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Featured Article

New Uses of Old Tools? Greenhouse Gas Mitigation with Agriculture Sector Policies

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Abstract *Following the failure of legislative proposals for a multi-sector greenhouse gas (GHG) cap-and-trade policy, the shift in focus to energy sector policies ignores the perhaps substantial potential for GHG mitigation from agriculture/forestry. We review estimates of the current U.S. agriculture sector contribution to GHG mitigation from a portfolio of existing sector policies in bioenergy, conservation, and research and development to compare accomplishments across programs. We then consider what opportunities and challenges may exist for increasing sector GHG mitigation by retargeting and/or expanding current programs—or for bioenergy-related mitigation, implementing proposed new programs—to serve as an alternative to cap-and-trade.*

Key words: Climate change, Greenhouse gas (GHG) mitigation, Agriculture and forestry sector, Bioenergy policy, Conservation programs, Agricultural R&D programs.

JEL codes: Q16, Q42, Q54, Q58.

Since legislation to create a multi-sector, national greenhouse gas (GHG) cap-and-trade program stalled in the U.S. Senate in 2010, the national policy discussion has shifted to achieving a broader set of goals—including energy security, economic competitiveness, and cleaner air—with policies focused on the energy sector. Energy is the source of 86% of GHG emissions in the United States (U.S. Environmental Protection Agency [EPA] 2013, pp. 3-1); therefore, placing a high priority on the energy sector for achieving GHG mitigation makes sense. Yet the agriculture and forestry sector, with a unique GHG profile, is a predominant source of non-energy emissions (U.S. EPA 2013, pp. 2-21, 2-24) in the economy: nitrous oxide and methane emissions from crop and livestock management far exceed the sector's energy-related emissions. Further, as a land-based sector, agriculture has the unique capacity to withdraw ("sequester") carbon dioxide (CO_2) from the atmosphere and store it in soil and biomass sinks. Consequently, the focus on energy ignores the perhaps substantial potential for non-energy-related GHG mitigation in the agriculture and forestry sector. Some "clean energy" policies

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can provide incentives to expand the production of bioenergy where energy and agriculture sector policies intersect, for example fuels and electricity based on agricultural and forestry feedstocks. However, such policies do not provide incentives to sequester soil and biomass carbon or reduce non-energy-related GHG emissions from crop and livestock management.

In this article we focus on mitigation in the agriculture/forestry sector in the absence of a national program targeting GHG mitigation, such as the failed cap-and-trade program. We review estimates of the current contribution to GHG mitigation of a portfolio of existing clean energy/bioenergy and agriculture and forestry sector policies. With that foundation we consider what opportunities and challenges may exist to increase mitigation through current programs—or in the case of bioenergy-related mitigation, the proposed new programs—to enable them to serve as an alternative to cap-and-trade, for example, by retargeting program expenditures and/or expanding the program's scale. To provide a benchmark for the economic potential for additional mitigation, we present estimates of the agriculture and forestry sector mitigation in the United States if cap-and-trade were introduced (with current programs in place).

Among bioenergy policies, we assess the current federal Renewable Fuels Standard (RFS2), which mandates increasing levels of renewable fuel blending into the fuel supply through 2022 (U.S. EPA 2011, p. 2). In addition, we consider the potential impact of proposed federal renewable/clean energy standards. We then turn to agriculture/forestry sector programs for conservation and research and development (R&D). The conservation programs provide financial assistance to agricultural and forestry land owners or operators for voluntarily retiring land from production or adopting environmentally-friendly practices. Though none of the conservation programs was designed primarily to promote GHG mitigation, many of the practices are well understood to achieve GHG mitigation; as a consequence, conservation programs have been recognized as a (potential) tool in the mitigation tool kit. In contrast, public (and private) funding for agricultural R&D is in the early stages of exploration as a tool to promote mitigation. R&D has been a major source of increased productivity in the sector over the last six decades or more. Though only a small share of investment is targeted to GHG mitigation, we highlight recent research that suggests R&D to increase agricultural productivity may be an important, yet overlooked source. After outlining the evidence regarding current mitigation impacts of these programs, our inquiry considers the future potential for new uses of these old tools.

A contribution of this article is that it compiles government data on the scale of programs and estimates of actual past and/or potential future GHG mitigation and associated costs, in order to compare accomplishments across programs and policies. For estimates of mitigation and mitigation costs, we employ government estimates (where available) as a starting point, but discuss sources of uncertainty or incompleteness in the accounting of GHG mitigation. Where appropriate and feasible, we highlight research findings that suggest a scale of adjustments that might be appropriate to account for leakage, non-additionality, and reversals of carbon sequestration in the mitigation estimates (including new estimates of the latter for the largest conservation program, the Conservation Reserve Program). For agricultural R&D, we rely on recent simulations linking R&D to mitigation from associated productivity gains, and present the first estimate in the literature (to our

knowledge) of the cost of agricultural mitigation in the United States through productivity-enhancing R&D.

