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Demand for land to grow corn for ethanol increased in the United States by 4.9 million hectares between 2005 and 2008, with wide-ranging effects on wildlife, including habitat loss. Depending on how biofuels are made, additional production could have similar impacts. We present a framework for assessing the impacts of biofuels on wildlife, and we use this framework to evaluate the impacts of existing and emerging biofuels feedstocks on grassland wildlife. Meeting the growing demand for biofuels while avoiding negative impacts on wildlife will require either biomass sources that do not require additional land (e.g., wastes, residues, cover crops, algae) or crop production practices that are compatible with wildlife. Diverse native prairie offers a potential approach to bioenergy production (including fuel, electricity, and heat) that is compatible with wildlife. Additional research is required to assess the compatibility of wildlife with different composition, inputs, and harvest management approaches, and to address concerns over prairie yields versus the yields of other biofuel crops.

Keywords: corn, biofuel, grassland, wildlife, cellulosic ethanol

Agriculture has a major effect on the status and integrity of natural ecosystems. Improvements in agricultural practices over the last century have increased productivity and thus the footprint for land and resource use is smaller than it otherwise would have been. However, modern agriculture still adversely affects habitat conservation, water and air quality, carbon sequestration in the soil, and soil fertility (e.g., Foley et al. 2005).

To mitigate the environmental impacts caused by agriculture, the US federal government has developed and implemented various land conservation programs, the most prominent of which is the Conservation Reserve Program (CRP; see, e.g., *www.ncga.com/files/pdf/ConservingLandFutureGenerations.pdf*). The original purpose of the CRP, a voluntary program that pays rent annually to landowners who enroll their agricultural land and convert it to perennial grasslands, was to support commodity prices, reduce soil erosion, and improve water quality on highly erodible croplands (FAPRI 2007). The CRP has also benefited wildlife (e.g., Reynolds 2005, Herkert 2007, Niemuth et al. 2007, Riffell et al. 2008), and the program has evolved over time to more explicitly target benefits beyond soil erosion, including the enhancement of wildlife habitat.

Biofuel production offers the potential to bolster energy security, support rural economies, and reduce greenhouse gas (GHG) emissions. However, biofuel production also has potentially large land-use impacts. Greater demand for biofuels has caused—and may continue to cause—retired croplands to be put back into crop production (Secchi and Babcock 2007, Searchinger et al. 2008). Current US law mandates production of 136 billion liters of biofuel by 2022, which is 740% more than was produced in 2006. High gas prices also contribute to the demand for biofuel production, but given current subsidies and mandates, expansion of biofuel production is assured even if gas prices drop. That expansion may threaten some of the gains the CRP and other land conservation programs have made over the last two decades in the conservation of wildlife, ecosystem services, and biodiversity.

This article provides a framework for assessing the potential environmental impacts of existing and prospective methods of bioenergy production, with a focus on impacts on wildlife. We focus on the effects of biofuel feedstock production on wildlife, although we recognize that wildlife conservation is only one of the benefits that society derives from its lands. We believe that ecosystem services, including wildlife

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production, require special consideration because these services are typically external from market considerations and incentives, making them vulnerable to loss from unintended consequences of policy or shifts in market forces.

Although biofuel in the form of ethanol is the current focus of bioenergy production in the United States, we use the more inclusive term "bioenergy" to include all useful forms of energy that can be extracted from biological crops, residues, or wastes (i.e., liquid fuel, electricity, heating, cooling). Bioenergy includes biodiesel made from fats and oils (e.g., soy oil and canola oil), ethanol made from sugars and starches (e.g., corn grain and sugarcane), cellulosic ethanol (ethanol made from plant biomass either through fermentation or thermochemical processes), and bioelectricity and bioheat (e.g., from biomass burners or gasifiers). We consider the effects of biomass production on both terrestrial and aquatic systems. We define wildlife broadly to include all nondomesticated animals, although we focus primarily on birds because (a) they are the primary species of management concern in grasslands at risk of conversion to bioenergy crops, and (b) there are limitations in the primary literature on potential impacts on other species.

A continuum of effects on wildlife

Bioenergy can be produced using a variety of feedstocks and methods. If nonurban land use is classified along a continuum of intensity of use ranging from intensive agriculture to nature preserves, bioenergy can be produced across almost the entire continuum. At one end of the spectrum, bioenergy can be produced with intensively managed monocultures of annual food crops. This method of production can have large environmental consequences, including habitat loss and the off-field impacts of fertilizer and pesticide runoff (e.g., Foley et al. 2005). Toward the other end of the spectrum, bioenergy can be produced by sustainably harvesting biomass from systems with high plant diversity and low agriculture input (Tilman et al. 2006).

The quality of habitat and the production of ecosystem services on a landscape are affected by several aspects of agricultural production (figure 1). The value of an area as wildlife habitat is influenced by the vegetation type, including plant diversity and whether these plants are invasive; the timing and frequency of harvest; stubble height; refugia; and landscape context. Whether the bioenergy crop represents a net gain or loss of habitat depends on the type of land that it is replacing. Agriculture production in one area can affect habitat in another through fertilizer runoff, pesticide drift, and sedimentation of aquatic habitat. The value of the ecosystem services produced on and around a bioenergy crop field is influenced by the field's productivity, the interannual variability of productivity, the nutrient uptake of crops, rates of carbon sequestration, and hunting leases, among other factors. Many of the environmental impacts of bioenergy production on agricultural fields can be minimized by low-input systems with diverse native species. However, the major drawback of less-intensive systems is that more land is generally required to generate a given amount of energy than would be required by more-intensive systems that use fertilizer, pesticide, and monocultures of high-yield cultivars to maximize productivity. Here we consider a range of methods for producing bioenergy, starting with corn and moving on to less-intensive methods, and evaluate their observed and potential impacts on wildlife.

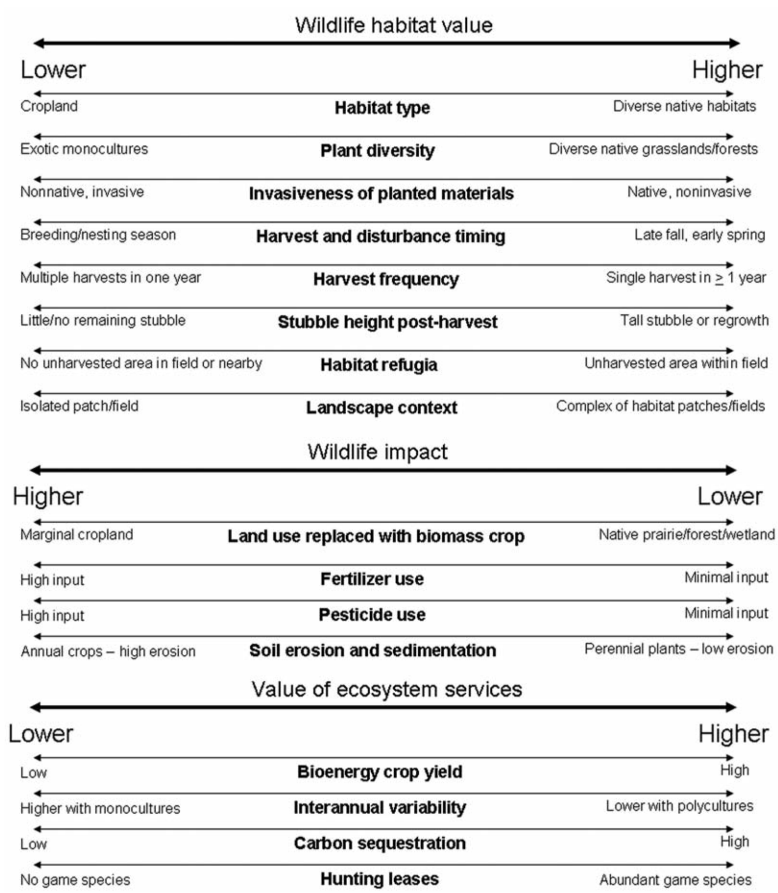

Figure 1. Factors influencing wildlife habitat value, wildlife impacts, and ecosystem services of bioenergy crops. For each factor, the qualities associated with greater wildlife or ecosystem service benefit (or less impact) are listed on the right side of the figure, and the qualities that are associated with less wildlife or ecosystem service benefit (or greater impact) are listed on the left side of the figure.

