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Threats and opportunities for freshwater conservation under future land use change scenarios in the United States

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Abstract

Freshwater ecosystems provide vital resources for humans and support high levels of biodiversity, yet are severely threatened throughout the world. The expansion of human land uses, such as urban and crop cover, typically degrades water quality and reduces freshwater biodiversity, thereby jeopardizing both biodiversity and ecosystem services. Identifying and mitigating future threats to freshwater ecosystems requires forecasting where land use changes are most likely. Our goal was to evaluate the potential consequences of future land use on freshwater ecosystems in the coterminous United States by comparing alternative scenarios of land use change (2001–2051) with current patterns of freshwater biodiversity and water quality risk. Using an econometric model, each of our land use scenarios projected greater changes in watersheds of the eastern half of the country, where freshwater ecosystems already experience higher stress from human activities. Future urban expansion emerged as a major threat in regions with high freshwater biodiversity (e.g., the Southeast) or severe water quality problems (e.g., the Midwest). Our scenarios reflecting environmentally oriented policies had some positive effects. Subsidizing afforestation for carbon sequestration reduced crop cover and increased natural vegetation in areas that are currently stressed by low water quality, while discouraging urban sprawl diminished urban expansion in areas of high biodiversity. On the other hand, we found that increases in crop commodity prices could lead to increased agricultural threats in areas of high freshwater biodiversity. Our analyses illustrate the potential for policy changes and market factors to influence future land use trends in certain regions of the country, with important consequences for freshwater ecosystems. Successful conservation of aquatic biodiversity and ecosystem services in the United States into the future will require attending to the potential threats and opportunities arising from policies and market changes affecting land use.

Keywords: conservation, freshwater biodiversity, land use modeling, scenarios, water quality

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Introduction

Globally, freshwater ecosystems deliver vital resources to humans while supporting 10% of all known species and nearly 50% of the world's fishes (Carrizo *et al*., 2013). However, because of the strong human dependence on fresh waters, changes in land use, water course alterations, and the introduction of species have led to widespread water pollution, habitat degradation, and biodiversity loss (Malmqvist & Rundle, 2002; Dudgeon *et al*., 2006). As a result, freshwater ecosystems are one of the most – if not the most – endangered class of ecosystems in the world (Dudgeon *et al*., 2006). Without significant changes to the current unsustainable use of water resources, future degradation of river, lake, and wetlands will jeopardize both biodiversity and critical ecosystem services relied upon by humanity (Sala *et al*., 2000; Rockström & Karlberg, 2010).

Human activities have reached a scale where we affect vital planetary processes (Foley *et al*., 2005), and these alterations have pervasive negative effects on freshwater biodiversity by reducing species richness, distribution patterns, and food web interactions (Vörösmarty *et al*., 2010; Carpenter *et al*., 2011). Human land use changes, such as the expansion of urban and crop cover, are probably the greatest future threat to freshwater biodiversity (Sala *et al*., 2000; Allan, 2004). Indeed, in many parts of the world, increasing human population and development pressures, create a double squeeze on freshwater ecosystems from both cropland

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and urban expansion. The expansion of croplands increases the amount of sediments, nutrients, and pesticides entering fresh waters (Meyer *et al*., 1988; Schaller *et al*., 2004). In addition, hydrological alterations used to support agricultural systems can reduce in stream flows and groundwater stores (Scanlon *et al*., 2007), attenuate flood pulses, and reduce riparian habitat (Poff *et al*., 1997) and native fish movement (Schlosser, 1995). Similarly, even seemingly small proportions of urban land cover (i.e., 10–20%; Allan, 2004) can lead to substantial increases in the amount of chemical and thermal pollution in rivers (Hope, 2012), and decreases in stream-channel habitat structure and biodiversity (Roy *et al*., 2003; King *et al*., 2011).

The potential ecological impact of future land use changes on freshwater ecosystems has received much less attention than for terrestrial ecosystems (Langpap *et al*., 2008). In the United States, recent studies have demonstrated that water quality (Brown & Froemke, 2012) and fish habitats (Esselman *et al*., 2011) are already impaired by land use in most of the country. At the same time, substantial changes in land use are likely in the United States (Radeloff *et al*., 2012; Sleeter *et al*., 2012), and these changes have the potential to affect the ecological condition of freshwater ecosystems. Indeed, projections of housing density alone suggest that the number of watersheds stressed due to urbanization could double by 2030 (Theobald *et al*., 2009).

Informed decision making for biodiversity conservation will require accounting for future land use changes across a variety of socioeconomic scenarios (Peterson *et al*., 2003; Polasky *et al*., 2011). In the context of freshwaters, such a forward-looking perspective must include exploring the potential effects of conservation policies or crop market changes, understanding where and when crop and urban cover are likely to achieve or exceed critical levels (Allan, 2004), and identifying places where human modification in land cover is likely to remain low. These types of information can allow decision makers to identify potential threatened areas, design proactive mitigation strategies, and seek restoration opportunities.

Here, we evaluated the potential consequences of future land use changes for freshwater ecosystems by comparing projected patterns of future land use change with current patterns of freshwater biodiversity and water quality impairment. In particular, we were interested in how alternative policy scenarios that translate into differential land use changes might affect important areas for freshwater conservation and management. Specifically, our objectives were to:

1. Use an econometric model to quantify future land use changes in watersheds across the coterminous United States under alternative policy scenarios for the period 2001–2051;
2. Evaluate future land use changes in areas of freshwater biodiversity significance;
3. Compare future patterns of land use change with current patterns of water quality impairment.

