# Measuring and mitigating agricultural greenhouse gas production in the US Great Plains, 1870–2000

William J. Parton[a,1], Myron P. Gutmann[b], Emily R. Merchant[c], Melannie D. Hartman[a], Paul R. Adler[d], Frederick M. McNeal[d], and Susan M. Lutz[a]

[a]Natural Resource Ecology Laboratory, Colorado State University, Fort Collins, CO 80523; [b]Institute of Behavioral Science and Department of History, University of Colorado, Boulder, CO 80309; [c]Institute for Social Research and Department of History, University of Michigan, Ann Arbor, MI 48106; and [d]US Department of Agriculture, Agricultural Research Service, University Park, PA 16802

Edited by Christopher B. Field, Carnegie Institution of Washington, Stanford, CA, and approved June 15, 2015 (received for review August 27, 2014)

**The Great Plains region of the United States is an agricultural production center for the global market and, as such, an important source of greenhouse gas (GHG) emissions. This article uses historical agricultural census data and ecosystem models to estimate the magnitude of annual GHG fluxes from all agricultural sources (e.g., cropping, livestock raising, irrigation, fertilizer production, tractor use) in the Great Plains from 1870 to 2000. Here, we show that carbon (C) released during the plow-out of native grasslands was the largest source of GHG emissions before 1930, whereas livestock production, direct energy use, and soil nitrous oxide emissions are currently the largest sources. Climatic factors mediate these emissions, with cool and wet weather promoting C sequestration and hot and dry weather increasing GHG release. This analysis demonstrates the long-term ecosystem consequences of both historical and current agricultural activities, but also indicates that adoption of available alternative management practices could substantially mitigate agricultural GHG fluxes, ranging from a 34% reduction with a 25% adoption rate to as much as complete elimination with possible net sequestration of C when a greater proportion of farmers adopt new agricultural practices.**

modeling | Great Plains | biogeochemistry | greenhouse gases | agricultural management

As the extent and role of greenhouse gases (GHGs) in climate change increase in importance, the need for more refined estimates has grown. In addition, improved estimates of specific economic sectors, with better temporal and spatial detail, have become critical. This need is especially acute for agriculture because it is dispersed spatially throughout the world and has contributed GHGs over a long period, although with variation through time, and also across space. In this paper, we describe GHG fluxes from all types of agriculture in the US Great Plains from 1870 to 2000, based on innovative estimates made at the county level for cropping, use of inputs and equipment, and livestock production. The results show two long-term trends. The trend for cropping [measuring changes in carbon (C), nitrogen (N) release, and methane ($CH_4$) uptake] shows that GHG release, measured as gigagrams (Gg) of carbon dioxide ($CO_2$)-carbon equivalent (Ce), peaked in the 1930s and has substantially declined since. The trend for livestock production, agricultural inputs, and equipment use reveals long-term growth in GHG fluxes, reflecting changes in the size and structure of livestock herds in the Great Plains, plus an increased use of synthetic fertilizers, irrigation, and tractors. In the context of US GHG production as a whole (as estimated by the annual GHG inventory of the US Environmental Protection Agency) (1), we estimate that the Great Plains contributes less than 5% of US agricultural GHGs. Nonetheless, this analysis provides important information about the relationship between agricultural practices, the environmental setting of those practices, and the GHG consequences of agriculture in a large region, which is useful for consumers, producers, and policy makers alike.

The US Great Plains is a globally important agricultural region, providing both the US and world economies with grain, fiber, and meat. This region contains more than 30% of the US agricultural land area, and accounts for more than 50% of winter wheat and more than 30% of beef production in the country. The Great Plains region is located in the central United States, generally west of the 98th meridian and east of the Rocky Mountains (Fig. 1). The semiarid temperate climate of this region encompasses enormous subregional diversity, reflected in mean annual temperatures ranging from more than 20 °C in Texas to less than 0 °C in North Dakota, with annual precipitation levels ranging from 700 to 200 mm along an east-west gradient. The native vegetation is primarily grassland, with mixed-grass prairie in the east and shortgrass steppe in the west (3, 4).

Agricultural production in the Great Plains increased dramatically between 1870 and 2000, facilitated by changes in farming techniques, all of which have led to significant GHG releases. Historically, soil cultivation has represented the largest agricultural source of GHGs, producing both C (30–50% soil C losses) and nitrous oxide ($N_2O$) (5). N mineralization rates, enhanced by cropping, increase the $N_2O$ released from the soil, with the application of N fertilizer causing further $N_2O$ emissions (6). The other major sources of GHGs include $CH_4$ from cattle and other livestock production (7); fossil fuels used in fertilizer

## Significance

**The US Great Plains is an agricultural production center for the global market and a source of greenhouse gas (GHG) emissions. This article uses historical data and ecosystem models to estimate the magnitude of annual GHG fluxes from all agricultural sources (cropping, livestock, irrigation, fertilizer production, and tractor use) from 1870 to 2000. Carbon (C) emissions from plow-out of native grasslands peaked in the 1930s and were the largest agricultural source of GHG emissions at the time. Soil C emissions subsequently declined, whereas GHG fluxes from other activities increased. The results inform knowledge about the relationship between agriculture and its environmental setting and show that available alternative management practices could substantially mitigate the environmental consequences of agricultural activities without reducing food production.**

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

The authors declare no conflict of interest.

This article is a PNAS Direct Submission.

Data deposition: The data reported in this paper have been deposited at the Inter-university Consortium for Political and Social Research (ICPSR) at the University of Michigan. ICPSR 31681: Great Plains Population and the Environment Data: Biogeochemical Modeling Data 1860–2003.

[1]To whom correspondence should be addressed. Email: william.parton@colostate.edu.

This article contains supporting information online at www.pnas.org/lookup/suppl/doi:10.1073/pnas.1416499112/-/DCSupplemental.

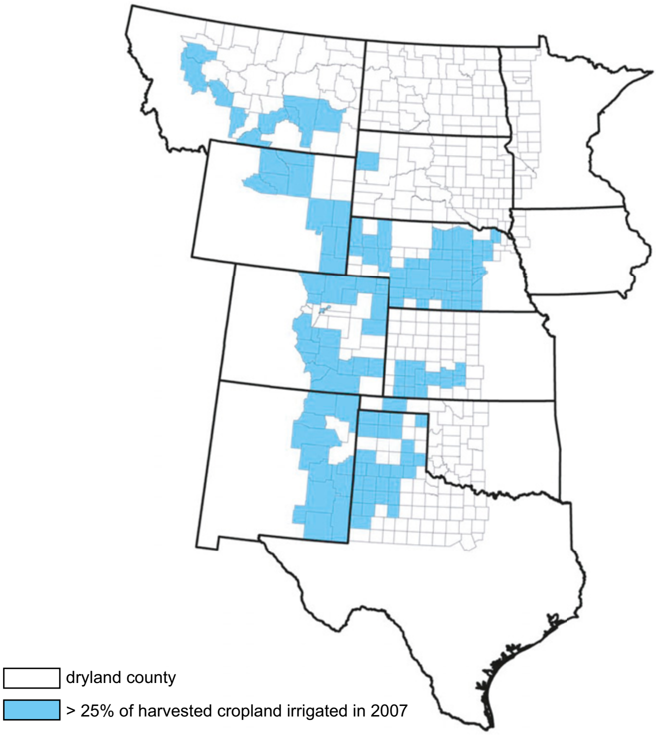


**Fig. 1.** Irrigation in the US Great Plains. Counties shaded in light blue irrigated at least 25% of cropland area in 2007; in the unshaded counties, dryland cropping accounts for 75% or more of cropland (2).

production; and C released from burning of fossil fuels by tractors, irrigation pumps, and other farm equipment.