Current and projected ethanol production and land requirements

In the United States, growing demand for corn ethanol, largely fueled by production subsidies and gasoline blending mandates, has led to an increase in the amount of land used to produce corn (figure 2b, 2c). Most of the recent expansion in corn area has come at the expense of land previously used for other crops, especially soybeans (figure 2c). Some land that

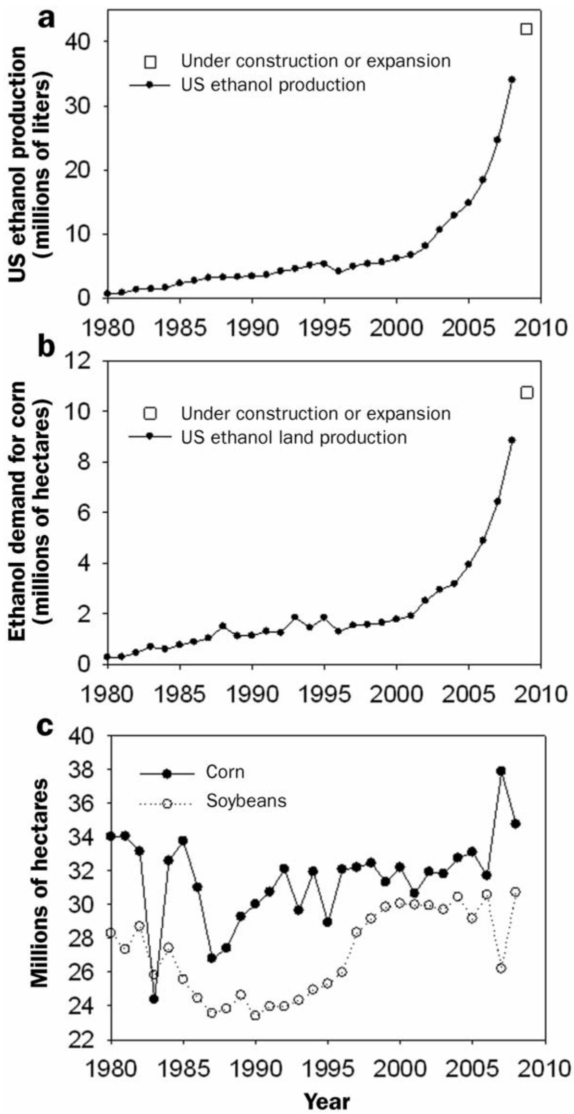

Figure 2. US ethanol production (RFA 2008), land demand for ethanol production, and area planted for corn and soybeans (USDA 2009). Land demand for ethanol production is based on each year's actual yields, area planted, and area harvested (USDA 2009).

had previously been planted alternately with corn and soybeans now is planted continuously with corn. This practice lowers yields and increases nutrient additions and emissions, as discussed below. Also, some land that is now used to produce corn was under perennial vegetation, primarily grasses, just several years ago.

Data on exactly how much grassland has been converted to corn production are not available. However, several lines of evidence indicate that grassland has been and will be converted to crop production as a result of the higher demand for corn.

First, the amount of land enrolled in the CRP peaked at 14.9 million hectares (ha) in September 2007. In October 2007, CRP lands had declined by 931,000 ha (USDA 2007). Of those lands no longer in the program, 850,000 ha were grasslands, and the remainder had been enrolled to promote tree or wetlands conservation practices. Second, the Food, Conservation, and Energy Act of 2008 reduced the total area that may be enrolled in the CRP to 12.9 million ha by 2010, which ensures that the trend of expiring CRP acres and declining enrollments will continue. This mandate reduces the ceiling of allowable area, but it does not provide a floor of required area, so it is unclear how deep the loss of CRP-enrolled lands will ultimately be. The US Department of Agriculture has projected that CRP area will bottom out at 12.2 million ha in 2013 before rebounding to 12.9 million ha in 2017 (USDA 2009). Economic analyses, however, suggest the potential for deeper losses. Secchi and Babcock (2007) estimated that 49% to 61% of the land enrolled in the CRP in Iowa would eventually be converted back to cropland if corn prices were fixed at \$3 or \$4 per bushel, respectively, for an extended period. Given that corn prices ranged from \$3 to \$7 per bushel in 2008 and are projected to remain greater than \$3.65 until 2018 (USDA 2009), a significant drop in CRP area in Iowa is likely to occur. As a final piece of evidence of CRP losses, the Farm Service Agency indicates that more than 345,000 ha of the 3.2 million ha of CRP land in the prairie pothole region of the Northern Great Plains expired in 2007. Another 1.4 million ha will expire from 2008 to 2012 unless new opportunities to reenroll in CRP become available (figure 3).

Not all of the grassland being converted to cropland has been cropped in the past. Some of the land currently being converted to cropland is native prairie that has been pastured but never plowed. This land is vulnerable to conversion as a result of both higher crop prices and profits, and challenging grazing economics. For example, cropland conversion totaled more than 203,000 ha of native prairie in North Dakota, South Dakota, and Montana between 2002 and 2007 (Scott Stephens, Director of Conservation Planning and Programs, Ducks Unlimited, Bismarck, North Dakota, personal communication, 30 March 2009), and 5.2% (36,540 ha) of remaining native grassland in the Missouri Coteau of North Dakota and South Dakota was lost from 1984 to 2003 (Stephens et al. 2008).

Significant investment in the ethanol industry over the past few years, buoyed by renewable fuel mandates and industry subsidies, means that corn-ethanol production capacity in this country will continue to grow strongly. As of late 2008, the United States had a 42-billion-liter annual capacity (RFA 2008). Ongoing construction (including new plants and expansion of existing plants) will result in a production capacity of about 50 billion liters. Capacity would have to go even higher to meet goals in the renewable fuels standard of the Energy Independence and Security Act of 2007 (EISA; Sissine 2007), which mandates production of 57 billion liters of biofuel by 2015, all of which are expected to be made from corn grain. Assuming an industrywide conversion rate of 10.6 liters of ethanol per bushel of corn (current conversion rates are about 10.4 liters per bushel; FAPRI 2008), an average annual corn yield of 417 bushels per ha (current yields are 380 bushels per ha; USDA 2009), and that 98% of