Materials and methods

Objective 1 – scenarios of future land use

We quantified future land use changes in US watersheds using an econometric-based land use model developed by Radeloff *et al.* (2012), which projects nationwide changes in land use from 2001 to 2051 at 100-m pixel resolution. The model predicts changes in urban, crop, pasture, forest, and rangelands using an econometric multinomial logit function from Lubowski *et al.* (2006), reflecting observed landowner decisions in response to expected economic benefits of each type of land use (i.e., net return). The results from this econometric estimation specify probabilistic land use transition matrices for each combination of land use type (from the 2001 National Land Cover Database[1] or NLCD), soil characteristics (from the Soil Survey Geographic Database[2]), and county. These transition probabilities were estimated based on approximately 800 000 plot-level observations of past land use change during the 1990s from the Natural Resources Inventory (NRI[3]), together with county-level information about potential economic returns from each possible land use and the costs of conversion (Radeloff *et al.*, 2012). Only private lands were allowed to change use; public lands such as national parks and other protected areas were assumed to remain in the same land cover.

Econometric models can simulate the effects of alternative assumptions regarding the level of net returns to various land uses, thus making it possible to compare alternative scenarios of future land use change. The level of future net returns can be policy-induced through subsidies or taxes, or market-induced through assumptions about future demand for land-based commodities. For the purpose of this study, we quantified future land use changes under four different scenarios: *Business As Usual*, which projected future conditions with no subsidies or taxes other than the ones present when the model was developed (i.e., 1990s conditions); *Forest Incentives*, which provided a US$ 100/acre subsidy for land entering forest (afforestation) and a US$ 100/acre tax for clearing land of forest (a surrogate for a REDD mechanism); *High Crop Demand*, which assumed a 2% annual increase in all crop prices and maintains all lands in the Conservation Reserve Program (CRP); and *Urban Containment*, which restricted urban expansion to metropolitan counties only (as

[1]Available at: http://www.mrlc.gov/nlcd2001.php
[2]Available at: http://soils.usda.gov/survey/geography/ssurgo/
[3]Available at: http://www.nrcs.usda.gov/technical/nri/

defined by the US Census), thus reflecting potential zoning or smart-growth regulation.

The *Business As Usual* scenario reflected 1990s land use trends dominated by urbanization and declining cropland, which were driven by market-determined prices for commodities, and continuation of policies such as agricultural subsidies, CRP, and mortgage deduction. The other scenarios were modifications of *Business As Usual*. The US $ 100/acre subsidy under the *Forests Incentives* translates into a US $ 50/ton carbon price, thereby reflecting a forceful carbon policy (Lubowski *et al.*, 2006). Finally, the 2% annual increase in crop prices used in the *High Crop Demand* scenario reflected observed trends during boom periods.

We refined the original land use model used in Radeloff *et al.* (2012) by making the economic returns to all uses endogenous with respect to land use change. The exception was crop prices in the High Crop Demand scenario, which were assumed to increase at an exogenously specific rate (2%). By making the economic returns endogenous, an expansion of the forest area under the *Forest Incentives* scenario increased the supply of timber, which reduced timber prices and the economic returns to forest. This reduced the probability that additional land will be converted to forest, and increases the likelihood that it will remain in crops and other uses. Such 'slippage' effects have been found with the CRP and other land use policies (Wu, 2000). The land use projections used here have also been used in Hamilton *et al.* (2013) to quantify future land use around protected areas, and in Martinuzzi *et al.* (2013) to explore potential land use pressures in conservation areas.

We summarized land use within HUC8 hydrological units from the US Geological Survey's Watershed Boundary Dataset (n = 2111; U.S. Geological Suvey, 2010). For each of the four scenarios, we calculated net changes in urban and crop cover (our surrogates for human land uses), and natural vegetation cover (including forests, natural grasslands, and natural shrublands) from 2001 to 2051 within each HUC8 watershed. We calculated these area-based changes as percentages of total watershed area. For example, a 6% expansion in crop cover means that a given watershed was projected to have a net gain of crop cover equivalent to 6% of the total watershed area.

We mapped the percentage changes in urban, crop, and natural vegetation cover across watersheds to assess geographic patterns of land use change between 2001 and 2051. In parallel, we summarized the total number of watersheds expected to be below, within, and above critical land-cover threshold values for impact upon aquatic biodiversity and water quality. These thresholds differed between urban and crop cover, following Allan (2004). Only 10–20% of total urban or impervious surface [or even less (King *et al.*, 2011)] causes a rapid decline in water quality in a watershed, whereas a watershed may still be in 'good' condition with 30–50% crop cover (Allan, 2004). Thus, because the total urban area in a given watershed is typically smaller than the area of natural vegetation or crop cover, a change in urban cover was considered 'substantial' if it was greater than 5% of the watershed area, but we considered changes of more than 10% to be substantial for crops and natural vegetation.

Objective 2 – land use changes and freshwater biodiversity

To forecast land use effects on aquatic biodiversity, we quantified the rarity-weighted species richness for fishes and amphibians within each watershed. For this, we collated spatial data from NatureServe (2010) and the International Union for Conservation of Nature (International Union for Conservation of Nature, 2010) representing distributions for 802 native freshwater fishes and 275 native amphibians, respectively. Fish occurrences were already summarized by watersheds, and we summarized the amphibian species' distributions by watershed. We calculated a combined rarity-weighted richness (RWR; Williams *et al.*, 1996; Abell *et al.*, 2011b) for fishes and amphibians combined, defined as:

$$\mathrm{RWR}_h = \sum_{s=1}^{S_h} 1/N_s$$

where S_h is the number of species in a watershed i, and N_s is the total number of watersheds occupied by species s. The benefit of analyzing RWR is that it integrates two common measures of biodiversity: species richness (number of species) in a defined area, and range limitation (rarity) of each of the species present (Redford *et al.*, 2003; Abell *et al.*, 2011b). Thus, high values of RWR can arise from high local (alpha) diversity, or from the presence of species with very limited distributions, or both together. Following Abell *et al.* (2011b) we used the upper quartile of RWR values to identify watersheds of freshwater biodiversity significance. Then, for each of the four different socioeconomic scenarios, we compared future changes in urban, crop, and natural vegetation, inside and outside of the watersheds of biodiversity significance.