Agricultural GHG production in the Great Plains is also tightly tied to public policy actions, primarily those of the US government. The plow-out of the plains would not have been as rapid without the Homestead Act of 1862, which distributed land at no cost to those homesteaders willing to cultivate it. In addition, the 20th century reduction in the extent of cropland (described later) was spurred by agricultural support policies that arose out of the economic and climatic events of the 1930s and are still in existence, in various forms, today.

Previous studies of the GHG fluxes associated with agriculture in North America and Europe have used field measurements (8, 9), inventory methods (10), economic land use models (e.g., the "Forest and Agricultural Sector Optimization Model with Greenhouse Gases") (11, 12) or ecosystem models (e.g., the Daycent model used here) (13, 14) to estimate GHG sources or sinks at either the present time or at specific points in the past or future. Although some methods estimate future GHG fluxes under projected scenarios (14, 15) and others compare the GHG production of current and past agricultural production systems (10, 13), no previous study has estimated cumulative GHG emissions from all agricultural sources from initial plow-out to the present.

The historical and complete perspective offered by this article provides important insights into the environmental impact of agriculture. First, the magnitude of GHG emissions is a result not just of land use practices but also of land use change, which is not captured by synchronic analysis (8). Second, the soil GHG flux at any point in time depends on the previous land use history (8, 12, 16, 17). Third, some practices that enhance soil C sequestration, such as fertilizer application and irrigation, increase GHG emissions from $N_2O$ release and fossil fuel burning (9, 18). This paper estimates GHG fluxes from "on-farm" activities, and also includes the energy used for irrigation pumping, tractors, and the production of synthetic fertilizer. The Daycent ecosystem model was used to estimate soil GHG fluxes associated with cropping in each of 476 Great Plains counties (19). A complete description of the procedure used to estimate all of the GHG fluxes is provided in *SI Materials and Methods*.

## Great Plains Agricultural Land Use History

Land use in the Great Plains region has changed dramatically over the past 150 y, and can be divided into three sequential regimes (Fig. 2). Grazing dominated during the first regime, with cattle succeeding bison after the US Civil War. The second regime began in the 1870s, with crop-based agriculture introduced first in the eastern plains and then moving westward through the 1930s. The third regime dates from the mid-1930s and is characterized by the rapid adoption of such new agricultural technologies as (*i*) mechanical cultivation equipment, (*ii*) inorganic fertilizers, (*iii*) pump irrigation, and (*iv*) new high-yielding crop varieties bred for synergy with other new inputs (22–24). This period has also seen a substantial increase in beef cattle production in the Great Plains, both in absolute numbers and as a proportion of all livestock, especially from the 1940s to the 1960s (Fig. 2*D*).

The dominant crops in the northern and central Great Plains are winter wheat, spring wheat, and corn. Winter wheat, sorghum, corn, and cotton are the most prevalent crops in the south (23). Dryland cropping predominates, except along the major river systems and over the Ogallala aquifer, where irrigation water is available. The amount of irrigated agricultural land increased substantially in the 1950s, and then stabilized by the 1980s, with over 15 million acres irrigated with pumped water (Fig. 2*A*). Land was increasingly retired from crop production beginning in the 1950s, largely through enrollment in federally sponsored conservation programs. An analysis of the most recent US Census of Agriculture (through 2012) suggests that about 20% of the 1950s dryland cropping area is no longer used for that purpose, with about half of that change a result of increased irrigated cropping and about half a result of a long-term reduction in cropland. The data also show a 31% decrease in land in the Conservation Reserve Program (CRP) between its maximum in 2007 and 2012. We suspect that improved corn prices have led farmers to convert CRP land into cropland. The consequence of plowing out CRP land would be to release C from the soil and increase $N_2O$ emissions (25).

Total production for all crops was low before 1940. Production rose thereafter as farmers experienced rapid increases in corn, hay, cotton, and wheat yields from the 1940s to the 1970s; corn yields have continued to grow since then (Fig. 2*B*). Parton et al. (6) attribute the larger crop yields to the increased use of fertilizer (Fig. 2*C*), new crop varieties (22), improved soil tillage practices for wheat, use of herbicides and insecticides, and the expansion of irrigation, all of which generates a yield increment of 100–300% in cotton, hay, and corn production.

Patterns of livestock raising have also changed over the past 150 y. The number of domestic animals rapidly increased from 1870 to 1900 (Fig. 2*D*), by which time more than 12 million beef cattle had replaced the bison herds on the Great Plains. After a period of stability from 1900 to 1935, the number of beef cattle increased by 120% from 1935 to 1970, whereas the number of horses and dairy cattle declined due to the substitution of mechanical power for horse power in agriculture and the concentration of dairies in other parts of the country. Many of these cattle reside in feedlots, a major consumer of the increasing quantities of corn and hay produced under irrigation.

Although many of the changes in agricultural practices just described were accomplished by the 1970s, change has continued, and efforts to improve agricultural productivity and efficiency are reflected in the results reported here. The previously unidentified estimates stated in this article were made using the

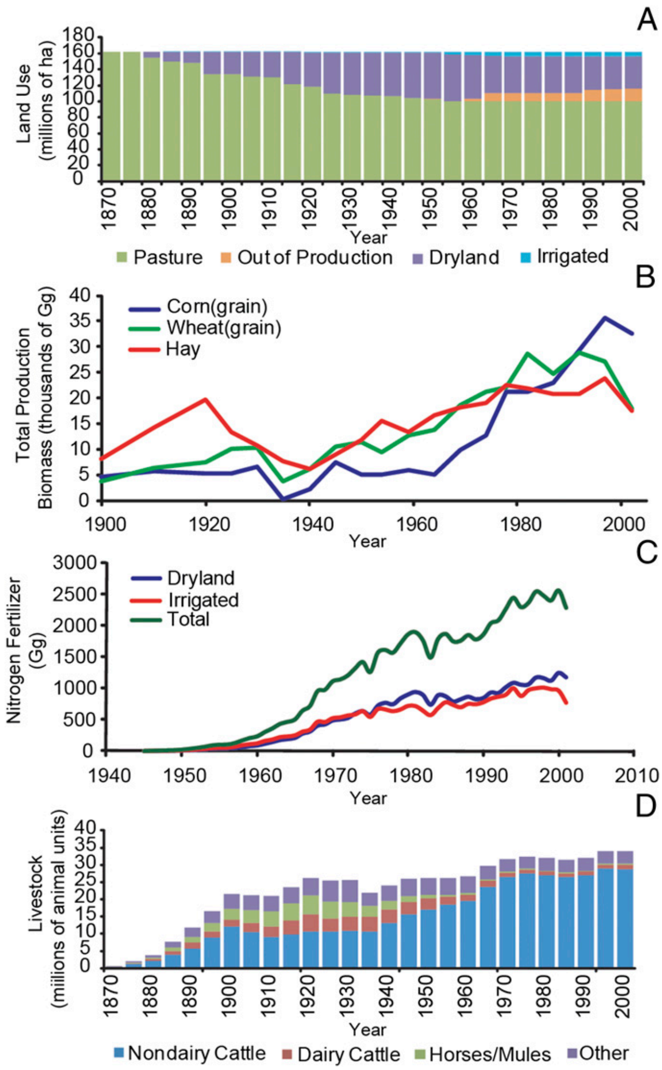


**Fig. 2.** Changing agricultural practices in the US Great Plains, 1870–2000. (*A*) Transition from native pasture (Pasture) to dryland cropland (Dryland) in the first half of the 20th century, followed by an increase in irrigated cropland (Irrigated) and the restoration of previously cropped land (Out of Production) (20). (*B*) Total plant production of three major crops (corn, wheat, and hay) (6). (*C*) Amount of N fertilizer applied to dryland cropland (Dryland), the amount applied to irrigated cropland (Irrigated), and the total of the two (Total) (6). (*D*) Livestock populations expressed as animal units (nondairy cattle = 1, dairy cattle = 1.4, horse = 1, mule = 1, pig = 0.4, sheep = 0.1, chicken = 0.033) (20, 21).