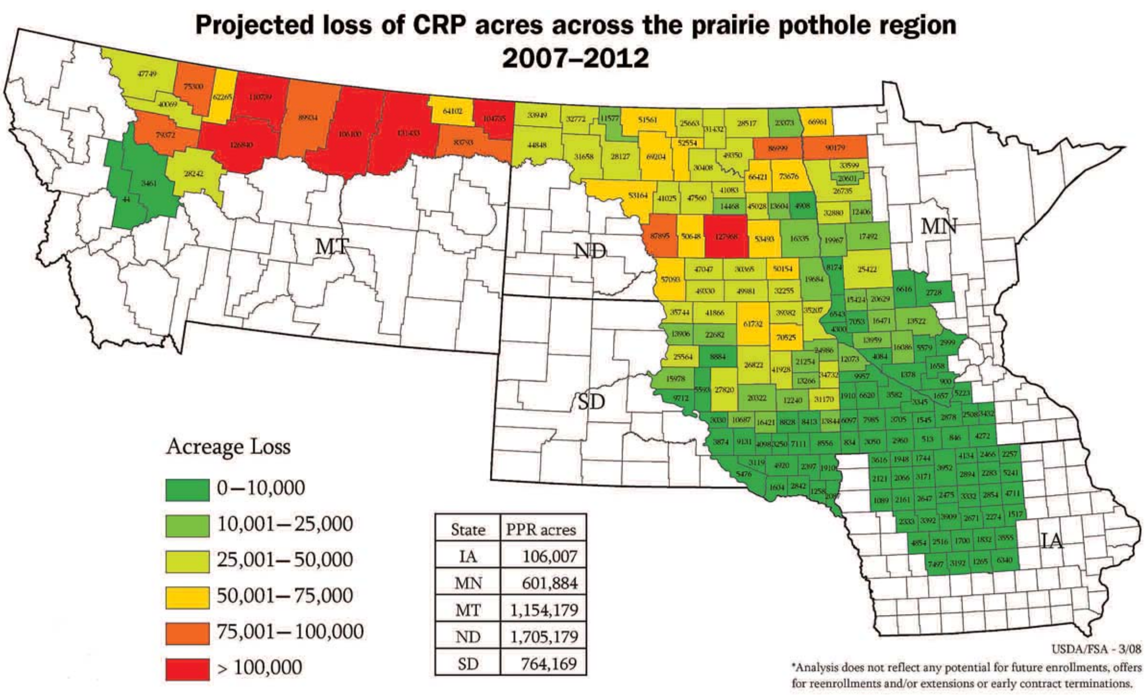

Figure 3. Projected loss of Conservation Reserve Program (CRP) acres across the prairie pothole region from 2007 to 2012. This is calculated as the sum of the acres whose contracts expire in these years and whose owners have declined or were not offered the chance to reenroll in CRP.

planted hectares are harvested (historic rate), meeting the 57-billion-liter mandate with corn ethanol would require about 13.1 million ha of planted corn, or about 6.7 million more ha of corn than was planted for ethanol production in 2006. The net increase in demand for cropland will be less than this, because corn ethanol production also yields the coproduct "distiller's grain," which is used as animal feed and displaces corn and soybean meal (e.g., Klopfenstein et al. 2008), but the land-use impact of this displacement is poorly quantified. We expect that some of the expansion of corn ethanol production will come at the expense of perennial grassland conversion, judging from the analyses and observed losses discussed above.

Potential impacts of corn ethanol on wildlife and fisheries

The conversion of CRP land to cropland has potentially significant impacts on grassland wildlife such as nesting birds and mammals (Reynolds 2005, Herkert 2007, Niemuth et al. 2007). For example, Herkert (2007) showed that population trends for the Henslow's sparrow (*Ammodramus henslowii*) in Illinois counties were related to the amount of CRP land, and attributed the recovery of this species primarily to the increase in perennial grasslands created by the CRP. Results from a study on the value of CRP to grassland birds in North and South Dakota indicated that almost two million birds of five grassland nesting species would be lost without the CRP in those two states (Niemuth et al. 2007). Reynolds (2005) estimated that CRP habitat in the prairie pothole region adds an additional 2.1 million ducks annually to the fall flight.

To meet the greater demand for corn, many farmers have planted corn in the same field continuously from year to year. Compared with the more common corn and soybean rotation, continuous corn planting requires more fertilizer inputs (Katsvairo and Cox 2000), results in greater nitrogen leaching, is more susceptible to buildups of soil pathogens, and lowers annual yields by about 14% (Pikul et al. 2005). Lower yields from continuous corn mean that, in a given year, more land is required to meet the same demand, thus increasing competition with wildlife for land. Corn also requires more fertilizer than soybeans do, especially when it does not follow soybeans in a crop rotation (soybeans increase soil fertility because they fix atmospheric nitrogen). Moreover, it is more difficult to use conservation tillage on continuous corn because the buildup of residue leads to lower yields in subsequent years (Wilhelm and Wortmann 2004). Thus, continuous corn planting may reduce the amount of land in conservation tillage and intensify soil erosion.

Conversion of grassland to corn has significant impacts on freshwater ecosystems. Intact grasslands retain soil and nitrogen—for example, the amount of nitrate leaving tile-drained CRP grasslands was 98% lower than the amount leaving continuous corn (Randall et al. 1997). Sediment increases turbidity, raising temperatures and degrading habitat for coldwater fish such as trout. Nitrates are carried through freshwater systems, leading to algal blooms and hypoxia, creating "dead zones" such as the one in the Gulf of Mexico. In 2007, the dead zone in the Gulf of Mexico was 65% larger than average (1990–2006), and in 2008 it reached its second-largest size ever at 20,689 square kilometers (km^2)

(NOAA 2007, 2008). Producing the mandated 57 billion liters of corn ethanol will make it practically impossible to meet the federal goal of reducing the dead zone to less than 5000 km^2, according to Donner and Kucharik (2008).

Ethanol production requires substantial water use. Ethanol factories use 3 to 5 liters of water to produce 1 liter of ethanol (Keeney and Muller 2006). However, water usage in ethanol production is dwarfed by the amount of water needed to grow corn. Irrigated corn requires about 785 liters of irrigation water for every liter of ethanol produced (Aden 2007). About 19% of US corn comes from irrigated land (figures on area irrigated are from USDA [2004]; irrigated yields data are from Aden [2007]). This means that ethanol, on average, requires about 147 liters of irrigation water for every liter of ethanol produced. About 70% of this water is lost in crop production (primarily through transpiration and evaporation), and about 30% is returned to the surface and groundwater through runoff and infiltration (Mubako and Lant 2008). Although water may be used sustainably, in some places it is being removed at unsustainable rates from aquifers or it competes with other uses of surface waters, including the maintenance of aquatic biodiversity (Roberts et al. 2007).

Potential bioenergy sources

There are other possible options for future bioenergy sources, many of which would quite likely replace wildlife habitat with bioenergy crops and negatively affect wildlife. However, at least two ways of producing bioenergy may be compatible with wildlife. The first is to use biomass sources that do not require additional land, and thus do not increase the footprint of agriculture. The second is to produce biomass with land-use practices that are compatible with wildlife. Biomass sources that do not require additional land include wastes such as agricultural residues, cover crops, and, potentially, algae. Practices that are compatible with wildlife may include a variety of perennial biomass crops. Whether a particular project has effects that are negative, neutral, or positive for wildlife will depend on explicit consideration of wildlife impacts in the project-planning stages, and on actions taken to avoid incompatible land uses and management practices.

