5-20-2014

Projected land-use change impacts on ecosystem services in the United States

Joshua J. Lawler
University of Washington

David J. Lewis
Oregon State University

Erik Nelson
Bowdoin College

Andrew J. Plantinga
University of California, Santa Barbara

Stephen Polasky
University of Minnesota Twin Cities

See next page for additional authors

Projected land-use change impacts on ecosystem services in the United States

Joshua J. Lawler[a,1], David J. Lewis[b], Erik Nelson[c], Andrew J. Plantinga[d], Stephen Polasky[e,1], John C. Withey[f], David P. Helmers[g], Sebastián Martinuzzi[g], Derric Pennington[h], and Volker C. Radeloff[g]

[a]School of Environmental and Forest Sciences, University of Washington, Seattle, WA 98195; [b]Department of Applied Economics, Oregon State University, Corvallis, OR 97331; [c]Department of Economics, Bowdoin College, Brunswick, ME 04011; [d]Bren School of Environmental Science and Management, University of California, Santa Barbara, CA 93106; [e]Department of Applied Economics and Department of Ecology, Evolution, and Behavior, University of Minnesota, St. Paul, MN 55108; [f]Department of Biological Sciences, Florida International University, Miami, FL 33199; [g]SILVIS Laboratory, Department of Forest and Wildlife Ecology, University of Wisconsin–Madison, Madison, WI 53706; and [h]World Wildlife Fund, Washington, DC 20090

Contributed by Stephen Polasky, March 27, 2014 (sent for review July 12, 2013)

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 United States 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 with 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 and 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.

econometric model | incentives | at-risk birds | game species | amphibians

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 United States 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 United States 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 United States 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 5-y 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 5-y 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 (Table 1: (*i*) forest incentives [incentives for afforestation and

Significance

Land-use change affects the provision of ecosystem services and wildlife habitat. We project land-use change from 2001 to 2051 for the contiguous United States 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.

Author contributions: J.J.L., D.J.L., E.N., A.J.P., S.P., J.C.W., and V.C.R. designed research; E.N., A.J.P., J.C.W., D.P.H., S.M., and D.P. performed research; J.J.L., D.J.L., E.N., A.J.P., S.P., J.C.W., and V.C.R. analyzed data; and J.J.L., D.J.L., E.N., A.J.P., S.P., J.C.W., S.M., and V.C.R. wrote the paper.

The authors declare no conflict of interest.

Data deposition: A land use/land cover database exists at the SILVIS Laboratory at the University of Wisconsin–Madison. Contact D.P.H. or V.C.R. for information about accessing the data.

[1]To whom correspondence may be addressed. E-mail: jlawler@uw.edu or polasky@umn.edu.

This article contains supporting information online at www.pnas.org/lookup/suppl/doi:10.1073/pnas.1405557111/-/DCSupplemental.

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	Not applicable
High crop demand	Land-use changes accounting for 10% increase in crop prices every five years relative to the 1990s Trend scenario	Not applicable
	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 nonmetropolitan counties.	Habitat, timber production, carbon storage, food production

reduced deforestation similar to carbon sequestration incentives (e.g., ref. 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 nonmetropolitan 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 (Figs. 1 and 2 and Fig. S1) with rapid urban growth (Figs. 1*D* and 2*A*) and loss of rangelands and pasture (Figs. 1 *B* and *E* and 2*A*). 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 United States, respectively. Forest land showed modest increases overall but had a complex pattern of gains and losses (Figs. 1*C* and 2*A*).

Comparing the projections for the two baseline scenarios clearly demonstrates the importance of underlying drivers of land-use change (Figs. 1*A* and 2*A*). In the high crop demand scenario cropland is projected to have a large increase (28.2 million ha) compared with 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. 2*B*). 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 5 y) and changes in agriculture area.

Both land-use change scenarios also result in overall increases in carbon storage and timber production (Fig. 2 *C* and *D*). 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. 2*C*).