best available data and methods. As farmers and scientists look to the future, many emerging agricultural best practices show promise for continued productivity coupled with reduced environmental consequences. The implementation of these best-management practices offers the potential for reducing GHG fluxes from Great Plains agricultural systems. Examples of these practices are well known and well documented, and include (*i*) use of slow-release N fertilizer and nitrification inhibitors with the potential to reduce soil $N_2O$ fluxes by 14–58% for irrigated and dryland agricultural systems (26), (*ii*) use of no-tillage cultivation practices that can produce a net C storage of 7.0–20.0 g of C per $m^{-2}\cdot y^{-1}$ for dryland systems (27, 28) and 15.0–83.9 g of C per $m^{-2}\cdot y^{-1}$ for irrigated systems (29, 30), and (*iii*) use of new inoculation techniques for cattle that can result in a 20–40% reduction in $CH_4$ production (7). The adoption of no-tillage farm practices would also reduce the amount of energy consumed by farm equipment and increase both soil water storage and yields in dryland wheat systems. Currently, no-tillage cultivation is used by fewer than 15% of wheat farmers (the dominant crop) in the Great Plains (31). Recent papers suggest the previous estimates of soil C storage under no-till agriculture have been overestimated (32, 33) because most of the soil C measurements did not consider the changes to soil C below the plow zone (0- to 20-cm layer). In contrast to that finding, a recent review paper supports the results from earlier studies that show substantial increases in soil C under no-tillage cultivation for dry agricultural regions similar to the Great Plains (34, 35).

## Results

**GHG Fluxes by Land Use Category.** Estimates of GHG fluxes related to cropping are combined from separate analyses of four agricultural land use systems (dryland cropping, irrigated cropping, land removed from crop production, and pasture land). Fig. 3 shows GHG emissions annually by land use category, with separate results for system C [positive when lost (released) and negative when sequestered (stored)], soil $N_2O$ emissions (always positive), and $CH_4$ absorbed by the soil (always negative). The GHG fluxes for $N_2O$ and $CH_4$ are shown as $CO_2$-Ce fluxes using standard corrections needed to account for the differential GHG warming potentials of $N_2O$ and $CH_4$ ($N_2O$ is 298-fold more effective than $CO_2$, and $CH_4$ is 25-fold more effective than $CO_2$). System C includes soil C (>95% of total C) and live and dead plant C pools, with most of the changes in system C resulting from changes in soil C. These results are weighted to reflect the actual amount of land in each use category across the region. Because a relatively small amount of land was used for irrigated cropping or taken out of production (Fig. 3), the scale for Fig. 3 *B* and *C* is an order of magnitude smaller than the scale for the panels representing dryland cropping (Fig. 3*A*) and pasture (never-cropped) land (Fig. 3*D*).

Simulated soil GHG fluxes from dryland cropping in the Great Plains (Fig. 3*A*) show a steady increase in system C loss (primarily from the soil C pool) and $N_2O$ fluxes from 1900 to 1930, a result of the progressive plow-out of native prairie grasslands that culminated in most parts of the region during the 1930s. System C losses slowed after the 1930s, partly as a result of the stabilization of the amount of land in crop production and partly because the loss of soil C from cultivation markedly diminished after the first 20 y of cultivation (36). A general pattern of low-level system C sequestration began in the 1960s, resulting from a reduction in the number and intensity of soil tillage events. At the same time, however, soil $N_2O$ fluxes increased by as much as 100% due to an increase in the use of N fertilizer (Fig. 2*C*).

Model results for irrigated cropping systems in the Great Plains (Fig. 3*B*) show low levels of system C loss before the 1950s because relatively few counties irrigated before that time. Irrigated agriculture was initiated after 1950 in most areas where land had already been plowed for dryland cultivation, and thereby depleted of soil C. As irrigation became more widespread on former dryland cropping areas between the 1950s and the 1970s, both system C storage (removing GHGs from the atmosphere) and soil $N_2O$ fluxes (releasing GHGs into the atmosphere) increased dramatically (6). These increases reflect the enhancement of plant production and C inputs to the soil associated with irrigation (6, 37), as well as the expansion of irrigated land (Fig. 2*A*). System C storage peaked in the 1970s and then decreased as the soil C storage potential of the irrigated soils was reached. Soil $N_2O$ fluxes increased in tandem with N fertilizer application beginning in the 1950s (Fig. 2*C*). Irrigated cropping represents a net sink of GHGs from 1950 through the 1980s, with fluxes reaching their lowest negative values in the 1970s and then turning positive by the 1990s as the soil saturated with C and $N_2O$ fluxes continued to increase.

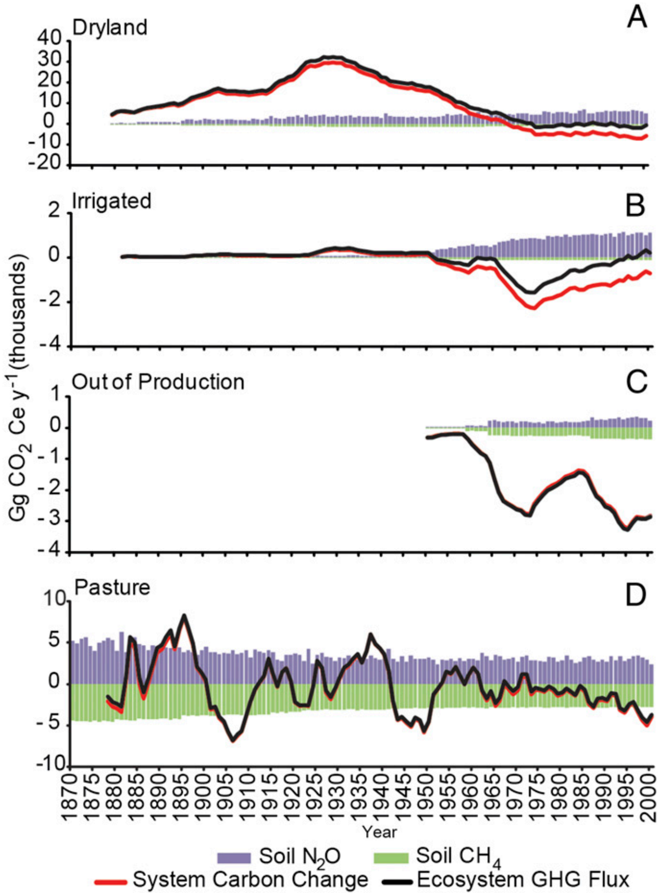


**Fig. 3.** Estimates of GHG fluxes by land use category showing soil $N_2O$ emissions (always positive) and $CH_4$ consumption (always negative), and the change in system C since the previous year (either positive or negative), expressed in gigagrams of $CO_2$-Ce (Gg $CO_2$-Ce $y^{-1}$): dryland cropland (*A*), irrigated cropland (*B*), dryland cropland restored to grassland (*C*), and native pasture (*D*). Positive values indicate GHG sources into the atmosphere, whereas negative values indicate a terrestrial sink. The solid black line represents the 9-y moving average of the total GHG flux (sum of soil $N_2O$, $CH_4$, and system C change). The red solid line represents the 9-y moving average of the change in total system C.