Objective 3 – land use changes and water quality

To compare future patterns of land use change with current patterns of water quality impairment, we used Brown & Froemke's (2012) ratings of water quality degradation in each of 18 HUC2-level hydrological regions of the United States. These ratings summarize the current level of anthropogenic threats to water quality on a 0–1 scale (0 is the lowest threat) based on GIS maps of stressors like housing density, roads, agriculture, livestock, atmospheric deposition, and mines. Hydrological regions (HUC2) are larger than watersheds (HUC8); the average area for the HUC2 units is 43 Million ha vs. 0.4 Million ha for the HUC8. We scaled up our land use change projections to hydrological regions by calculating the total land use change values across all the HUC8 watersheds within each HUC2 region. Our interpretation of the significance of projected land use changes for water quality was qualitative; we assumed that expansion of urban or cropland area will exacerbate current threats to water quality, while decreases in these intensive human land uses would diminish water quality degradation.

Results

Land use projections

Under the *Business As Usual* scenario, 30% of all the HUC8-level watersheds were projected to see urban expansion of 5–10%, particularly in Eastern United Stated and in California (Fig. 1). A few watersheds along the coast (4% of 2111) were projected to see urban expansion greater than 10%. Consequently, the number of watersheds with greater than 20% urban cover was projected to double, from 104 (5%) to 212 (10%) watersheds, and those with greater than 10% urban cover were projected to triple, from 14% to 43% (Table 1). At the same time, crop cover under the *Business As Usual* scenario was projected to decline in Midwestern watersheds, while natural vegetation was projected to expand in this region but decline in other parts of the East and Central United Stated (Fig. 1). The number of watersheds with >30% crop cover was projected to decrease by 16% (Table 1). Finally, under *Business As Usual*, about 20% of all watersheds were expected to see substantial changes in crop or natural vegetation cover (i.e., changes greater than 10%).

The other three scenarios also projected that most of the land use changes would occur in the eastern half of the country. Compared to *Business As Usual*, the *Forest Incentives* scenario projected a greater expansion of natural vegetation in Midwestern watersheds, with most watersheds in this region expecting 10–20% forest expansion (vs. 5–10% under *Business As Usual*; Fig. 1). In the *Urban Containment* scenario, the most notable result was the much lower rates of urban growth projected in Eastern watersheds, but with little effect in watersheds of California. At the national level, the proportion of watersheds projecting substantial urban expansion (5% or more) declined from 34% under *Business As Usual* to 13% under *Urban Containment* (Table 2). Finally, the *High Crop Demand* scenario projected few changes in land use for the Midwest (in contrast to the other scenarios) but 5–20% crop expansion in the south-central and southeastern United States watersheds (Fig. 1). As a result, the number of watersheds with >30% crop cover was projected to increase by 23% between 2001 and 2051 under the *High Crop Demand* scenario (compared to a 16% reduction under *Business As Usual*; Table 1).

Watersheds of freshwater biodiversity significance

Rarity-weighted richness (RWR) at the watershed scale ranged from 0.0 to 4.2. Focusing on the upper quartile of RWR values, we identified 820 (39%) watersheds of biodiversity significance, which were concentrated in the Southeast (i.e., the South Atlantic Gulf and Tennessee hydrological regions), Ohio, California, and some parts of the Southwest (Fig. 2). Less than 10% of watersheds in the Missouri, Arkansas White Red, and New England hydrological regions were of biodiversity significance, and no watersheds in the upper Colorado region had high RWR.

Relative to current land use (2001, see Fig. 3 top), we found that watersheds of biodiversity significance contain substantial amounts of both urban lands and natural vegetation. One-fifth of these watersheds (164) had >10% urban cover, representing about half of all of US watersheds that exceeded the 10% threshold (302, see Table 1). Crop cover was less common; only 9% of watersheds of biodiversity significance had >30% crop cover. Finally, 70% of watersheds of biodiversity significance also had high amounts of natural vegetation (>50% cover).

For the future, our model projected notable land use changes in watersheds of biodiversity significance. In the *Business As Usual* scenario, for example, nearly half of these watersheds were predicted to increase >5% in urban cover, especially in eastern watersheds, and along the West coast (Fig. 3; Table 2). The number of watersheds of biodiversity significance containing >10% urban cover was projected to triple, from 20% to 59%, and those with >20% urban cover were projected to double, from 6% to 14% (Table 1). On the other hand, under *Business As Usual*, projected changes greater than 10% for crop and natural vegetation cover were restricted to just a few watersheds (Fig. 3).

Among the other scenarios of land use change by 2051, the *Urban Containment* and *High Crop Demand* scenarios had the strongest effects. In eastern watersheds with high biodiversity, *Urban Containment* greatly reduced the rates of urban expansion while *High Crop Demand* increased the presence of crop cover (see Fig. 3). Overall, 22% of watersheds of biodiversity significance were projected to see >10% crop cover expansion under the *High Crop Demand* scenario, vs. 2% under *Business As Usual* (Table 2). The *Forest Incentives* scenario yielded similar results to *Business As Usual*, as most of the changes in forest cover associated with this scenario were outside of the watersheds of biodiversity significance.

Threats to water quality and future land use changes

According to Brown & Froemke (2012), the hydrological regions currently experiencing highest threat from water quality are all in the Northeast and Midwest, including the Mid-Atlantic, Great Lakes, Ohio, and Upper Mississippi (Table 3). Under *Business As Usual* conditions, these regions were projected to expand the