Wastes can be used to create bioenergy (fuel, heat, electricity) without requiring additional land. Potential sources include wastes from agricultural, municipal, animal, food industry, and forestry sources. Depending on how much cellulosic ethanol efficiencies can be improved, it would require 199 million to 282 million metric tons of biomass to meet the current renewable fuels standard of 79 billion liters of advanced biofuel by 2022 (mandated in addition to the 57 billion liters that can be supplied by corn ethanol). The US Department of Energy and the Department of Agriculture (Perlack et al. 2005) estimated that with 25% increases in yield, annual supplies of crop residues could provide 244 million metric tons (however, maintaining soil organic carbon may limit potential residue removal; Wilhelm et al. 2007), process residues could provide 36 million metric tons, and manure could provide 40 million metric tons (Perlack et al. 2005) of material suitable for bioenergy production. Animal waste from concentrated feeding operations can produce methane that can be burned to produce electricity. Forestry waste is available from logging and sawmills, forest thinning (e.g., for fuel-load reduction), packaging and durable good wastes, and from storm- or pest-damaged trees. However, the retention of fine and coarse woody debris after logging is essential to maintain the wildlife value of forests (Pedlar et al. 2002). To avoid unintended consequences, plans to increase the removal of woody biomass from logged sites need to be carefully evaluated for their potential impacts on wildlife. Although the use of mill waste does not carry such risks, the potential to expand that use is relatively small since most mill waste is already used for energy or other coproducts. The unexploited capacity of forestry waste residues for bioenergy production is estimated at 70 million metric tons annually in the United States, with an additional potential of 54 million metric tons annually from fuel-load reductions (Perlack et al. 2005).

The most commonly discussed agricultural by-product is corn stover (leaves and stalks remaining in a field after harvest). Corn stover is produced in large quantities, may be relatively inexpensive, and is a uniform feedstock. However, the use of corn stover raises environmental concerns because of increased soil erosion (Graham et al. 2007) and further depletion of soil organic carbon stocks (Wilhelm et al. 2007). If concerns about wind and soil erosion are addressed, some 54 million metric tons of stover could be collected annually (Graham et al. 2007). However, this does not take into account concerns about depleting organic soil carbon stocks, which not only would increase carbon dioxide emissions, and thus contribute to climate change, but also may reduce yields (Wilhelm et al. 2007). Promisingly, long-term research suggests stover removal may be sustainable in terms of yields, soil quality, and soil carbon if practiced in combination with no-till farming (Moebius-Clune et al. 2008). The use of stover or other agricultural residues or cover crops could reduce the amount of habitat converted to bioenergy production because it can be supplied from land currently planted in corn. Conversely, if corn stover boosts the profits associated with corn production, this could lead to increased corn production and greater conversion of habitat to corn. The use of corncobs in cellulosic ethanol production would increase the amount of ethanol produced per ha by about 25% over the use of corn alone, without raising concerns over reductions in soil carbon.

Algae, which do not require soil for growth, have also been proposed as a source of bioenergy (Sheehan et al. 1998). Algae can be grown in freshwater or saltwater, and thus conflicts with wildlife can be avoided more readily than is the case with other bioenergy crops. Algae can also have extremely high yields (45 metric tons per ha per year). From an aquatic wildlife perspective, however, there could be unintended impacts on habitat quality (e.g., the release of modified algae could invade natural ecosystems).

Several energy crops have been proposed, including native species such as switchgrass (*Panicum virgatum*) and big bluestem (*Andropogon gerardii*), and exotic species such as *Mis-*

canthus (*Miscanthus giganteus*), common reed (*Phragmites australis*), reed canary (*Phalaris arundinacea*), hybrid poplar (*Populus* spp.), and camelina (*Camelina sativa*). Often, the introduction of exotic plant species produces undesirable consequences for native habitats and native wildlife species. Native wildlife species have not evolved with monocultures of exotic plants, and they may not be able to use such monocultures as habitat. For example, *Miscanthus* produces nine-foot-tall thickets (similar to bamboo) that are unlike the plant communities with which native North American species have evolved. Proposals to plant woody crops in areas typically dominated by grasslands raise similar concerns about wildlife impacts.

In general, the net effect of crops on wildlife will depend on the land use that they are replacing. Perennial energy crops are likely to provide better habitat than annual crops. For example, compared with corn, monocultures of switchgrass benefit some bird species of management concern, while other bird species have shown no benefit (Murray et al. 2003). Similarly, *Miscanthus* may provide better habitat than annual crops, although this may be a transient response associated with greater weed abundance in recently established *Miscanthus* fields (Bellamy et al. 2009). However, perennial crops can be grown in places not suited to existing crops, such as some existing grassland, thus potentially posing a broader threat of conversion to wildlife habitat than existing biofuel crops.

Assessing potential impacts on wildlife

Biomass crops may provide habitat if they are similar to native ecosystems, depending on the harvest management of the crops. In addition, biomass crops may pose a risk of off-field negative impacts if they become invasive and spread beyond field borders. Similarity to native ecosystems, harvest management, and invasive potential are reviewed below.

Risk of invasiveness. Biomass crops may pose a risk of becoming invasive if exotic crop species are used, if exotic or native species are modified through breeding or genetic engineering, or if species native to the United States are used outside their home range (Raghu et al. 2006, Barney and Ditomaso 2008). If native species are bred to increase yield, they may differ significantly from unmodified cultivars. Native or exotic species may be genetically modified to promote cultivation, yield, or other characteristics affecting bioenergy usage. Breeding and genetic modification of species may make species more likely to become invasive, as desirable agronomic traits such as a fast growth rate and high establishment success are also associated with successful invasive species (CAST 2007). Because biomass crops are typically harvested after they have set seed, there is opportunity for propagule spread before harvest or during transport. This increases the risk of invasion, which rises with greater propagule pressure (the number of seeds that are released to the environment). *Miscanthus giganteus* is a naturally occurring hybrid with sterile seed, which reduces its risk of becoming invasive. However, *M. giganteus* still poses a risk of invasion through rhizomes; further, continued sterility is not guaranteed, and any variety with viable seed could spread rapidly (Raghu et al. 2006).

Similarity to native ecosystems. The diverse prairie ecosystem has been proposed as a bioenergy source with unique benefits for wildlife and carbon sequestration (Tilman et al. 2006). Diverse prairie is dominated by perennials, obviating the soil erosion, energetic, and financial costs associated with annual planting. When cropland is planted to perennial plants, soil carbon increases (FAPRI 2007). Diverse prairie communities have higher rates of carbon sequestration than do monocultures or low-diversity prairie (Tilman et al. 2006). In particular, seed mixes that include legumes, which fix nitrogen, result in dramatically increased rates of carbon storage compared with the mixes of several warm-season grasses commonly used in conservation practices (Fornara and Tilman 2008). The risk of invasion is greatly reduced when using native species of local ecotype. Because these communities are relatively self-sustaining, few fertilizers or herbicides are needed (at least after initial establishment), reducing the environmental and energetic costs associated with these inputs. Thus, even though perennial monocultures tend to require lower inputs than do annual crops, diverse prairie grasses require even fewer inputs.

Diverse communities also benefit wildlife. Experimental manipulations of biodiversity show that insect diversity is positively correlated with plant diversity (Haddad et al. 2001). The nectar produced by forbs in grasslands supports insects that can benefit insect-pollinated crops in nearby fields (Ockinger and Smith 2007). The benefit of plant diversity to wildlife also appears to hold higher up the food chain—for example, a survey of Wisconsin grasslands found that the diversity of birds was positively correlated with plant diversity (Sample 1989). Thus, although perennial monocultures and perennial polycultures both provide more wildlife benefits than corn does, diverse mixtures provide the most.

Harvest management. Without periodic management to reduce the litter layer and encourage new growth, grasslands produce less biomass (Knapp and Seastedt 1986) and lose their habitat value for many wildlife species (e.g., Roth et al. 2005). This highlights the potential for biomass harvests to increase the wildlife value of grasslands, but that potential will be realized only if wildlife values and landscape context are taken into consideration in harvest planning.