Simulations of the abandonment of dryland cropping (out of production; Fig. 3*C*) show a dramatic increase in system C storage, resulting from the accumulation of soil C with the cessation of cropping and tillage (38–40). Increases in C storage over time reflect both the restoration of grassland (perennial grasses have more root growth than annual crops) and increases in the amount of land taken out of production, particularly during the 1960s and then again with the implementation of the CRP in the 1980s (41). Restoration of native soil fertility takes 100 y or more unless fertilizer is added to the system. However, even the slow accumulation of C more than offsets soil $N_2O$ fluxes, which remain low in the restored grassland because of low soil fertility. Net GHG fluxes in this system are therefore negative throughout the period.

Modeled GHG fluxes for pasture (Fig. 3*D*) are neutral on the whole, with positive GHG emissions from soil $N_2O$ balanced by negative fluxes from $CH_4$ uptake by the soil. System C fluxes vary substantially from year to year with precipitation, showing a general pattern of net C losses during dry years and net C gains during wet years. Growing season precipitation is negatively correlated with growing season maximum air temperature (42, 43). System C storage is positively correlated with growing season precipitation and negatively correlated with growing season maximum air temperature; thus, pasture system C levels increase (C sequestration – negative fluxes) during cold and wet summers and decrease (C release – positive fluxes) during hot and dry summers. This pattern is clearly demonstrated by C losses during the dry 1930s and C uptake during the wet 1990s. Details can be found in *SI Materials and Methods*.

**All Sources of GHG Emissions.** Fig. 4*A* summarizes the results from Fig. 3. As cropping expanded from the late 19th century into the 20th century, an ever-larger quantity of GHG emissions was released into the atmosphere, peaking in the 1930s at levels as high as 35,000 Gg of $CO_2$-Ce per year. The most prominent source of GHG emissions during this period was the loss of soil C associated with the plowing of native grasslands, a practice that had nearly ended by the early 1930s. Soil C losses diminished rapidly from 1930 into the 1950s, with soil C sequestration starting after the 1960s, as a result of both soil C stabilization in dryland systems and soil C storage in irrigated cropland and restored grasslands (cropland out of production). The cultivation of native grasslands also produced a pattern of increasing soil $N_2O$ fluxes beginning in 1900. $N_2O$ emissions rose by another 30% between 1940 and 2000 due to increased fertilizer application in irrigated and dryland cropping systems. After 1970, C sequestration in restored grasslands and irrigated cropland offset $N_2O$ emissions and produced net negative GHG fluxes in the system as a whole. This pattern of GHG sequestration was enhanced in the 1990s when above-average rainfall increased C storage in native prairie and restored grasslands.

Since the 1930s, farmers have increased plant productivity by (*i*) using gasoline and diesel equipment to cultivate, plant, and harvest their lands; (*ii*) applying synthetic fertilizer to raise crop productivity; and (*iii*) irrigating fields to ensure sufficient moisture for plant growth. GHG fluxes from tractor fuel peaked in the 1950s and then declined, with most energy used throughout the period for cultivation rather than planting or harvesting (Fig. 4*B*). The decline since the 1950s is largely due to less intensive cultivation and a shift from gasoline to more energy-efficient diesel engines. The scale of GHG fluxes from equipment is relatively small, with the peak just over 3,500 Gg of $CO_2$-Ce per year, compared with peaks from the soil systems 10-fold as high in the 1930s (Fig. 4*A*). GHG fluxes from energy used for fertilizer production and irrigation pumping increased steadily after onset of use in the 1940s (Fig. 4*B*). Fluxes from irrigation pumping have remained constant since the 1970s, as the amount of irrigated land has stabilized and pumps have shifted from gasoline to electric power. At the same time, GHG emissions associated with fertilizer production increased rapidly from virtually nothing in the 1930s and 1940s to a peak in the late 1960s of nearly 4,700 Gg of $CO_2$-Ce per year, followed by a slow decline to about 3,500 Gg of $CO_2$-Ce per year in the late 1990s.

The last element in our comprehensive estimate of GHG emissions from Great Plains agriculture is the $CH_4$ produced by the livestock sector. Overall, livestock numbers increased slowly but steadily in the area during the 20th century (Fig. 2*D*). The use of horses for maintaining agricultural lands decreased as tractors were adopted. In addition, the number of dairy cattle decreased as other regions of the United States began to specialize in dairy production. Nondairy cattle were quickly substituted, becoming consumers of the region's increasing corn production. As generators of enteric and manure $CH_4$, livestock are large and growing contributors to regional GHG fluxes (Fig. 4*C*).

Fig. 4*D* combines all of the elements discussed above into previously unidentified estimates of annual net GHG emissions from US Great Plains agriculture between 1870 and 2000 (presented numerically in Table 1). Early exploitation of the soil by agriculture (from the late 19th century until the 1930s) produced