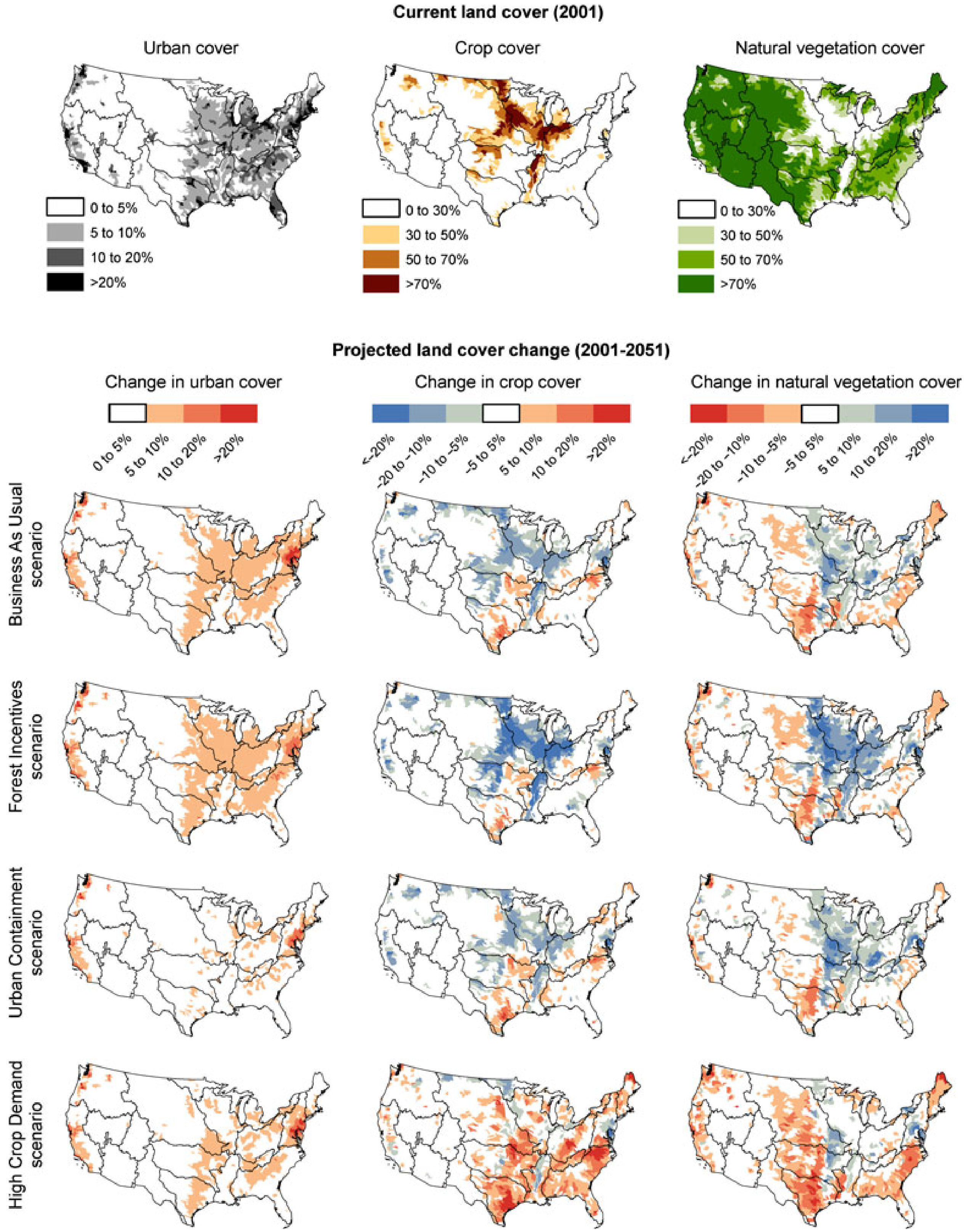

Fig. 1 Projected land use changes at the watershed scale (HUC8; n = 2111) for the period 2001–2051 under different policy and economic scenarios. The boundaries of the hydrological regions (n = 18) are shown in black. In the bottom panel, red and blue colors represent increases or decreases in potential threat to freshwater ecosystems, respectively.

Table 1 Distribution of urban, crop, and natural vegetation cover across watersheds for 2001 and 2051, and under different scenarios. Results are presented for all watersheds (n = 2111) and for watersheds of biodiversity significance (n = 820, in italics)

		Number of watersheds in the coterminous United States (n = 2111)/*number of watersheds of biodiversity significance (n = 820)*									
Total land cover		2001		Business as usual		Forest incentives		Urban containment		High crop demand	
Urban	0–5%	1187	*331*	660	*142*	648	*140*	1022	*244*	683	*148*
	5–10%	622	*325*	545	*192*	524	*184*	581	*296*	617	*245*
	10–20%	198	*111*	694	*369*	728	*382*	324	*181*	614	*318*
	>20%	104	*53*	212	*117*	211	*114*	184	*99*	197	*109*
Crop	0–30%	1653	*746*	1726	*770*	1774	*788*	1710	*769*	1548	*695*
	30–50%	210	*51*	233	*41*	270	*31*	230	*42*	299	*100*
	50–70%	150	*19*	145	*9*	67	*1*	154	*9*	195	*23*
	70–100%	98	*4*	7	*0*	0	*0*	17	*0*	69	*2*
Natural vegetation	0–30%	408	*98*	350	*90*	251	*70*	334	*83*	430	*117*
	30–50%	323	*152*	389	*166*	442	*153*	378	*160*	397	*182*
	50–70%	421	*225*	498	*252*	545	*290*	449	*213*	472	*245*
	70–100%	959	*345*	874	*312*	873	*307*	950	*364*	812	*276*

Table 2 Distribution of land use change classes across watersheds between 2001 and 2051, under different scenarios. Results are presented for all watersheds (n = 2111) and for watersheds of biodiversity significance (n = 820, in italics)

