to suggest that many tree species will migrate and forest productivity in the U.S. will change due to climate change. For example, Kirilenko and Sedjo (22) suggest that productivity in many forests will increase as a result of CO_2 fertilization. We do not account for the fertilization effect in our study, which would raise our estimates of timber production and carbon sequestration in forests. However, increases in productivity due to fertilization could be offset by climate change factors such as increasing frequency of forest fires (23). Finally, species ranges will change as the climate changes with many species shifting their distributions to track suitable climates (24). Incorporating climate-change impacts is an important next step in providing more realistic future projections.

References

1. Lubowski, R.N., A.J. Plantinga, and R.N. Stavins. 2006. Land-use change and carbon sinks: Econometric estimation of the carbon sequestration supply function. *Journal of Environmental Economics and Management* 51(2):135-152.

2. Plantinga, A.J. 1996. The effect of agricultural policies on land use and environmental quality. *American Journal of Agricultural Economics* 78(4):1082-1091.

3. Train, K. 2003. *Discrete Choice Methods with Simulation*. New York: Cambridge University Press.

4. U.S. Department of Agriculture (USDA). 2009. *Summary Report: 2007 National Resources Inventory*, Natural Resources Conservation Service, Washington, DC, and Center for Survey Statistics and Methodology, Iowa State University, Ames, Iowa.

5. Lubowski, R.N. 2002. Determinants of land-use transitions in the United States: econometric analysis of changes among the major land-use categories. Ph.D. Dissertation, Harvard University.

6. U.S. Department of Agriculture. 1973. Land Capability Classification. Soil Conservation Service Agricultural Handbook No. 210, Washington, DC.

7. Alig, R.J., D.M. Adams, and B.A. McCarl. 1998. Impacts of incorporating land exchanges between forestry and agriculture in sector models. Journal of Agricultural and Applied Economics 30(2): 389–401.

8. Haim, D. 2011. Three essays on the economics of climate change, land use and carbon sequestration. Ph.D. Dissertation, Oregon State University.

9. Haim, D., R.J. Alig, A.J. Plantinga, and B. Sohngen. 2011. Climate change and future land use in the United States: An economic approach. *Climate Change Economics* 2(1):27-51.

10. Lugo, A.E., S.-L.Brown, R. Dodson T.-S. Smith, and H.H. Shugart. 1999. The Holdridge Life Zones of the conterminous United States in relation to ecosystem mapping. *Journal of Biogeography* 26(5):1025-38.

11. Homer, C., C. Huang, L. Yang, B. Wylie, and M. Coan. 2004. Development of a 2001 National Landcover Database for the United States. *Photogrammetric Engineering and Remote Sensing* 70(7):829–840.

12. Smith, J.E., L.S. Heath, K. Skog, and R. Birdsey. 2006. Methods for calculating forest ecosystem and harvested carbon with standard estimates for forest types of the United States. Gen. Tech. Rep. NE-343. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northeastern Research Station.

13. Bliss, N.B., S.W. Waltman, and L. West. 2009. Detailed mapping of soil organic carbon stocks in the United States using SSURGO [abs.], *in* Fall Meeting, San Francisco, CA, Dec. 14-18, 2009, Eos Transactions, Suppl., v. 90, no. 52: Washington, D.C., American Geophysical Union, p. B51F-0367.

14. U.S. Department of Agriculture, Agricultural Research Service (USDA-ARS). 2011. USDA National Nutrient Database for Standard Reference, Release 25. http://www.ars.usda.gov/Services/docs.htm?docid=22771

15. Adams, D., R. Haynes, and A.Daigneault. 2006. Estimated timber harvest by U.S. region and ownership, 1950-2002. Gen. Tech. Rep. PNW-GTR-659. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

16. Lawler, J.J., E. Nelson, M. Conte, S.L. Shafer, D. Ennaanay, and G. Mendoza. 2011. Modeling the impacts of climate change on ecosystem services, in *Natural Capital: Theory and Practice of Mapping Ecosystem Services*, P. Kareiva, H. Tallis, T. Ricketts, G. Daily, and S. Polasky. New York: Oxford University Press.