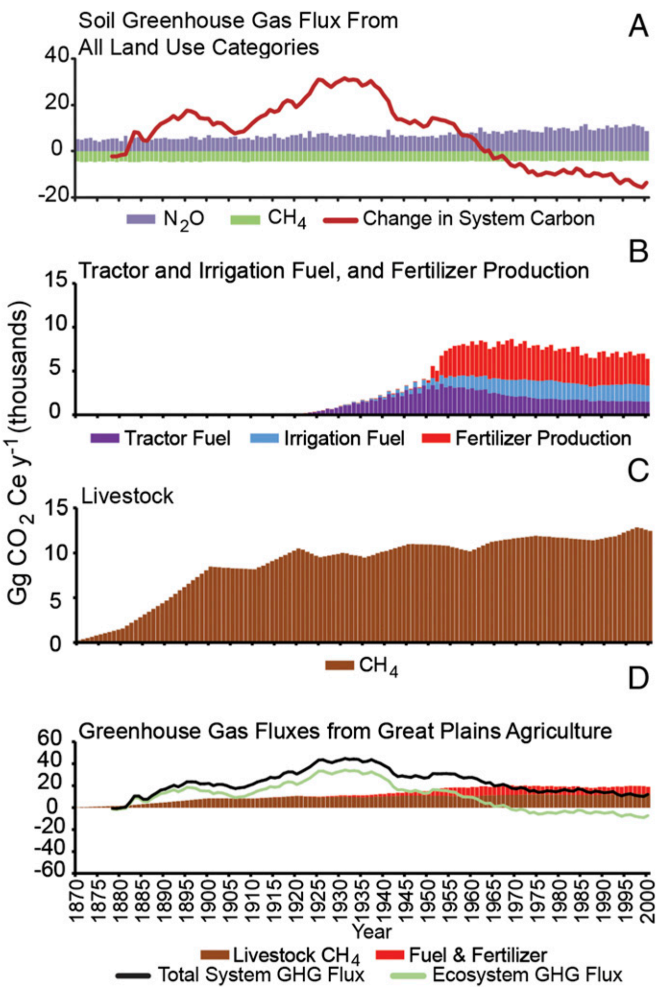


**Fig. 4.** GHG fluxes from 1870 to 2000 (Gg $CO_2$-Ce $y^{-1}$). (*A*) Total emissions from the four land use categories in Fig. 3 (Dryland Cropland, Irrigated Cropland, Out of Production Land, and Native Pasture). (*B*) Tractor fuel use for planting, harvesting, and cultivation (Tractor Fuel); fuel use for irrigation pumping (Irrigation Fuel); and GHG emissions from fertilizer production (Fertilizer Production). (*C*) Livestock enteric $CH_4$ and manure $CH_4$ emissions. (*D*) GHG fluxes from all agricultural sources, including livestock enteric $CH_4$ and manure $CH_4$ emissions (Livestock $CH_4$, brown bars); fuel used for tractors, irrigation pumping, and fertilizer production (Fuel & Fertilizer, pink bars); soil $N_2O$, $CH_4$, and system C change from the four land use categories [Ecosystem GHG Flux, solid light green line], representing the 9-y moving average of the change in the soil/plant system GHG fluxes (sum of soil $N_2O$, $CH_4$, and system C change); and the sum of all sources (Total System GHG Flux, solid black line), representing the 9-y moving average of the total GHG flux from soil/plant, tractor fuel, irrigation pumping, fertilizer manufacture, and livestock production.

a net positive GHG flux, peaking in the particularly dry 1930s at an average rate of 45,000 Gg of $CO_2$-Ce per year. After that time, contributions from soil systems diminished, whereas contributions from livestock increased, reaching 12,000 Gg of $CO_2$-Ce per year in the 1990s, due to extensive use of the region for cattle feeding. By the 1950s, fuel consumed for irrigation, mechanized cultivation, and fertilizer production had also become an important source of net GHG emissions, at about 7,000 Gg of $CO_2$-Ce per year by the end of the century. Total GHG fluxes have decreased since the period of soil plow-out (they are now close to 12,000 Gg of $CO_2$-Ce per year) but are nonetheless steady and positive.

The uncertainty in these results is presented in *SI Materials and Methods*, which shows that the absolute value of the uncertainty generally increases from 1860 to 2003, with secondary uncertainty maxima from 1915 to 1935. Absolute uncertainty ranges from 239 Gg of $CO_2$-Ce per year in 1871 to 877 Gg of $CO_2$-Ce per year in 1975. Increasing absolute uncertainty after the 1950s is correlated with increases in livestock $CH_4$; soil $N_2O$ emissions; and fuel consumed for irrigation, cultivation, and fertilizer production. The high uncertainty in GHG fluxes from 1915 to 1935 is associated with the high soil C losses resulting from cultivation of the soil and expansion of agriculture. The relative annual uncertainty of total GHG fluxes (annual absolute uncertainty divided by observed annual net GHG flux) ranges from 1 to 206%, with the highest relative uncertainty associated with low net GHG fluxes (values close to zero).

**GHG Reductions Using Best-Management Practices.** The combined impact of best-management practices for Great Plains farming in the 1990s (current GHG fluxes from farming) was determined by (*i*) assuming that no-tillage practices would lead to an additional storage of 50 g of C per $m^{-2} \cdot y^{-1}$ in irrigated cropping (29, 30) and an additional storage of 10.0 g of C per $m^{-2} \cdot y^{-1}$ in dryland cropping (27, 28), (*ii*) a 30% reduction in soil $N_2O$ fluxes by using improved fertilizer techniques (26), and (*iii*) a 30% reduction in $CH_4$ (7) due to improved cattle management (the method of estimation is provided in *SI Materials and Methods*). Due to the uncertainty of farmer adoption rates, we made our assessments assuming a range of 25%, 50%, 75%, and 100% (44). Results (Table 2) show a 34% reduction in net GHG fluxes, with 25% of farmers adopting the new methods. Most of the reduction (60%) resulted from the use of no-tillage cultivation practices, 25% resulted from the $CH_4$ mitigation from cattle, and 15% resulted from the use of improved fertilizer. A 75% increase in farmer adoption rates resulted in a 102% reduction, whereas a 100% rate increase resulted in a 136% reduction (C sequestration).

## Discussion

The results of this research reveal a complex picture, which is neither optimistic nor pessimistic, regarding agriculture's contribution to net GHG emissions, especially if we look to the future. If we begin with the historical account of estimated emissions from land management (Fig. 4*A*), the conclusion may be encouraging, suggesting that after a century of soil exploitation through cropping, the agricultural systems of the Great Plains had begun to stabilize by the 1970s, with relatively modest emissions of GHGs and the potential for C sequestration to offset soil $N_2O$ emissions. However, Table 1 shows that conclusion to be inappropriate. Even if cropping systems have sequestered some C since the 1960s, all other parts of the agricultural system (including those parts that facilitated C sequestration) continue to produce net positive GHG fluxes, with the largest contributions coming from livestock production and smaller, yet nontrivial, amounts coming from equipment use, fertilizer, and irrigation. Although there is uncertainty in these estimates, that uncertainty is small compared with the historical changes in GHG fluxes over 130 y. The historical patterns described here show the important roles played by the series of technological transformations that have swept over agriculture since 1870. The patterns themselves partly reflect public policies that have altered where and how these lands have been used for agriculture. Within this human-dominated system, however, it is critical to notice the fundamental influence of climatic variation, which is apparent in Daycent model results for never-cropped land (pasture; Fig. 3*D*). Independent of technological change and policy impacts, when precipitation was high and temperature was low, C was stored. Conversely, when temperature was high and precipitation was low, C was released. Interannual changes in system C on pasture land ranged from a storage of 41 g of C per square meter in the wet year of 1942 to a release of 27 g of C

**Table 1. Net GHG fluxes from all sources (gigagrams of $CO_2$-Ce): Annual average by decade**

| Decade | Pasture | Dryland | Irrigated | Out of production | Tractor | Irrigation | Fertilizer | Livestock | Total |
|---|---|---|---|---|---|---|---|---|---|
| 1870 | −1,491 | 441 | 0 | 0 | 0 | 0 | 0 | 824 | −225 |
| 1880 | 4,121 | 8,266 | 28 | 0 | 0 | 0 | 0 | 2,967 | 15,382 |
| 1890 | 1,624 | 13,328 | 94 | 0 | 0 | 0 | 0 | 6,378 | 21,423 |
| 1900 | −4,154 | 15,597 | 109 | 0 | 0 | 0 | 0 | 8,318 | 19,871 |
| 1910 | 1,651 | 22,027 | 83 | 0 | 0 | 0 | 1 | 9,205 | 32,967 |
| 1920 | −1,173 | 31,230 | 392 | 0 | 383 | 0 | 7 | 9,914 | 40,715 |
| 1930 | 4,636 | 25,359 | 194 | 0 | 1,410 | 127 | 18 | 9,811 | 41,398 |
| 1940 | −4,453 | 18,283 | 220 | 0 | 2,503 | 455 | 55 | 10,749 | 27,516 |
| 1950 | 376 | 9,900 | −306 | −352 | 3,162 | 1,101 | 2,527 | 10,645 | 26,630 |
| 1960 | −301 | 2,969 | −872 | −2,457 | 2,571 | 1,659 | 3,960 | 11,153 | 18,341 |
| 1970 | −945 | −937 | −1,017 | −1,910 | 1,944 | 2,018 | 3,868 | 11,800 | 14,475 |
| 1980 | −735 | 1,030 | −392 | −2,200 | 1,697 | 1,811 | 3,660 | 11,553 | 16,108 |
| 1990 | −4,109 | −2,311 | 293 | −2,919 | 1,574 | 1,890 | 3,500 | 12,300 | 9,929 |

per square meter in the dry year of 1934. Table 1 indicates that in the 1990s, fuel (tractor and irrigation), fertilizer, and livestock in the Great Plains produced 19,263 Gg of $CO_2$-Ce of GHG emissions (Fig. 4*D*). Because it was a particularly wet decade, 21% of this release was absorbed by the pasture system, but results for pasture in the dry 1930s suggest that the pasture system could also increase the release of GHG emissions by the same percentage under less favorable weather conditions. Climatic changes, therefore, have the potential to alter yearly net GHG fluxes greatly.