		Number of watersheds in the coterminous United States (n = 2111)/*number of watersheds of biodiversity significance (n = 820)*							
Change in land cover		Business as usual		Forest incentives		Urban containment		High crop demand	
Urban	0–5%	1403	*449*	1311	*424*	1846	*670*	1641	*554*
	5–10%	632	*323*	725	*343*	209	*111*	409	*225*
	10–20%	55	*34*	63	*43*	44	*30*	44	*30*
	>20%	21	*14*	12	*10*	12	*9*	17	*11*
Crop	<−20%	23	*4*	173	*26*	11	*3*	4	*2*
	−20 to −10%	242	*40*	200	*47*	196	*36*	43	*5*
	−10 to −5%	211	*49*	169	*55*	210	*44*	103	*25*
	−5 to 5%	1452	*606*	1448	*612*	1474	*604*	1223	*386*
	5–10%	155	*103*	107	*67*	188	*116*	438	*225*
	10–20%	26	*16*	14	*13*	29	*15*	250	*148*
	>20%	2	*2*	0	*0*	3	*2*	50	*29*
Natural vegetation	<−20%	7	*5*	7	*5*	6	*5*	35	*17*
	−20 to −10%	87	*42*	72	*36*	53	*26*	254	*127*
	−10 to −5%	360	*180*	333	*146*	201	*108*	456	*203*
	−5 to 5%	1300	*466*	1192	*472*	1407	*516*	1221	*426*
	5–10%	256	*86*	148	*71*	290	*97*	93	*30*
	10–20%	87	*37*	263	*80*	120	*56*	44	*15*
	>20%	14	*4*	96	*10*	34	*12*	8	*2*

most in urban cover (5–10% growth), coupled with some decline in crop cover (2–8%) and some increase in natural vegetation (up to 6%). On the other hand, the hydrological regions with the lowest threats to water quality, such as the Upper Colorado, Great Basin, and Rio Grande, were projected to see minimal changes in future land cover (Table 3). Accordingly, *Business As Usual* yielded a strong positive relationship between current level of threats to water quality and future urban expansion at the scale of HUC2 hydrological regions (see Fig. 4a).

Under the *Forest Incentives* scenario, some of the regions with the highest threats to water quality were projected to see further expansion of natural vegetation (up to 14% in the Upper Mississippi region; Fig. 4; Table 3). In contrast, the *Urban Containment* reduced

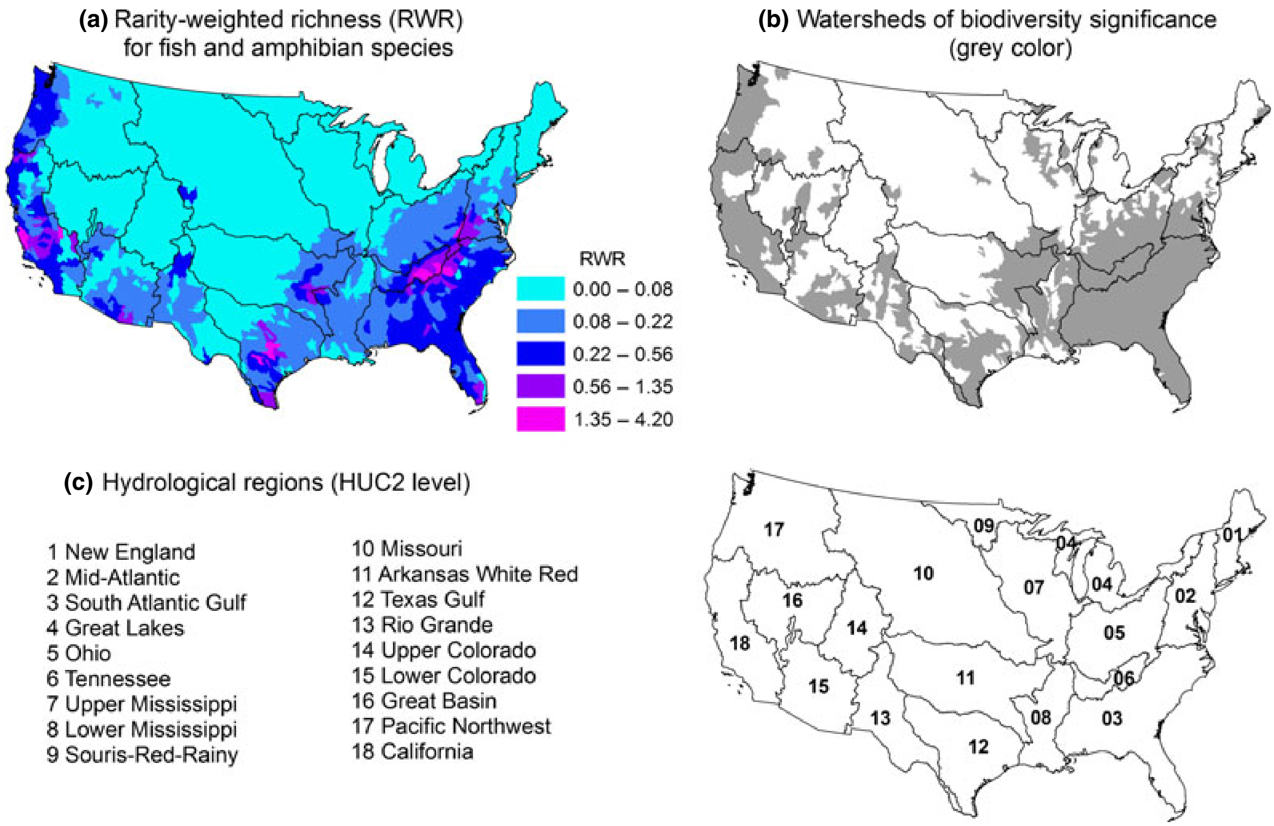

Fig. 2 Values of rarity-weighted richness (RWR) for fish and amphibian species combined across watersheds (n = 2111; Fig. 2a). The watersheds of biodiversity significance are defined as the upper quartile of the RWR values (Fig. 2b). The bottom section displays the boundaries and names of the hydrological regions (HUC2 level, n = 18; Fig. 2c).

rates of urban growth in threatened areas for water quality, and increased natural vegetation cover compared to *Business As Usual*. Finally, the *High Crop Demand* scenario resulted in small increases in crop cover for the most threatened areas compared to present conditions (2001), with the greatest increase in regions that currently experience moderate threats to water quality such as the Texas Gulf, South Atlantic Gulf, and Tennessee (Table 3; Fig. 4).