Harvest management of biomass fields will play a large role in determining vegetation structure, and thus the fields' value for wildlife habitat. Harvest management considerations include the seasonal timing of harvest, the height at which vegetation is harvested, and the proportion of available grassland that is harvested. Grassland bird species are adapted to particular ranges of habitat conditions (e.g., Sample and Mossman 1997). For example, some species prefer short stubble, which allows them to detect predators, and other species prefer long stubble, which allows them to avoid detection by predators (Whittingham et al. 2006). Extensive harvest of vegetation will very likely favor grassland birds requiring short, sparse vegetation (e.g., grasshopper sparrow [*Ammodramus savannarum*] and Savannah sparrow [*Passerculus sandwichensis*]) and negatively affect those requiring tall, dense vegetation (e.g., sedge wren [*Cistothorus platensis*] and Henslow's sparrow). The best harvest scenario is likely to be one that produces a mosaic of harvested and unharvested patches, but further research is needed to determine the appropriate scale of these patches. Small habitat patches may suffer higher predation rates, making these patches population sinks rather than sources.

The proper time to harvest depends on the species of management concern, whether those species are migratory or resident, and the timing of the life-cycle events that have the greatest impact on populations (nesting, brood rearing, winter, etc.). Harvest should not occur during the established primary nesting season (PNS) (figure 4). Biomass could be harvested either before or after PNS. From a wildlife perspective, having multiple harvest times (early fall, postfrost, early spring) could provide a mosaic of habitat conditions suiting a wider range of species, as well as provide feedstocks to a biomass facility at different times of the year. However, depending on the species of management concern, either fall or spring harvests may be preferred. Harvesting in early spring would collect less biomass because of lodging (i.e., plants falling over) during the winter, but may be beneficial if biomass storage space is limited, and would benefit wildlife that require winter or residual cover, such as harriers (*Circus cyaneus*), pheasants (*Phasianus colchicus*), sedge wren, and Henslow's sparrow (George et al. 1979, Evrard and Bacon 1998, Roth et al. 2005). Early spring harvests must occur before the established PNS for each state to minimize impacts on grassland birds. Fall harvests typically occur after the first killing frost, well after the PNS for grassland birds. Earlier harvests, timed to coincide with the end of the nesting season, may benefit wildlife by allowing sufficient regrowth to provide winter cover and spring nesting. However, the effects of earlier harvest on the productivity and composition of the biomass crop are not well known and should be monitored to avoid unintended shifts in composition.

Residual cover (i.e., stubble) is of paramount importance to nesting ducks and other birds, particularly early nesting species such as mallard (*Anas platyrhynchos*) and northern pintail (*Anas acuta*) that arrive on northerly breeding grounds before the onset of the growing season (e.g., Jarvis and Harris 1971). Nest success for grassland nesting ducks increases with the height, structure, and amount of residual cover on the landscape. However, because it is unclear what stubble height would allow both sufficient nesting habitat for ducks and reasonable biomass yield, research is needed to understand the trade-offs between leaving stubble for ground-nesting birds and other wildlife and harvesting stubble for increased biomass yields.

Stubble may also benefit soils and yields. Stubble may reduce soil erosion caused by wind, particularly in northern climates that experience snowfall. The presence of stubble will help catch and maintain snow cover, which can improve spring soil moisture and may boost yields of desired perennial grasses. Research is needed to determine whether there is a relationship between stubble height and subsequent yields, and if so, what minimum and maximum stubble heights will produce the desired benefits.

The ideal proportion or configuration of unharvested to harvested land to maximize the wildlife benefit is not yet known. For example, would it be better to leave 20% of each field unharvested, or to let one out of five fields go unharvested, to serve as refuges? Research on nesting waterfowl in the prairie pothole region clearly indicates that nesting success

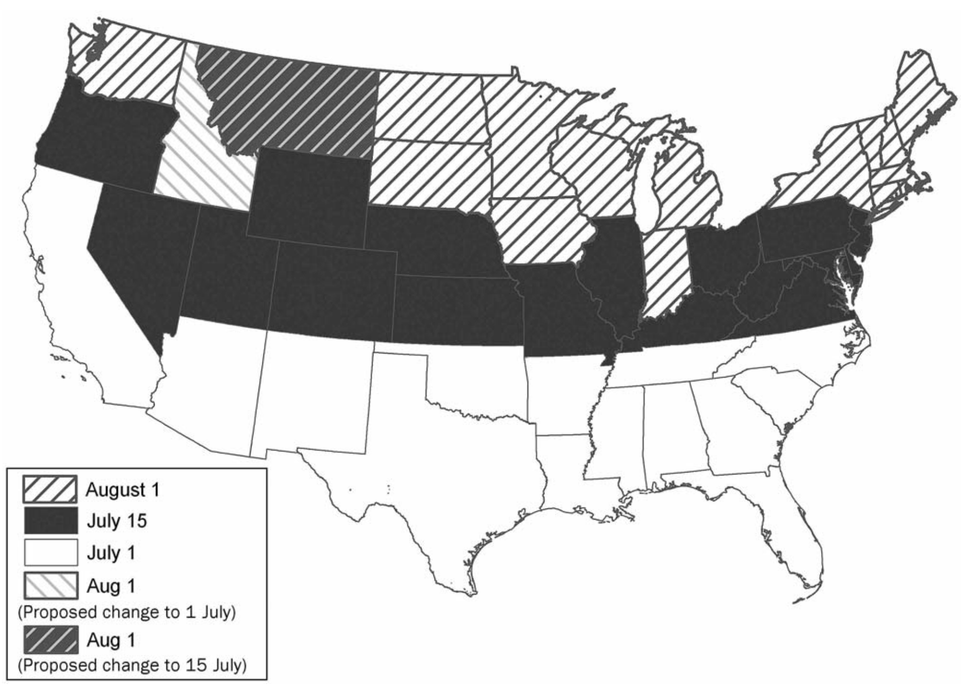

Figure 4. Established ending dates for primary nesting season for the purposes of management on Conservation Reserve Program (CRP) lands (USFSA 2008). These dates are established by National Resources Conservation Service rulemaking under federal law, and any management that occurs on CRP land, such as emergency haying or mid-contract management, must occur outside of primary nesting season.

increases with the amount of perennial cover, as measured across a range of scales. Species abundance of grassland birds is highly dependent on landscape context (e.g., Cunningham and Johnson 2006). Larger blocks of grassland are more likely to provide nesting and winter cover for a wide range of bird species (e.g., Winter et al. 2006). Wildlife-friendly bioenergy crops are most likely to achieve the most wildlife benefit if they are components of landscapes that already contain a large portion of grasslands, rather than isolated fragments among cropland. This could be especially beneficial for area-sensitive, grassland wildlife species such as prairie chickens and Henslow's sparrows. Bioenergy facilities will also benefit from being located in landscapes with high perennial biomass production. Thus, if bioenergy demand is met with biomass production that is compatible with wildlife, the location of bioenergy facilities near grassland habitat could benefit both wildlife and the bioenergy industry.

Even grasslands primarily managed for wildlife could provide biomass for bioenergy. Biomass on these lands could periodically be harvested as part of normal establishment and management practices aimed at, for example, controlling invasive weeds such as Canada thistle (*Cirsium arvense*) and leafy spurge (*Euphorbia esula*); providing an alternative to burning to control woody encroachment or litter buildup; and supplying short, sparse vegetation for species requiring this structure.