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 with the 1990s trend (Wilcoxon signed-rank test, $V = 5{,}337$, $n = 194$, $P < 0.001$, 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. 2 *E*–*H*). At-risk birds are the most sensitive to land-use change. Roughly one-third of these species are projected to lose more than 10% of their habitat (Fig. 2*H*).

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 (Figs. 3 and 4, and Figs. 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 (Figs. 3*A* and 4*A*). 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 (Figs. 3*B* and 4*A*). 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 (Figs. 3*C* and 4*A*). 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 relative to baseline, 18%). The forest incentives policy reduces food production by 10% (1.93×10^{14} kcal) compared with 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 with 13% of species under the forest incentives policy, and 16% under

SUSTAINABILITY SCIENCE

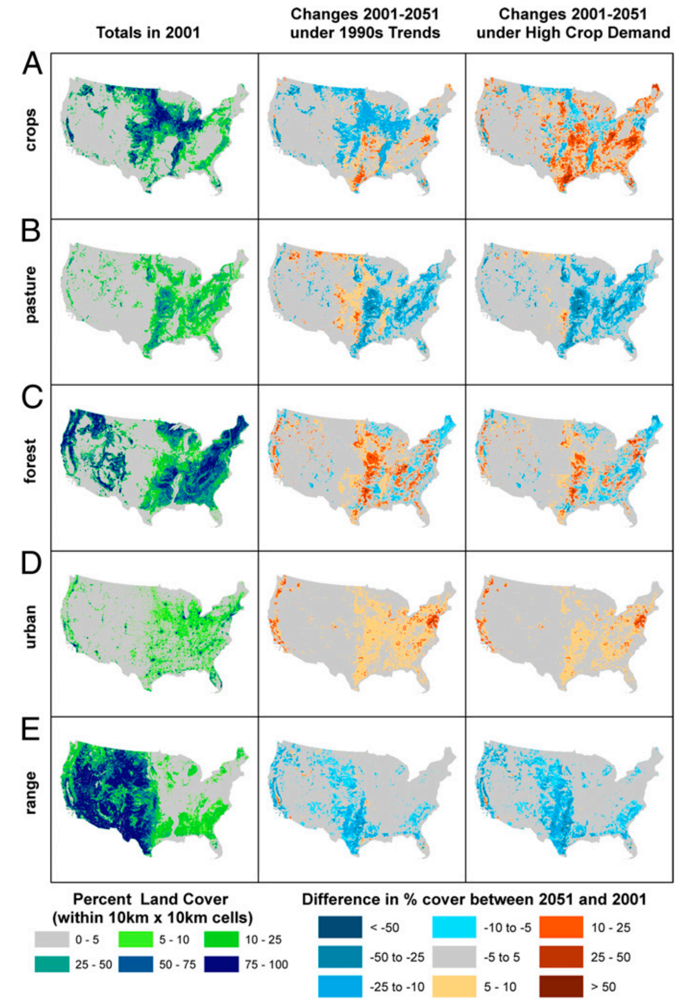

Fig. 1. Spatial patterns in land cover in 2001 and changes between 2001 and 2051 under two baseline scenarios, 1990s trends and high crop demand, for crops (*A*), pasture (*B*), forest (*C*), urban (*D*), and range (*E*).

the urban containment policy. All groups of species do better under the natural habitats policy (Fig. 4 *E–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, owing to expansion of forest land under our baseline scenarios. However, 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 owing to increases in crop yields. We assume a 6% increase in yield every 5 y, 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-y period (7) but consistent with projections showing positive but declining growth in US 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 seem 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.