Discussion

We found that future land use changes are likely to continue to threaten freshwater ecosystems in the United States, but alternative policy- and market-driven scenarios yielded, for some regions, substantially different outcomes with respect to both water quality and biodiversity. Such explorations of the location and consequences of future land use changes are an essential step toward identifying and mitigating anthropogenic threats to freshwater ecosystems (Vörösmarty *et al.*, 2010; Carpenter *et al.*, 2011). Interestingly, substantial future land use changes were projected for watersheds in the eastern half the country irrespective of the policy scenario. This projected growth in human land use was notable, for example, in Southeastern watersheds, which generally have high biodiversity of both fish and amphibian species. Across the nation, future urban expansion was a common threat, but we found that conservation policies could potentially alleviate urban growth in Eastern watersheds. Given that water quality and freshwater biodiversity in the East are already highly stressed (Vörösmarty *et al.*, 2010; Wickham *et al.*, 2011; Brown & Froemke, 2012), our scenario analysis provided important insight into the potential for limiting further degradation.

Watersheds of biodiversity significance (e.g., the southeastern United States, and California) and those that already exhibit high levels of water quality stress (e.g., the Midwest), were projected to receive some of the highest pressures from urbanization. Indeed, all scenarios in this study showed a net increase in urban cover, suggesting that urban land use will continue to be a major threat to freshwater ecosystems, and reinforcing findings based on housing projections alone (Theobald *et al.*, 2009). With increasing urban cover,

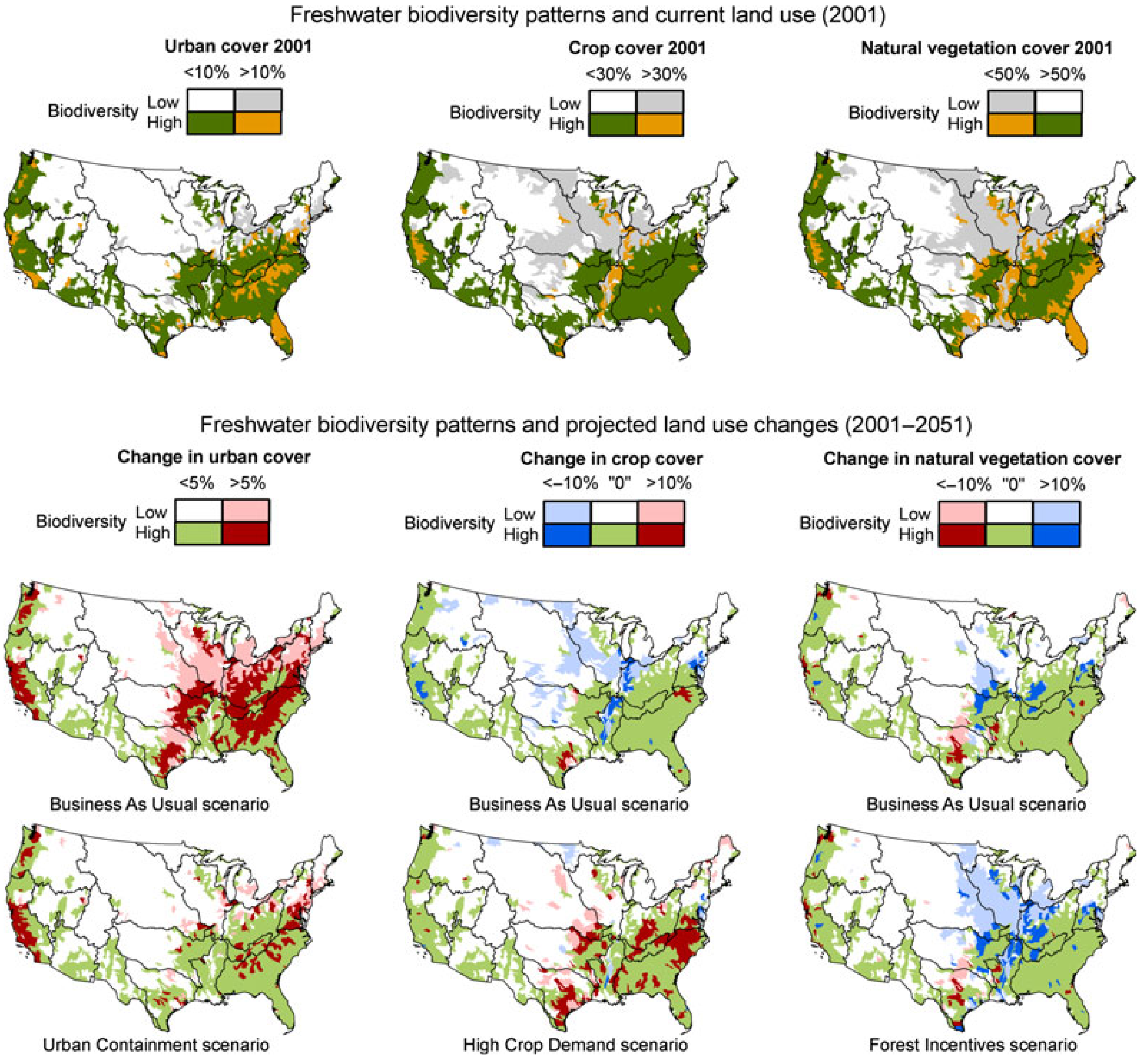

Fig. 3 Comparison of future land use changes for watersheds of biodiversity significance under different policy and economic scenarios for 2001–2051. The classes used to describe the amount of land cover and percent land cover changes are simplified to denote the most relevant results and scenarios (additional information is included in Table 2). In the bottom panel, red and blue colors represent increases or decreases in potential threat to freshwater ecosystems, respectively.

these regions are likely to see increases in pollutants and temperature of runoff water as well as changes in habitat structure and hydrology (Allan, 2004; Abell *et al*., 2011a), as well as further reductions in biotic integrity, increased homogenization of fish assemblages (Morgan & Cushman, 2005; Scott, 2006) and reduced amphibian species richness and abundance (Hamer & McDonnell, 2008).