Assessing the feasibility of wildlife-friendly bioenergy crops

Although it is possible to produce biomass in ways that are compatible with wildlife, there are several open questions about its feasibility. Specifically, native perennial crops need to be feasible from the standpoints of economic, agronomic, and technological considerations, and of land and seed availability, if they are to become a significant portion of the energy portfolio.

One concern associated with the use of low-input, native prairie grasses is their yield relative to that of other proposed bioenergy crops (Schmer et al. 2008). Because there are no direct comparisons of native prairie grasses with other potential biomass crops using the same site, soils, and climate, it is premature to draw firm conclusions about yield differences. Comparisons of yields from different biomass crops at different sites, often with different rates of fertilization or other management practices, are problematic because farm trials generally occur on high-yielding cropland, whereas prairie yields are often measured on low-yielding lands that are unsuitable for farming. In farm trials, yields of fertilized switchgrass in North Dakota, South Dakota, and Nebraska averaged between 5.2 and 11.1 metric tons per ha (Schmer et al. 2008). In those same states, unfertilized prairie yields ranged from 3.4 to 5.7 metric tons per ha (Risser et al. 1981). *Miscanthus* is among the highest-yielding biomass crops, with fertilized yields in Europe averaging 22 metric tons per ha (Heaton et al. 2004). *Miscanthus* has high water demands, and this yield average included irrigated field trials (Heaton et al. 2004). Yields of unfertilized native prairie grasses of up to 13.7 metric tons per ha have been reported from Illinois (Oesterheld et al. 1999). These examples illustrate high-yield potential from prairie grasses but also reveal a gap between reported yields for prairies and fertilized bioenergy crops. Direct comparisons of different potential biomass crops and native prairie on similar soils and under similar fertilization and irrigation regimes are needed to accurately quantify yield differences on a given site. It may also be possible to fertilize native prairie in a way that increases its yields while maintaining its wildlife value. Although fertilization typically reduces the diversity of plant communities, it may be possible to maintain plant diversity in communities that are both fertilized and harvested (Collins et al. 1998).

Establishing diverse mixtures of native perennial vegetation is expensive at present, in part because of high seed costs, which may initially hinder the large-scale establishment of diverse prairie grasses for bioenergy production. To encourage the use of diverse mixtures and their associated wildlife benefits, the government, bioenergy industry, and conservation community would need to work together to increase supply and lower seed prices or otherwise offset the higher cost of seeds. Cost-share programs could share establishment costs for projects resulting in quantifiable benefits for targeted wildlife populations or for projects allowing public access for recreation. Federal, state, and nongovernmental wildlife organizations could help provide the technical expertise needed for successfully establishing native grasslands, reducing costs associated with poor establishment.

Production costs of native grasses are estimated at $39 to $61 per metric ton, including land rental rates (Tiffany et al. 2006). This would be reduced to $22 to $33 per metric ton if land rental rates were excluded (cost estimates do not include any capital or hired labor costs). Transportation costs vary greatly depending on the size of the source area (Tiffany et al. 2006), and average costs increase from about $3.48 to $12.08 per metric ton as the source radius increases from 16 to 80 km. Community-scale projects with modest biomass requirements or higher-yielding crops would allow smaller source areas for biomass production, significantly reducing transportation costs.

It is unclear whether biomass fermentation processes currently under development will call for uniform feedstocks, which could limit the use of diverse prairie in ethanol production. However, diverse plantings can be burned to produce heat and electricity, or gasified to produce heat and electricity, or gasified to produce syngas, which is converted through the Fischer-Tropsch or other catalytic processes to gasoline, diesel, or ethanol (McKendry 2002). Cogeneration, the production of both heat and electricity, can be an extremely efficient way to extract energy from biomass through either burning or gasification (McKendry 2002).

Recent global analyses suggest that approximately 385 million to 472 million ha of abandoned farmland could be used to produce approximately 1.4 billion to 2.1 billion metric tons of biomass annually (Campbell et al. 2008). In the

United States, CRP contracts allow haying and grazing management, if that is written into the CRP contract with the landowner. However, harvest should follow the management guidelines in the contract, and some lands should not be eligible for harvest because of their slope; the presence of wetlands; or their importance to wildlife species of local, state, or national concern. Additional research is needed to identify where suitable lands occur in sufficient densities to support bioenergy facilities.

Landscape and adaptive management considerations

Whether bioenergy production is beneficial to wildlife or not will depend on many factors in addition to the composition of the crops. Most important, it will depend on the landscape context in which the bioenergy crops are planted. To deal with these external factors, managers should have explicit objectives, defined at the correct scale, and use adaptive management to tailor practices to local and changing conditions.

Management for wildlife could focus on overall biodiversity, on particular species groups, or on specific species. Managing for specific species is often the easiest task, especially when the ecological needs of the species are well understood. The US Fish and Wildlife Service has developed a series of habitat evaluation tools that help land managers evaluate impacts on fish and wildlife habitat resulting from changes in water or land use, as well as assess the suitability of habitat for fish and wildlife species (USFWS 1980). Species-specific habitat suitability index (HSI) models (USFWS 1981) use quantitative relationships between environmental variables and habitat suitability to arrive at a numerical index of habitat suitability (USFWS 1981). There are currently HSI models for 157 species, many of which use prairie or grassland habitats (USGS 2008).

When managing to maximize overall biodiversity, it may be possible to choose umbrella species to represent the habitat needs of other groups (e.g., Mac Nally and Fleishman 2004). Generally, a diverse array of native plants provides food for a diversity of native herbivores and nectivores, particularly insects, which in turn can provide food for a diversity of birds (Sample 1989, Haddad et al. 2001).

It is not possible to know a priori the exact management practices and species combinations that will simultaneously optimize bioenergy production and benefit wildlife under all conditions over time. Therefore, in parallel with an emerging bioenergy industry, both ongoing monitoring and experiments are important to provide site-specific information and allow the industry to adapt as learning occurs, technologies emerge, and conditions change. Adaptive management includes clearly defined and measurable management objectives, monitoring or experiments to assess progress toward objectives, and adjustment in response to measured outcomes (e.g., Wilhere 2002). Inclusion of such adaptive management experiments in major bioenergy projects offers the best chance of creating projects that provide both bioenergy and wildlife benefits.

Policy and carbon emissions

US policy promoting biofuels has been driven primarily by interest in energy independence, rural economic development, and reducing GHG emissions. Given the large land-use implications of biofuels policy, the wildlife conservation implications of policy also merit consideration. Production of biofuel crops that leads to direct or indirect clearing of natural habitats will harm wildlife and, when the full costs of production and use are considered, are likely to increase carbon emissions (Fargione et al. 2008, Searchinger et al. 2008). Thus, policies requiring biofuels to meet carbon emission standards (now being discussed in the implementation and interpretation of EISA and various state policies) are likely to benefit wildlife by discouraging some types of conversion of natural habitat resulting from biofuel production. The establishment of carbon markets that provide economic incentives to reduce carbon emissions from natural ecosystems will also benefit wildlife. Maintaining or increasing terrestrially stored carbon, however, is not enough to guarantee wildlife benefits. For example, growing *Miscanthus* for bioenergy and converting native grassland to do so would most likely have negative impacts on wildlife even though it would probably reduce carbon emissions from petroleum use. This indicates a need for policy that goes beyond carbon considerations to explicitly address sustainability standards for biomass production, including the impacts on wildlife.