Our results suggest that use of best agricultural management practices has the potential to reduce net GHG fluxes from the Great Plains agricultural system greatly, depending on the rate of adoption by farmers (a 34–136% reduction as rates of adoption increased from 25–100%). Most of the reduction (60%) resulted from the use of no-tillage cultivation practices, 25% resulted from the $CH_4$ mitigation from cattle, and 15% resulted from the use of improved fertilizer. The use of no-tillage cultivation has increased over the past 10 y (44), and has the potential to reduce GHG fluxes for the Great Plains greatly, because only 14% of land planted in wheat, the dominant crop, used no-tillage practices during 2008 (31). The reduction in GHG fluxes from C sequestration due to the use of no-tillage practices primarily occurs during the first 20 y following initiation of the practice; however, the GHG reductions from the use of $CH_4$ mitigation from cattle and improved fertilizer result in permanent annual reductions. These reductions from the Great Plains agricultural system could potentially contribute to the goal of the Obama administration to reduce GHG fluxes from agricultural systems in the United States by more than 25% during the next 10 y (45, 46). The technologies to implement these best-management practices are currently available; however, farmers will not adopt them unless financial incentives are offered, given the resultant cost increase for raising both crops and livestock.

This research has estimated the overall production of positive net GHG fluxes from Great Plains agriculture at about 2.9 million Gg of $CO_2$-Ce between 1870 and 2000. Although this figure is large, almost 50% of the total emissions are a result of the expansion of dryland agriculture before 1950. Clearly, the major sources of GHG fluxes have evolved over time. Before 1950, most emissions came from soil C losses related to cultivation. Currently, the majority of GHG fluxes result from livestock production and energy used by farm equipment, fertilizer synthesis, and irrigation (Table 1). As a result of these findings, this article suggests that the use of existing best-management practices could greatly reduce GHG emissions from US Great Plains agricultural systems if economic incentives were available to promote their use.

## Materials and Methods

The Daycent ecosystem model was used to estimate soil GHG fluxes associated with cropping in each of 476 Great Plains counties (19). The model is driven by county-level weather and soil data (42, 43, 47) and detailed assumptions about daily agricultural management practices. These practices include cultivation, planting, irrigation, fertilizer application, and harvesting over the simulation period, and were derived from historical documents reflecting historical changes in crop varieties, technology, and cropping techniques (37). Each major dryland and irrigated rotation system, as well as unplowed native grassland, was modeled separately, as was land removed from crop production either before or under the CRP. The model was verified and validated with yield data from national agricultural databases (19) (www.nass.usda.gov) and scaled to the county level using historical agricultural census data (20). Daycent output includes interannual change in system C and annual amounts of $N_2O$ release and $CH_4$ absorption. GHG flux is calculated by converting each component to $CO_2$-Ce and summing over components. Calculation of the uncertainty in our estimates of GHG fluxes from each component of agriculture and for the total annual agricultural GHG flux in the Great Plains is detailed in *SI Materials and Methods*.

The assumptions about agricultural management practices that drive the Daycent model also provided estimates of historical amounts of tractor use, irrigation, and fertilizer application in each county over the simulation period. Fuel consumed during tractor use was estimated using the Agricultural Machinery Management Data from the American Society of Agricultural and Biological Engineers Standards (48, 49). Fuel consumed in irrigation pumping was estimated by combining data from the National Agricultural Statistics Service's 2008 Farm and Ranch Irrigation Survey (50) with energy price data from the Energy Information Administration (51–55). Historical GHG emissions

**Table 2. Summary of mitigation best practices**

| Scenario | System before improvements | Reduction from no-till | Improved fertilizer | Improved livestock feed | Total change | System total with improvements |
|---|---|---|---|---|---|---|
| Absolute value (and percent change) due to mitigation strategies | | | | | | |
| Current practices | 102,177 | | | | | 102,177 |
| 25% use best practices | 102,177 (0%) | −20,788 (−20%) | −5,401 (−5%) | −8,604 (−8%) | −34,794 (−34%) | 67,383 |
| 50% use best practices | 102,177 (0%) | −41,576 (−41%) | −10,803 (−11%) | −17,208 (−17%) | −69,587 (−68%) | 32,590 |
| 75% use best practices | 102,177 (0%) | −62,364 (−61%) | −16,204 (−16%) | −25,812 (−25%) | −104,381 (−102%) | −2,204 |
| 100% use best practices | 102,177 (0%) | −83,153 (−81%) | −21,606 (−21%) | −34,416 (−34%) | −139,174 (−136%) | −36,997 |

from the production of N fertilizer were calculated by combining the current global warming intensity from production of commercial N fertilizer products with both the historical changes in the type of N fertilizer products used by farmers (www.aapfco.org/publications.html; refs. 56–58) and changes in the efficiency of producing the different types of N fertilizer products (59). $CH_4$ produced by livestock enteric fermentation and manure management was calculated using Intergovernmental Panel on Climate Change tier 1 emission factors for dairy and nondairy cattle, horses, mules, pigs, sheep, and chickens (60). Regional totals for all sources of GHG flux were derived by aggregating county-level estimates. Full details are available in *SI Materials and Methods*.

At the beginning of our analysis, energy use and GHG fluxes were clearly bounded by the local farm experience. They became less so over time in two ways. First, the energy used in pump-driven irrigation, which was powered by gasoline engines in the 1940s and 1950s and partly converted to diesel later, is now largely powered by electricity. The GHG fluxes involved in electricity generation are now remote from the agricultural enterprise. Second, synthetic fertilizer has been substituted for locally produced manure since the 1950s. We have included both the energy used for irrigation pumping and the energy used to produce synthetic fertilizer in our analysis because to exclude either one at any given point in time would disrupt the long-term aspect of our work, for which irrigation pumping and fertilizer production and use are essential.

Potential reductions in GHG fluxes due to improved agricultural practices were estimated by taking the calculated GHG contributions by agricultural category for the 1990s and reducing them by fixed amounts of C stored for no-tillage cropping practices and by percentages of GHG flux for improved fuel use, fertilizer, and cattle management. Because we do not know the proportion of farmers who might make use of these improved practices, we then estimated the overall and agricultural category impacts on GHG fluxes assuming that 25%, 50%, 75%, and 100% of farmers would do so. A more detailed description of the methods used can be found in *SI Materials and Methods*.

**ACKNOWLEDGMENTS.** This research was supported by Grant R01HD33554 from the Eunice Kennedy Shriver National Institute of Child Health and Human Development; the Shortgrass Steppe Long-Term Ecological Research Site [National Science Foundation (NSF) Grant DEB-0823405] at Colorado State University; US Department of Agriculture (USDA) cooperative research projects (Grants 58-5402-4-001 and 59-1902-4-00); USDA Ultraviolet-B (Grant 2014-34263-22038); USDA National Institute of Food and Agriculture (Grant 2015-67003-23456); and the Inter-university Consortium for Political and Social Research at the Institute for Social Research, University of Michigan. We thank Cindy Keough for her assistance with the Daycent model, Glenn Deane for statistical consultation, and Laurie Richards for editorial assistance. The National Science Foundation supported some of M.P.G.'s work on this article.