The broad range of potential urbanization patterns indicated by our scenarios underscored the importance of accounting for water quality and freshwater biodiversity in policy setting. To the degree that the location of future expansion of land use change can be predicted, conservation planning can be used to identify areas where protection or restoration efforts can minimize the impact of emerging threats from land conversion. For the taxonomic groups that we evaluated, conservation actions could include supporting larger interstitial green spaces (Sushinsky *et al*., 2013), which in turn enhance or reconnect fragments of natural habitat for amphibians and fishes. Similarly, investments in smart infrastructure, such as implementing fish-friendly

Table 3 Current levels of water quality threat and future land use changes expressed at the scale of hydrological regions (n = 18), and under different scenario. Values reflecting current levels of water quality threat are from Brown & Froemke (2012); water quality threat level is on a 0–1 scale where 0 is lowest threat and 1 is highest threat from degraded water quality. Land use projections are for the period 2001–2051; projected land cover changes are expressed in percentage

		Projected land use changes under different scenarios (2001–2051)											
		Business as usual			Forest incentives			Urban containment			High crop demand		
Hydrological region	Waterquality threat level (2001)	Ur.	Cr.	Nat.	Ur.	Cr.	Nat.	Ur.	Cr.	Nat.	Ur.	Cr.	Nat.
1 New England	0.30	4.5	2.5	−6.2	4.4	2.2	−6.0	2.0	2.9	−4.1	3.5	7.5	−9.2
2 Mid-Atlantic	0.67	9.9	−2.8	1.5	8.1	−2.2	3.9	5.8	−0.7	3.9	8.5	1.2	2.0
3 South Atlantic Gulf	0.45	4.9	2.1	−2.6	5.3	0.3	−1.2	2.4	2.4	−0.6	4.2	8.5	−6.3
4 Great lakes	0.53	4.8	−2.4	1.1	4.9	−5.4	4.6	2.4	−0.8	1.8	4.0	3.9	−1.4
5 Ohio	0.58	6.0	−3.0	3.8	6.0	−6.4	7.5	2.6	−2.0	6.0	5.1	5.7	−0.7
6 Tennessee	0.49	5.5	3.7	2.9	5.5	2.5	4.0	2.4	4.1	5.4	5.0	9.0	−0.2
7 Upper Mississippi	0.66	5.3	−8.3	5.9	5.5	−15.5	13.9	1.9	−6.9	7.5	4.4	1.8	1.3
8 Lower Mississippi	0.37	4.0	−4.8	1.4	4.2	−8.8	5.8	1.1	−3.8	2.9	3.5	1.9	−1.6
9 Souris-Red-Rainy	0.32	3.7	−9.1	3.6	4.0	−13.6	8.5	0.7	−7.5	4.6	3.5	−5.0	2.3
10 Missouri	0.32	3.2	−2.8	0.1	3.3	−4.0	1.5	0.6	−1.8	1.3	2.9	3.0	−3.1
11 Arkansas White Red	0.43	4.1	−0.5	−1.2	4.3	−2.7	0.8	0.9	0.2	0.9	3.8	5.4	−4.9
12 Texas Gulf	0.38	4.5	3.1	−5.1	4.9	1.4	−3.4	1.9	3.5	−3.1	4.2	10.6	−10.1
13 Rio Grande	0.08	1.2	1.2	−2.9	1.2	1.2	−2.9	0.2	1.2	−2.0	1.2	2.0	−3.2
14 Upper Colorado	0.06	1.1	0.7	−1.1	1.0	0.7	−1.3	0.1	0.7	−0.2	1.0	1.2	−1.3
15 Lower Colorado	0.10	1.0	0.1	−1.5	0.9	0.1	−1.6	0.5	0.1	−1.1	0.9	0.5	−1.5
16 Great Basin	0.05	0.7	0.2	−0.2	0.6	0.2	−0.3	0.3	0.2	0.1	0.6	0.7	−0.6
17 Pacific Northwest	0.15	2.5	−1.8	−1.4	2.5	−2.0	−1.3	1.4	−1.7	−0.4	2.3	1.4	−3.2
18 California	0.28	4.1	−1.8	−2.5	4.3	−1.9	−2.6	3.6	−1.8	−1.9	3.9	1.5	−4.4

Ur., urban; Cr., crops; Nat., natural vegetation.

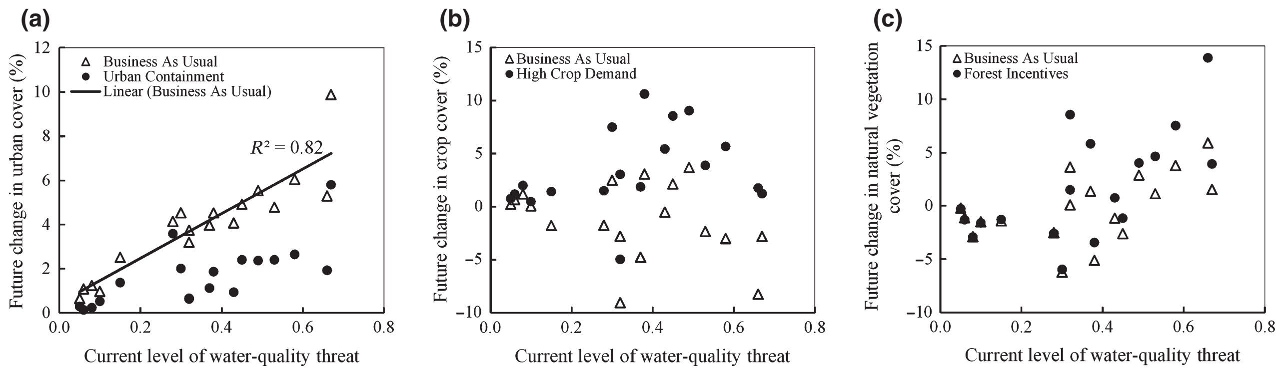

Fig. 4 Relationships between current patterns of water quality threat and future land use changes for urban (a), crop (b) and natural vegetation cover (c), at the scale of hydrological regions (n = 18). Values of current levels of water quality threat are from Brown & Froemke (2012). Only some scenarios of future land use change are shown (see Table 3 for additional results).

road crossing structures, could minimize impacts on freshwater dependent species, (Januchowski-Hartley *et al.*, 2013).