Summary and conclusions

The area in the United States devoted to corn crops is increasing, partially at the expense of perennial grasslands, with negative effects on wildlife and water quality. The recent corn ethanol boom has already been associated with the loss of more than 850,000 ha of set-aside grassland in the United States and with a 4.9-million-ha increase in corn cropland used for ethanol between 2005 and 2008. Evidence for current and future impacts on grasslands includes data on declining CRP enrollment, increasing corn area, conversion of virgin prairies, and economic analyses of future CRP enrollment. The increase in land area in grasslands from CRP starting in 1986 has had clear wildlife benefits for birds, fish, and other taxa, and for freshwater stream ecosystems in general. These benefits will erode, and wildlife populations and water quality will decline, as CRP land is lost. Thus, increased corn production for ethanol threatens wildlife and ecosystem services.

New conservation strategies are needed to protect grassland wildlife habitat. Increases in conservation payments, while needed, may reduce only a relatively small portion of expected habitat loss. Using new markets for biomass offers the tantalizing prospect of maximizing the amount of perennial grassland, land that could benefit wildlife, provide income to farmers, and contribute to domestic renewable energy production. By incorporating wildlife, water quality, carbon sequestration, and other ecosystem services in the up-front planning and consideration of biomass feedstocks, incentives could be used to encourage farmers to grow and harvest biomass for bioenergy using practices that simultaneously provide society with multiple benefits. Opportunities to harvest native perennial plants may provide incentives to keep land in current conservation programs. However, additional incentives or regulations would be required to ensure that planting and management decisions made on the basis of short-term biomass yield, or yield and carbon sequestration, also benefit wildlife.

We have suggested several important research directions that would help bioenergy fulfill its promise of sustainable energy production. These include bettering our understanding of the effects of conversion of natural habitat to bioenergy production; researching the effects of crop or plant community composition, annual harvests, refugia, stubble height, and minimal fertilization on sustainable yield and wildlife and plant diversity; and investigating the possibility of using biomass sources that do not require a bigger agricultural footprint, such as from agricultural and other wastes. Natural resource managers and environmental scientists are well positioned to inform the policies and practices of bioenergy production. Providing society with the multiple benefits of sustainable energy and a sustainable environment will require increased partnership between natural resource managers and the bioenergy industry.

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References cited
Aden A. 2007. Water usage for current and future ethanol production. Southwest Hydrology (September/October): 22–23.

Barney JN, Ditomaso JM. 2008. Nonnative species and bioenergy: Are we cultivating the next invader? BioScience 58: 64–70.

Bellamy PE, Croxton PJ, Heard MS, Hinsley SA, Hulmes L, Hulmes S, Nuttall P, Pywell RF, Rothery P. 2009. The impact of growing *Miscanthus* for biomass on farmland bird populations. Biomass and Bioenergy 33: 191–199. doi:10.1016/j.biombioe.2008.07.01

Campbell JE, Lobell DB, Genova R, Field CB. 2008. The global potential of bioenergy on abandoned agricultural lands. Environmental Science and Technology 42: 5791–5794.

[CAST] Council for Agricultural Science and Technology. 2007. Biofuel Feedstocks: The Risk of Future Invasions. CAST.

Collins SL, Knapp AK, Briggs JM, Blair JM, Steinauer EM. 1998. Modulation of diversity by grazing and mowing in native tallgrass prairie. Science 280: 745–747.

Cunningham MA, Johnson DH. 2006. Proximate and landscape factors influence grassland bird distributions. Ecological Applications 16: 1062–1075.

Donner SD, Kucharik CJ. 2008. Corn-based ethanol production compromises goal of reducing nitrogen export by the Mississippi River. Proceedings of the National Academy of Sciences 105: 4513–4518.

Evrard JO, Bacon BR. 1998. Northern harrier nest site characteristics. Passenger Pigeon 60: 305–312.

[FAPRI] Food and Agricultural Policy Research Institute. 2007. Estimating Water Quality, Air Quality, and Soil Carbon Benefits of the Conservation Reserve Program. FAPRI-UMC report 01-07. (10 June 2009; *www.brc.tamus.edu/swat/applications/FAPRI_UMC_Report_01_07.pdf*)

———. 2008. U.S. and World Agricultural Outlook, January 2008. FAPRI Staff Report 08-FSR 1. (10 June 2009; *www.fapri.missouri.edu/outreach/publications/2008/OutlookPub2008.pdf*)

Fargione J, Hill J, Tilman D, Polasky S, Hawthorne P. 2008. Land clearing and the biofuel carbon debt. Science 319: 1235–1238.

Foley JA, et al. 2005. Global consequences of land use. Science 309: 570–574.

Fornara DA, Tilman D. 2008. Plant functional composition influences rates of soil carbon and nitrogen accumulation. Journal of Ecology 96: 314–322.

George RR, Farris AL, Schwartz CC, Humburg DD, Coffey JC. 1979. Native prairie grass pastures as nest cover for upland birds. Wildlife Society Bulletin 7: 4–9.

Graham RL, Nelson R, Sheehan J, Perlack RD, Wright LL. 2007. Current and potential US corn stover supplies. Agronomy Journal 99: 1–11.

Haddad NM, Tilman D, Haarstad J, Ritchie M, Knops JMH. 2001. Contrasting effects of plant richness and composition on insect communities: A field experiment. American Naturalist 158: 17–35.

Heaton E, Voigt T, Long SP. 2004. A quantitative review comparing the yields of two candidate C-4 perennial biomass crops in relation to nitrogen, temperature and water. Biomass and Bioenergy 27: 21–30.

Herkert JR. 2007. Evidence for a recent Henslow's sparrow population increase in Illinois. Journal of Wildlife Management 71: 1229–1233.

Jarvis RL, Harris SW. 1971. Land-use patterns and duck production at Malheur National Wildlife Refuge. Journal of Wildlife Management 35: 767–773.

Katsvairo TW, Cox WJ. 2000. Economics of cropping systems featuring different rotations, tillage, and management. Agronomy Journal 92: 485–493.

Keeney D, Muller M. 2006. Water Use by Ethanol Plants: Potential Challenges. Institute for Agriculture and Trade Policy. (10 June 2009; *http://agobservatory.org/library.cfm?refID=89449*)

Klopfenstein, TJ, Erickson GE, Bremer VR. 2008. Board-invited review: Use of distillers by-products in the beef cattle feeding industry. Journal of Animal Science 86: 1223–1231.

Knapp AK, Seastedt TR. 1986. Detritus accumulation limits productivity of tallgrass prairie. BioScience 36: 662–668.

Mac Nally RM, Fleishman ER. 2004. A successful predictive model of species richness based on indicator species. Conservation Biology 18: 646–654.

McKendry P. 2002. Energy production from biomass, pt. 2: Conversion technologies. Bioresource Technology 83: 47–54.

Moebius-Clune BN, van Es HM, Idowu OJ, Schindlebeck RR. 2008. Long-term effects of harvesting maize stover and tillage on soil quality. Soil Science Society of America Journal 72: 960–969.

Mubako S, Lant C. 2008. Water resource requirements for corn-based ethanol. Water Resources Research 44: W00A02. doi:10.1029/2007WR006683

Murray LD, Best LB, Jacobsen TJ, Braster ML. 2003. Potential effects on grassland birds of converting marginal cropland to switchgrass biomass production. Biomass and Bioenergy 25: 157–175.