1. US EPA (2007) *Inventory of US Greenhouse Gas Emissions and Sinks: 1990–2005* (US Environmental Protection Agency, Washington, DC), Available at www.epa.gov/climatechange/Downloads/ghgemissions/07CR.pdf. Accessed March 6, 2014.
2. US Department of Agriculture (2009) *United States Census of Agriculture 2007, Volume 1, Chapter 2: County Level Data* (USDA, National Agricultural Statistics Service, Washington, DC), Available at www.agcensus.usda.gov/Publications/2007/Full_Report/Volume_1,_Chapter_2_County_Level/. Accessed August 1, 2011.
3. Sala OE, Lauenroth WK, Parton WJ, Trlica MJ (1981) Water status of soil and vegetation compartments in the shortgrass steppe. *Oecologia* 48(3):327–331.
4. Del Grosso S, et al. (2008) Global potential net primary production predicted from vegetation class, precipitation, and temperature. *Ecology* 89(8):2117–2126.
5. Matson PA, Parton WJ, Power AG, Swift MJ (1997) Agricultural intensification and ecosystem properties. *Science* 277(5325):504–509.
6. Parton WJ, Gutmann MP, Ojima D (2007) Long-term trends in population, farm income, and crop production in the Great Plains. *Bioscience* 57(9):737–747.
7. Boadi D, Benchaar C, Chiquette J, Masse D (2004) Mitigation strategies to reduce enteric methane emissions from dairy cows: Update review. *Can J Anim Sci* 84(3):319–335.
8. De Jong, et al. (1999) Land-use change and carbon flux between 1970s and 1990s in Central Highlands of Chiapas, Mexico. *Environ Manage* 23(3):373–385.
9. Flessa H, et al. (2002) Integrated evaluation of greenhouse gas emissions ($CO_2$, $CH_4$, $N_2O$) from two farming systems in southern Germany. *Agric Ecosyst Environ* 91(1-3):175–189.
10. Cavigelli MA, et al. (2012) US agricultural nitrous oxide emissions: Context, status, and trends. *Front Ecol Environ* 10(10):537–546.
11. Beach RH, et al. (2010) *Model Documentation for the Forest and Agricultural Sector Optimization Model with Greenhouse Gases (FASOMGHG). Report for the United States Environmental Protection Agency* (Research Triangle Institute International, Research Triangle Park, NC).
12. McCarl BA, Schneider UA (2001) Climate change. Greenhouse gas mitigation in U.S. agriculture and forestry. *Science* 294(5551):2481–2482.
13. Del Grosso SJ, Mosier AR, Parton WJ, Ojima DS (2005) DAYCENT model analysis of past and contemporary soil $N_2O$ and net greenhouse gas flux for major crops in the USA. *Soil Tillage Research* 83(1):9–24.
14. Tian H, et al. (2012) Contemporary and projected biogenic fluxes of methane and nitrous oxide in North American terrestrial ecosystems. *Front Ecol Environ* 10(10):528–536.
15. Cole CV, et al. (1997) Global estimates of potential mitigation of greenhouse gas emissions by agriculture. *Nutrition Cycling in Agroecosystems* 49(1-3):221–228.
16. Liebig MA, et al. (2005) Greenhouse gas contributions and mitigation potential of agricultural practices in northwestern USA and western Canada. *Soil Tillage Research* 83(1):25–52.
17. Johnson JMF, et al. (2005) Greenhouse gas contributions and mitigation potential of agriculture in the central USA. *Soil Tillage Research* 83(1):73–94.
18. Robertson GP, Grace PR (2004) Greenhouse gas fluxes in tropical and temperate agriculture: The need for a full-cost accounting of global warming potentials. *Tropical Agriculture in Transition—Opportunities for Mitigating Greenhouse Gas Emissions?*, eds Wassmann R, Vlek PLG (Springer, Dordrecht, The Netherlands), pp 51–63.
19. Hartman MD, et al. (2011) Impact of historical land-use changes on greenhouse gas exchange in the U.S. Great Plains, 1883-2003. *Ecol Appl* 21(4):1105–1119.
20. Gutmann M (2005) Great Plains population and environment data: Agricultural data, 1870–1997 United States (Machine-readable dataset, ICPSR 04254-v1) (University of Michigan, Inter-university Consortium for Political and Social Research, Ann Arbor, MI). Available at dx.doi.org.proxy.lib.umich.edu/10.3886/ICPSR04254. Accessed August 1, 2011.
21. Minnesota Department of Agriculture (2014) Animal Unit Calculation Worksheet. Available at www.mda.state.mn.us/animals/feedlots/feedlot-dmt/feedlot-dmt-animal-units.aspx. Accessed August 15, 2011.
22. Dalrymple DG (1988) Changes in wheat varieties and yields in the United States, 1919-1984. *Agric Hist* 62(4):20–36.
23. Cunfer G (2005) *On the Great Plains: Agriculture and Environment* (Texas A&M Univ Press, College Station, TX).
24. Olmstead A, Rhode P (2008) *Creating Abundance: Biological Innovation and American Agricultural Development* (Cambridge Univ Press, New York).
25. Follett RF, Varvel GE, Kimble JM, Vogel KP (2009) No-till corn after bromegrass: Effect on soil carbon and soil aggregates. *Agron J* 101(2):261–268.
26. Akiyama H, Yan X, Yagi K (2010) Evaluation of effectiveness of enhanced-efficiency fertilizers as mitigation options for $N_2O$ and NO emissions from agricultural soils: Meta-analysis. *Glob Change Biol* 16(6):1837–1846.
27. Conant R, Ogle S, Paul E, Paustian K (2011) Measuring and monitoring soil organic carbon stocks in agricultural lands for climate mitigation. *Front Ecol Environ* 9(3):169–173.
28. Ogle S, Breidt F, Paustian K (2005) Agricultural management impacts on soil organic carbon storage under moist and dry climatic conditions of temperate and tropical regions. *Biogeochemistry* 72(1):87–121.
29. Halvorson AD, Jantalia CP (2011) Nitrogen fertilization effects on irrigated no-till corn production and soil carbon and nitrogen. *Agron J* 103(5):1423–1431.
30. Mosier AR, Halvorson AD, Peterson GA, Robertsone GP, Sherrod L (2005) Measurement of net global warming potential in three agroecosystems. *Nutrition Cycling in Agroecosystems* 72(1):67–76.
31. Conservation Technology Information Center (2015) National Crop Residue Management Survey. Available at www.ctic.org/CRM/. Accessed April 24, 2015.
32. Baker JM, Ochsner TE, Venterea RT, Griffis TJ (2007) Tillage and soil carbon sequestration- What do we really know? *Agric Ecosyst Environ* 118(1-4):1–5.
33. Powlson DS, et al. (2014) Limited potential of no-till agriculture for climate change mitigation. *Nat Clim Change* 4(8):678–683.
34. Syswerda SP, Corbin AT, Mokma DL, Kravchenko AN, Robertson GP (2011) Agricultural management and soil carbon storage in surface vs. deep layers. *Soil Sci Soc Am J* 75(1):92–101.
35. Angers DA, Eriksen-Hamel NS (2008) Full-inversion tillage and organic carbon distribution in soil profiles: A meta-analysis. *Soil Sci Soc Am J* 72(5):1370–1374.
36. Haas H, Evans C (1957) *Nitrogen and Carbon Changes in Great Plains Soils as Influenced by Cropping and Soil Treatments* (USDA, Government Printing Office, Washington, DC), Technical Bulletin 1164.
37. Parton WJ, Gutmann MP, Williams SA, Easter M, Ojima D (2005) Ecological impact of historical land-use patterns in the Great Plains: A methodological assessment. *Ecol Appl* 15(6):1915–1928.
38. Burke IC, Elliott ET, Cole CV (1995) Influence of macroclimate, landscape position, and management on soil organic matter in agroecosystems. *Ecol Appl* 5(1):124–131.
39. Robles MD, Burke IC (1998) Soil organic matter recovery on Conservation Reserve Program fields in southeastern Wyoming. *Soil Sci Soc Am J* 62(3):725–730.
40. VandenBygaart AJ, et al. (2011) Impact of sampling depth on differences in soil carbon stocks in long-term agroecosystem experiments. *Soil Sci Soc Am J* 75(1):226–234.
41. Food Security Act of 1985, Pub. L. No. 99-198, 99 Stat. 1354 (Dec. 23, 1985).
42. Kittel TFG, et al. (2004) VEMAP Phase 2 Bioclimatic Database. I. Gridded historical (20th century) climate for modeling ecosystem dynamics across the conterminous USA. *Clim Res* 27(2):151–170.
43. Thornton PE, Running SW, White MA (1997) Generating surfaces of daily meteorological variables over large regions of complex terrain. *J Hydrol (Amst)* 190(3-4):214–251.
44. Horowitz J, Ebel R, Ueda K (2010) *"No-Till" Farming Is a Growing Practice*, Economic Information Bulletin 70 (USDA Economic Research Service, Washington, DC).
45. The White House, Office of the Press Secretary (2015) FACT SHEET: U.S. Reports its 2025 Emissions Target to the UNFCCC. Available at https://www.whitehouse.gov/the-press-office/2015/03/31/fact-sheet-us-reports-its-2025-emissions-target-unfccc. Accessed April 21, 2015.