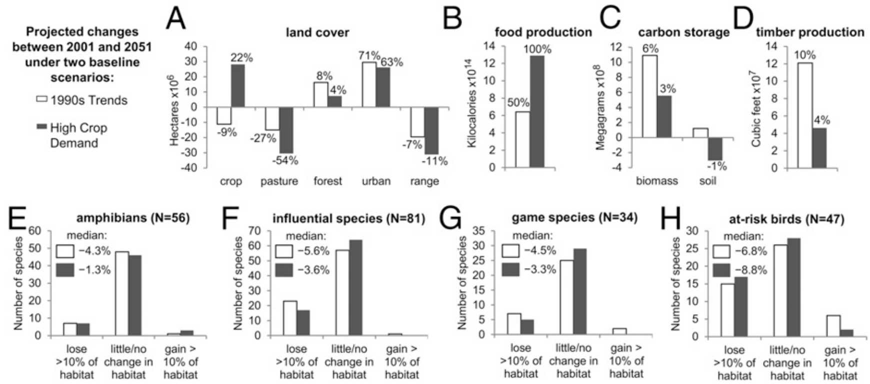

Fig. 2. Projected changes between 2001 and 2051 under the two baseline scenarios for (*A*) land cover, (*B*) food production, (*C*) carbon storage, (*D*) timber production, and area of prime habitat for different groups of wildlife species (*E–H*). The bars in *A–D* display the difference between 2051 and 2001 with labels for changes greater than 1%. Bars in *E–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 percent change across species in each group, by baseline scenario, is shown in *E–H*.

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 multiobjective 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., refs. 13 and 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.

Although 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 ha 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., refs. 16 and 20).

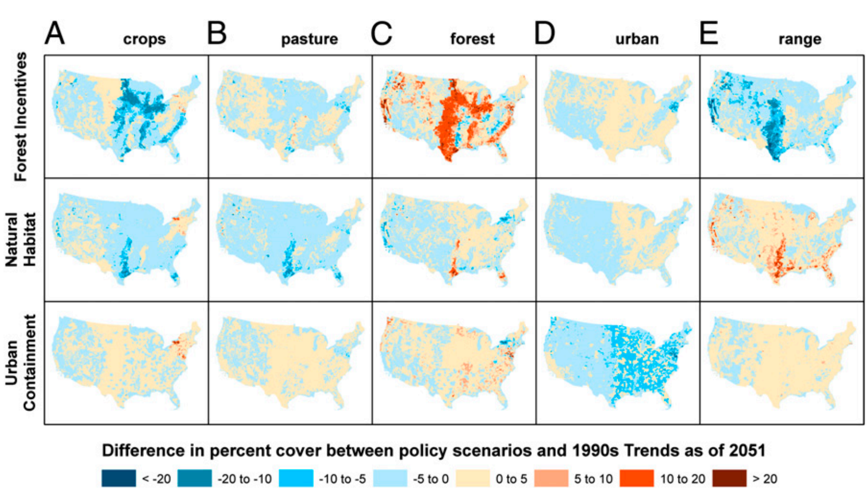

Fig. 3. (*A–E*) Spatial patterns in land cover changes under the three conservation policy scenarios (forest incentives, natural habitats, and urban containment) relative to projections based on the 1990s trends baseline scenario.