Niemuth ND, Quamen FR, Naugle DE, Reynolds RR, Esty ME, Shaffer TL. 2007. Benefits of the Conservation Reserve Program to Grassland Bird Populations in the Prairie Pothole Region of North Dakota and South Dakota. Report prepared for the US Department of Agriculture Farm Service Agency. RFA OS-IA-04000000-N34. (27 July 2009; *www.fsa.usda.gov/Internet/FSA_File/grassland_birds_fws.pdf*)

[NOAA] National Oceanographic and Atmospheric Administration. 2007. NOAA and Louisiana scientists say Gulf of Mexico "dead zone" could be largest since measurements began in 1985. (10 June 2009; *www.noaanews.noaa.gov/stories2007/s2891.htm*)

———. 2008. Survey cruise records second-largest "dead zone" in Gulf of Mexico since measurements began in 1985. (10 June 2009; *www.noaanews.noaa.gov/stories2008/20080728_deadzone.html*)

Ockinger E, Smith HG. 2007. Semi-natural grasslands as population sources for pollinating insects in agricultural landscapes. Journal of Applied Ecology 44: 50–59.

Oesterheld M, Loreti J, Semmartin M, Paruelo JM. 1999. Grazing, fire, and climate effects on primary productivity of grasslands and savannas. Pages 287–306 in Walker LR, ed. Ecosystems of Disturbed Ground. Elsevier.

Pedlar JH, Pearce JL, Venier LA, McKenney DW. 2002. Coarse woody debris in relation to disturbance and forest type in boreal Canada. Forest Ecology and Management 158: 189–194.

Perlack RD, Wright LL, Turhollow AF, Graham RL, Stokes BJ, Erbach DC. 2005. Biomass as feedstock for a bioenergy and bioproducts industry: The technical feasibility of a billion-ton annual supply. Joint study sponsored by the US Department of Energy and US Department of Agriculture. (27 July 2009; *www1.eere.energy.gov/biomass/.../final_billionton_vision_report2.pdf*)

Pikul JL, Hammack L, Riedell WE. 2005. Corn yield, nitrogen use, and corn rootworm infestation of rotations in the northern corn. Agronomy Journal 97: 854–863.

Raghu S, Anderson RC, Daehler CC, Davis AS, Wiedenmann RN, Simberloff D, Mack RN. 2006. Adding biofuels to the invasive species fire? Science 313: 1742.

Randall GW, Huggins DR, Russelle MP, Fuchs DJ, Nelson WW, Anderson JL. 1997. Nitrate losses through subsurface tile drainage in Conservation Reserve Program, alfalfa, and row crop systems. Journal of Environmental Quality 26: 1240–1247.

Reynolds RR. 2005. The Conservation Reserve Program and duck production in the United States prairie pothole region. Pages 144–148 in Allen AW, Vandever MW, eds. The Conservation Reserve Program—Planting for the Future: Proceedings of a National Conference, Fort Collins, Colorado, June 6–9, 2004. US Geological Survey, Biological Resources Discipline, Scientific Investigations Report 2005-5145.

[RFA] Renewable Fuels Association. 2008. Biorefinery Locations. (10 June 2009; *www.ethanolrfa.org/industry/locations/*)

Riffell S, Scognamillo D, Burger LW. 2008. Effects of the Conservation Reserve Program on northern bobwhite and grassland birds. Environmental Monitoring and Assessment 146: 309–323.

Risser PG, Birney EC, Blocker HD, May SW, Parton WJ, Wiens JA. 1981. The True Prairie Ecosystem. Hutchinson Ross.

Roberts MG, Male TD, Toombs TP. 2007. Potential impacts of biofuels expansion on natural resources; a case study of the Ogallala Aquifer region. Environmental Defense. (27 July 2009; *www.heartland.org/custom/semod_policybot/pdf/22233.pdf*)

Roth AM, Sample DW, Ribic CA, Paine L, Undersander DJ, Bartelt GA. 2005. Grassland bird response to harvesting switchgrass as a biomass energy crop. Biomass and Bioenergy 28: 490–498.

Sample DW. 1989. Grassland birds in southern Wisconsin: Habitat preference, population trends, and response to land use changes. Master's thesis. University of Wisconsin, Madison.

Sample DW, Mossman MJ. 1997. Management Habitat for Grassland Birds: A Guide for Wisconsin. Wisconsin Department of Natural Resources.

Schmer MR, Vogel KP, Mitchell RB, Perrin RK. 2008. Net energy of cellulosic ethanol from switchgrass. Proceedings of the National Academy of Sciences 105: 464–469.

Searchinger T, Heimlich R, Houghton RA, Dong FX, Elobeid A, Fabiosa J, Tokgoz S, Hayes D, Yu TH. 2008. Use of US croplands for biofuels increases greenhouse gases through emissions from land-use change. Science 319: 1238–1240.

Secchi S, Babcock BA. 2007. Impact of high corn prices on Conservation Reserve Program acreage. Iowa Ag Review (Spring): 13.

Sheehan J, Dunahay T, Benemann J, Roessler P. 1998. A Look Back at the U.S. Department of Energy's Aquatic Species Program—Biodiesel from Algae. National Renewable Energy Laboratory.

Sissine F. 2007. Energy Independence and Security Act of 2007: A Summary of Major Provisions. Congressional Research Service. CRS report no. RL34294. (10 June 2009; *http://energy.senate.gov/public/_files/RL342941.pdf*)

Stephens SE, Walker JA, Blunck DR, Jayaraman A, Naugle DE, Ringelman JK, Smith AJ. 2008. Predicting risk of habitat conversion in native temperate grasslands. Conservation Biology 22: 1320–1330.

Tiffany DG, Jordan B, Dietrich E, Vargo-Daggett B. 2006. Energy and chemicals from native grasses: Production, transportation and processing technologies considered in the Northern Great Plains. Department of Applied Economics, University of Minnesota.

Tilman D, Hill J, Lehman C. 2006. Carbon-negative biofuels from low-input high-diversity grassland biomass. Science 314: 1598–1600.

[USDA] US Department of Agriculture. 2004. Crop Production 2003 Summary. USDA.

———. 2007. Conservation Reserve Program Monthly Summary October 2007. USDA.

———. 2009. USDA Long-Term Agricultural Projection Tables. (10 June 2009; *http://usda.mannlib.cornell.edu/MannUsda/viewStaticPage.do?url=http://usda.mannlib.cornell.edu/usda/ers/94005/./2009/index.html*)

[USFSA] US Farm Service Agency. 2008. 2-CRP (Revision 4) Handbook. USFSA.

[USFWS] US Fish and Wildlife Service. 1980. Habitat Evaluation Procedures (HEP): USDI Fish and Wildlife Service. USFWS Division of Ecological Services.

———. 1981. Standards for the development of habitat suitability index models for use in the habitat evaluation procedures: USDI Fish and Wildlife Service. USFWS Division of Ecological Services.

[USGS] US Geological Survey. 2008. Habitat Suitability Index Models Series. (10 June 2009; *www.nwrc.usgs.gov/wdb/pub/hsi/hsiintro.htm*)

Whittingham MJ, Devereux CL, Evans AD, Bradbury RB. 2006. Altering perceived predation risk and food availability: Management prescriptions to benefit farmland birds on stubble fields. Journal of Applied Ecology 43: 640–650.

Wilhelm WW, Wortmann CS. 2004. Tillage and rotation interactions for corn and soybean grain yield as affected by precipitation and air temperature. Agronomy Journal 96: 425–432.

Wilhelm WW, Johnson JME, Karlen DL, Lightle DT. 2007. Corn stover to sustain soil organic carbon further constrains biomass supply. Agronomy Journal 99: 1665–1667.

Wilhere GF. 2002. Adaptive management in habitat conservation plans. Conservation Biology 16: 20–29.

Winter M, Johnson DH, Shaffer JA. 2006. Does body size affect a bird's sensitivity to patch size and landscape structure? The Condor 108: 808–816.

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