In contrast to urbanization, it was not clear whether changes in crop cover are likely to represent a major threat to freshwater ecosystems. Our baseline condition for projecting future land use change reflected 1990s trends such as urban expansion and declining croplands, and this might explain the potential decrease in crop cover projected in the Midwest (an area of high water quality stress) under *Business As Usual*, *Forest Incentives*, and *Urban Containment* scenarios. However, crop commodity prices have since gone up due to biofuel policies and other factors. Results from our *High*

Crop Demand scenario, which simulated increases in commodity prices, showed that reversing the 1990s trend could result in increasing threat to many areas with high biodiversity and/or with medium levels of water quality stress. Land use projections for 2100 also show different responses of crop cover, indicating that whether crop cover will actually increase or decrease across the United States depends on future socioeconomic conditions (Sleeter *et al.*, 2012). Thus, the main lesson from our scenarios was that crop commodity prices matter a great deal for freshwater conservation because they influence crop land use pressures in critical watersheds.

By extension, periods of relatively low crop commodity prices may create windows of opportunity for establishing easements or land-buy-back programs that benefit freshwater ecosystems. For instance, water quality in the Midwest is highly degraded, but our scenarios consistently indicated a high likelihood of reduction in crop cover that could alleviate nutrient loading, sedimentation, and other stressors. After prioritizing watersheds that provide drinking water sources or harbor high aquatic biodiversity, land purchases or easements could be used to enhance protection or restoration of water quality (Knight *et al*., 2011), and in stream barriers, channelization, and water diversions could be remediated to restore natural habitat structure and hydrology to benefit both fishes (Cross *et al*., 2011) and amphibians (Herrmann *et al*., 2005). In contrast, if future commodity prices continue to rise—as in our *High Crop Demand* scenario—then water quality and conservation outcomes may be strongly affected if crop cover expands into regions with high biodiversity (e.g., Southeast) that have only modest agricultural activity at present.

At the national scale, our analyses showed that federal policies, such as aiming to increase forest carbon sequestration through fiscal incentives (taxes, subsidies) or to limit urban sprawl through zoning regulations, could have positive consequences for freshwater ecosystems. Our *Forest Incentives* scenario suggested an incidental reduction in crop cover in some regions with high water quality degradation, which would create opportunities to ameliorate water resources. In parallel, the *Urban Containment* scenario resulted in decreased urban expansion in areas supporting relatively high freshwater species diversity. These potential outcomes are important because sustainability policies (i.e., REDD, payment for ecosystem services, smart growth) are increasingly under consideration for conserving and managing environmental resources, including within the US (LaRocco & Deal, 2011). Given that land use trajectories could be shaped by national policies in some areas with both rich biodiversity and high water quality stress, there may be many opportunities to align payment for ecosystem services and land use zoning regulations with programs aimed to manage and conserve freshwater ecosystems.

Although the implementation of sustainability policies could help protect freshwater biodiversity from future land use change in some critical regions, we also found that none of the conservation policies tested here sufficed to completely mitigate human impacts on priority regions. For example, our *Urban Containment* scenario, which strongly curtailed urban expansion in many watersheds, had minimal impact in California, which is a hot spot of freshwater biodiversity. Likewise, the *Forest Incentives* scenario still yielded high urban growth in some areas of high stress on water quality (e.g., Midwest). These limitations underscore the necessity of combining of national, regional, and local initiatives with on-the-ground management to effectively conserve freshwater ecosystems across the nation.

Our results provide an initial step toward understanding future threats to freshwater ecosystems in the United States and the potential impact of alternative policy and economic scenarios. However, there are several limitations of our analyses that need to be taken into account when interpreting our results. First, this study focused solely on threats related to land use, yet there are many other important threats to water quality and freshwater biodiversity such as invasive species (Hermoso *et al*., 2011; Januchowski-Hartley *et al*., 2011), physical alterations (Pépino *et al*., 2012; Pracheil *et al*., 2013), and climate change (Caldwell *et al*., 2012), which should also be considered. Second, we assessed threats based on changes in land use area without considering the spatial arrangement of land use within watersheds, the natural connectivity between watersheds, the local environmental conditions (climate and soil), or the legacy of past land uses. All of these factors can influence the ecological effects of future land use change (Harding *et al*., 1998; Gergel *et al*., 2002; Utz *et al*., 2009). Third, our biodiversity assessment relied in part on global-scale data on amphibian distributions that may have some local inaccuracies. As new data and models become available regarding both patterns of land use change (Sleeter *et al*., 2012) and the freshwater resources that are impacted, our ability to evaluate alternative scenarios will surely improve. For the moment, this study should be seen as a first exploration of future land use pressures on freshwater ecosystems that elucidates the broader consequences of economic policies and regulations.

Although substantial uncertainties remain about the effects of changing land use and climate on the world's freshwater ecosystems, conservation actions are needed as we approach the upper limit for human use and degradation of water beyond which the loss of essential ecosystem services (Carpenter *et al.*, 2011) and irreplaceable species (Dudgeon *et al.*, 2006) is likely. Detailed spatial analyses of both land use (e.g., Radeloff *et al.*, 2012; Seto *et al.*, 2012; Sun *et al.*, 2012) and freshwater biodiversity (e.g., Abell *et al.*, 2008) are now available at regional to global scales, offering many opportunities for policy-relevant analysis. Our results illustrate the powerful insights into the potential future of natural resources that can be gained from coupling these new land use simulations within a scenario-based approach.

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