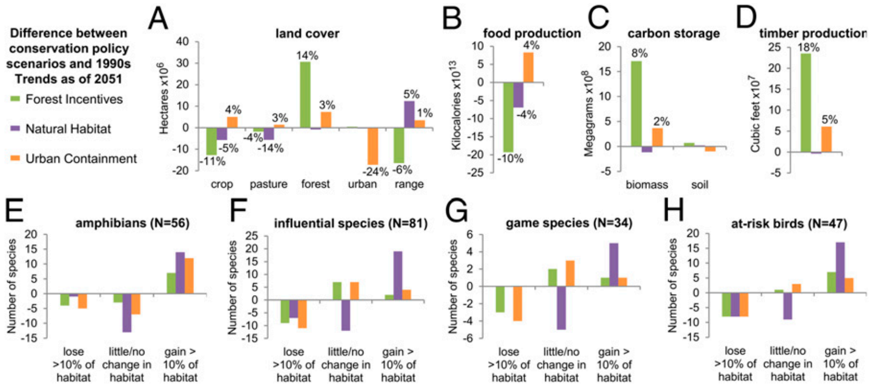

Fig. 4. Projected changes under the three conservation policy scenarios (forest incentives, natural habitats, and urban containment) relative to projections based on the1990s Trends scenario for (*A*) land cover, (*B*) food production, (*C*) carbon storage, (*D*) timber production, and area of prime habitat for different groups of wildlife species (*E*–*H*). The bars in *A*–*D* display the difference between the policy scenarios and 1990s trends projection as of 2051, with labels for changes greater than 1%. Bars in *E*–*H* show the increase or decrease in the number of species in the categories (defined in Fig. 2) under each policy scenario compared with 1990s trends baseline scenario.

Our model results can be compared with other relatively fine-grained model projections of regional land-use change scenarios (e.g., ref. 21), agent-based modeling approaches, and deterministic housing growth models (e.g., ref. 19).

Second, we combine an endogenous price modeling approach that captures the effect of changes in major land uses (agriculture, forestry, or 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., refs. 9 and 22). However, 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., ref. 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. Also, although climate change could affect certain scenarios and policies more than others we have left that analysis for further research (see *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 *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 National Resource Inventory (NRI) (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 2001–2051 for each county–soil type combination. The transition matrices account for movements of land among five NRI categories: crops, pasture, forest, urban, and range (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 5 y. 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 owing 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 resampled 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-y land-use transition matrices with the resampled 2001 map to generate an expected plot-level 2051 land-use map for the contiguous United States. 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 *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 y 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 United States, 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 with 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–S5, and Dataset S1 for more details.

ACKNOWLEDGMENTS. We are grateful for comments from three anonymous reviewers, which greatly improved the manuscript. We gratefully acknowledge support for this research from the National Science Foundation's Coupled Natural–Human Systems Program.

1. Millennium Ecosystem Assessment (2005) *Ecosystems and Human Well-being Biodiversity Synthesis* (World Resources Inst., Washington, DC).
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* (US Dept. of Agriculture, Washington, DC).
3. US 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. *J Environ Econ Manage* 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. *Proc Natl Acad Sci USA* 108(50):20260–20264.
7. US Department of Agriculture, National Agricultural Statistics Service (2012) Quick Stats 2.0. Available at http://nass.usda.gov/Quick_Stats/. Accessed December 20, 2013.
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. *Am J Agric Econ* 95:435–441.
10. Lobell DB, Schlenker W, Costa-Roberts J (2011) Climate trends and global crop production since 1980. *Science* 333(6042):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. *Environ Resour Econ* 48:177–218.
13. National Research Council (2005) *Valuing Ecosystem Service*: *Towards Better Environmental Decision-Making* (National Academies, 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. *J Environ Manage* 92(3):563–574.
19. Bierwagen BG, et al. (2010) National housing and impervious surface scenarios for integrated climate impact assessments. *Proc Natl Acad Sci USA* 107(49):20887–20892.
20. Goldstein JH, et al. (2012) Integrating ecosystem-service tradeoffs into land-use decisions. *Proc Natl Acad Sci USA* 109(19):7565–7570.
21. Sohl T, Loveland TR, Sleeter BM, Sayler KL, Barnes CA (2010) Addressing foundational elements of regional land-use change modeling. *Landscape Ecol* 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* 3:1250015-1–1250015-30.
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(3):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* (US Dept. 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. *Eos Trans* Suppl 90(52) (abstr 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. US Department of Agriculture, Agricultural Research Service (2011) USDA National Nutrient Database for Standard Reference, Release 24. Available at www.ars.usda.gov/ba/bhnrc/ndl. Accessed December 20, 2013.
30. American Bird Conservancy (2012) List of the birds of the United States with conservation rankings. Available at www.abcbirds.org/abcprograms/science/conservationchecklist/index.html. Accessed December 19, 2013.
31. Comer P, et al. (2003) *Ecological Systems of the United States*: *A Working Classification of U.S. Terrestrial Systems* (NatureServe, Arlington, VA).