17. Patterson, B.D., et al. 2003. Digital Distribution Maps of the Mammals of the Western Hemisphere, version 1.0. NatureServe, Arlington, Virginia.

18. Ridgely, R.S., et al. 2003. Digital Distribution Maps of Birds of the Western Hemisphere, version 1.0. NatureServe, Arlington, Virginia.

19. Comer, P., et al. 2003. Ecological Systems of the United States: a working classification of U.S. terrestrial systems. NatureServe, Arlington, Virginia.

20. Lobell, D., W. Schlenker, and J. Costa-Roberts. 2011. Climate trends and global crop production since 1980. Science 333: 616-620.

21. Nelson, Erik J., et al. 2013. Climate change's impact on key ecosystem services and the human well-being they support in the US. *Frontiers in Ecology and the Environment* 11: 483-893.

22. Kirilenko, A.P., and R.A. Sedjo. 2007. Climate change impacts on forestry. *Proceedings of the National Academy of Sciences* 104(50): 19697-19702.

23. Westerling, A.L., H.G. Hidalgo, D.R. Cayan and T.W. Swetnam. 2006. Warming and earlier spring increase Western U.S. forest wildfire activity. *Science* 313: 940-943.

24. Lawler J.J., S.L. Shafer, D. White, P. Kareiva, E.P. Maurer, A.R. Blaustein, and P.J. Bartlein. 2009. Projected climate-induced faunal change in the western hemisphere. *Ecology* 90: 588-597

Dataset S1. **Data file (Dataset_SI_1.xls) with the 194 vertebrate species used for estimating changes in amounts of wildlife habitat due to projected land-use change.** The species are in taxonomic order with their membership in the species groups used in our analysis are shown: amphibians, influential species (top carnivores and ecosystem engineers), game (hunted species), and declining birds (categorized by the American Bird Conservancy (2012) as 'vulnerable' or 'potential concern'). In addition we show each species' IUCN status (EN = endangered, VU = vulnerable, NT = near-threatened, and LC = least concern) and whether their prime habitat associations show a specialization ('forest' = all forest macrogroups, 'range' = all range macrogroups, 'none' = both forest and range macrogroups, may also include human-dominated land uses).

Figure S1. Change in biomass carbon stock (Mg / ha). Each dot represents a county (the dot is placed at its county's centroid). The top row of maps gives change over time for the two reference scenarios. The bottom row of maps gives differences as of 2051 when a scenario outcome is compared to the 1990s Trends outcome. In the bottom row of maps a positive (negative) number for a county means that the scenario value for that county is higher (lower) than the 1990s Trends value for that county. In all maps white space indicates no change for that county over time or between a scenario and the 1990s Trends as of 2051.

Figure S2. Change in soil carbon stock (Mg / ha of first 30 cm of soil profile). Each dot represents a county (the dot is placed at its county's centroid). The top row of maps gives change over time for the two reference scenarios. The bottom row of maps gives differences as of 2051 when a scenario outcome is compared to the 1990s Trends outcome. In the bottom row of maps a positive (negative) number for a county means that the scenario value for that county is higher (lower) than the 1990s Trends value for that county. In all maps white space indicates no change for that county over time or between a scenario and the 1990s Trends as of 2051.

Figure S3. Change in food production (Millions of KCals / ha). Each dot represents a county (the dot is placed at its county's centroid). We assume crop yields for all crops and locations increase 6% every 5 years. The top row of maps gives change over time for the two reference scenarios. The bottom row of maps gives differences as of 2051 when a scenario outcome is compared to the 1990s Trends outcome. In the bottom row of maps a positive (negative) number for a county means that the scenario value for that county is higher (lower) than the 1990s Trends value for that county. In all maps white space indicates no change for that county over time or between a scenario and the 1990s Trends as of 2051.