46. Showstack R (2015) White House submits greenhouse gas emission targets. *Eos (Washington, DC)* 96(8):6–7.
47. Soil Survey Staff (2006) State Soil Geographic (STATSGO) Database. U.S. General Soil Map (Natural Resources Conservation Service, United States Department of Agriculture, Washington, DC).
48. Standards ASAE EP496.3 (2006) *Agricultural Machinery Management Data* (American Society of Agricultural and Biological Engineers, St. Joseph, MI).
49. Standards ASAE D497.6 (2009) *Agricultural Machinery Management Data* (American Society of Agricultural and Biological Engineers, St. Joseph, MI).
50. US Department of Agriculture (2008) *Farm and Ranch Irrigation Survey* (USDA, National Agricultural Statistics Service, Washington, DC).
51. US Energy Information Administration. Electricity. Available at www.eia.gov/electricity/data/state/. Accessed February 1, 2014.
52. US Energy Information Administration. Natural gas. Available at www.eia.gov/electricity/cost_quality/. Accessed February 1, 2014.
53. US Energy Information Administration. LP gas, propane, butane. Available at www.eia.gov/dnav/pet/pet_pri_prop_dcu_r20_m.htm. Accessed February 1, 2014.
54. US Energy Information Administration. Diesel. Available at www.eia.gov/dnav/pet/pet_pri_gnd_dcus_r20_w.htm. Accessed February 1, 2014.
55. US Energy Information Administration. Gasoline and gasohol. Available at www.eia.gov/dnav/pet/pet_pri_gnd_dcus_stx_w.htm. Accessed February 1, 2014.
56. Vroomen H, Taylor H, US Department of Agriculture Economic Research Service (1992) *Fertilizer Use and Price Statistics, 1960-1991* (US Dept of Agriculture, Economic Research Service, Washington, DC), Statistical Bulletin (United States Department of Agriculture), No 842.
57. PRé Consultants (2011) SimaPro Life Cycle Analysis Software, version 7.2 (PRé Consultants, Amersfoort, The Netherlands). Available at www.pre-sustainability.com/simapro. Accessed February 1, 2014.
58. Adler PR, et al. (2012) *Mitigation Opportunities for Life Cycle Greenhouse Gas Emissions During Feedstock Production Across Heterogeneous Landscapes. Managing Agricultural Greenhouse Gases: Coordinated Agricultural Research Through GRACEnet to Address Our Changing Climat*e, eds Liebig M, Franzluebbers AJ, Follet RF (Elsevier, New York), pp 203–219.
59. IPCC (2007) Climate Change 2007: Working Group III: Mitigation of climate change, Section 7.4.3.2 Fertilizer manufacture, Figure 7.2. Design Energy Consumption Trends in World Ammonia Plants, IPCC Fourth Assessment Report. Available at www.ipcc.ch/publications_and_data/ar4/wg3/en/ch7s7-4-3-2.html. Accessed November 3, 2014.
60. Hongmin D, et al. (2006) *Agriculture, Forestry and Other Land Use, 2006 IPCC Guidelines for National Greenhouse Gas Inventories*, eds Eggleston HS, Buendia L, Miwa K, Ngara T, Tanabe K (IGES, Japan), Vol 4, pp 10.1–10.87.
61. Weaver JC (1954) Crop-combination regions in the Middle West. *Geogr Rev* 44(2):175–200.
62. Forster P, et al. (2007) *Changes in Atmospheric Constituents and in Radiative Forcing. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, eds Solomon S, et al. (Cambridge Univ Press, Cambridge, UK), pp 129–234.
63. US Department of Energy, Argonne National Laboratory (2014) Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) Model. Available at https://greet.es.anl.gov/. Accessed November 3, 2014.
64. US Environmental Protection Agency (2007) eGRID, version 1.1. Available at www.epa.gov/cleanenergy/energy-resources/egrid/index.html. Accessed February 1, 2014.
65. Polley HW, et al. (2010) Physiological and environmental regulation of interannual variability in CO2 exchange on rangelands in the western United States. *Glob Change Biol* 16(3):990–1002.
66. Xu L, Baldocchi DD (2004) Seasonal variation in carbon dioxide exchange over a Mediterranean annual grassland in California. *Agric For Meteorol* 123(1-2):79–96.
67. Frey C, Penman J, Hanle L, Suvi M, Ogle S (2006) Uncertainties. *2006 IPCC Guidelines for National Greenhouse Gas Inventories: General Guidance and Reporting*, eds Eggleston HS, Buendia L, Miwa K, Ngara T, Tanabe K (IGES, Hayama, Kanagawa, Japan), Vol 1, pp 3.1–3.66.
68. Ogle SM, et al. (2009) Scale and uncertainty in modeled soil organic carbon stock: Changes for US Croplands using a process-based model. *Glob Change Biol* 16(2):810–822.
69. Del Grosso SJ, Ogle SM, Parton WJ, Breidt FJ (2010) Estimating uncertainty in N2O emissions from US cropland soils. *Global Biogeochem Cycles* 24(1):1–12.
70. Del Grosso SJ, et al. (2000) General CH4 oxidation model and comparisons of CH4 oxidation in natural and managed systems. *Global Biogeochem Cycles* 14(4):999–1019.