The article is organized as follows. The second section presents the greenhouse gas profile and mitigation options for the agriculture and forestry sector. The third section focuses on the cap-and-trade legislation proposed in 2009, reporting estimates of the level and composition of agriculture and forestry mitigation achievable with the proposed program. The fourth section addresses the current and proposed energy programs and their implications for future energy-related mitigation by the agriculture and forestry sector. The following two sections focus on current agriculture and forestry conservation and R&D programs, respectively, to assess their current contributions and opportunities for further mitigation. A final section summarizes our findings.

U.S. Agriculture and Forestry Greenhouse Gas Profile and Mitigation Options

Forest and agricultural land uses encompass 89% of all land in the continental United States, with 35% in forest, 33% in pasture and range, and 21% in cropland (Nickerson et al. 2011).[1] In 2011 agriculture and forestry contributed around 9% of U.S. gross CO_2 equivalent (CO_2e) emissions[2] (including on-farm energy use), while net additions to carbon soil or biomass sinks in agriculture and forestry—through land use change and land management activities that sequester carbon in soils or biomass—offset 14% of U.S. gross emissions (U.S. EPA 2013).

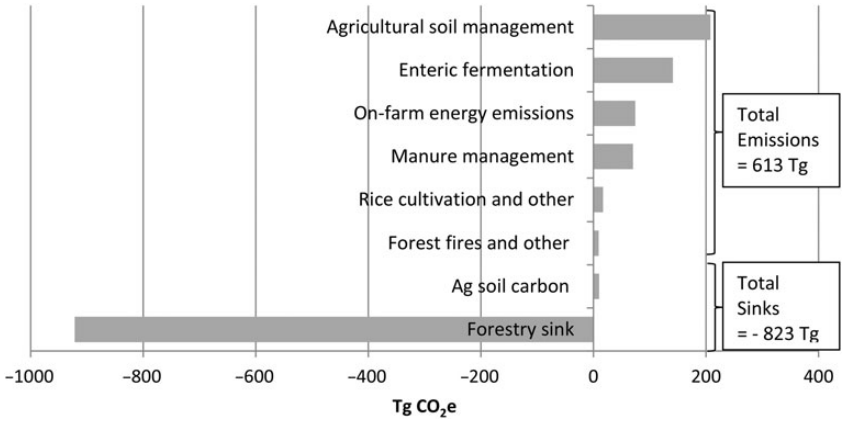
The GHG profile of the agriculture, forestry and land use sector—and the associated mitigation opportunities—differ substantially from the profile of other sectors. First, the sector is unique in providing opportunities for withdrawing carbon from the atmosphere through biological sequestration in carbon sinks in soils and biomass, particularly forests and grasslands. Second, agriculture is an emission-intensive sector: in addition to the energy-related CO_2 emission sources characteristic of most production sectors, the sector has unique crop and livestock sources of nitrous oxide (N_2O) and methane (CH_4) that dominate its emissions profile.

Crop and pasture soil management activities generate the most emissions, substantially due to N_2O emissions from using nitrogen-based fertilizers and other nutrients (Figure 1). The next largest sources are enteric fermentation (digestion in ruminant livestock), which emit CH_4, and manure management, which emit both CH_4 and N_2O. The remaining major agricultural category is energy-related emissions of carbon dioxide from on-farm fossil fuel use to support machinery use, irrigation, and crop drying, and from distributed electricity generation. Forestry emissions, a small part of the total, come primarily from forest fires. The carbon sink created by the sector consists of land use change from agricultural to forest land (afforestation) and forest management on continuing forest lands (the two activities are combined in the "Forestry carbon stock flux" category in Figure 1). In recent years, crop and pasture lands have shifted from sequestering a small

[1] *The remaining land uses are settlements (6%), wetlands (3%) and other (2%).*

[2] *See U.S. EPA (2013), pp. ES-2 - ES-3, for more details about how equivalency in terms of global warming potential relative to carbon dioxide is calculated.*

Figure 1 Agriculture and forestry greenhouse gas emissions and sequestration, 2011



Note: Negative values indicate carbon sequestration. Forestry sink includes afforestation and forest management.
Source: U.S. EPA 2013, Tables 2-12, 2-14, and 7-1.

quantity of carbon to emitting a small quantity of carbon each year ("Agricultural soil carbon stock flux" in Figure 1; U.S. EPA 2013, chapter 7).

The two primary classes of agriculture and forestry mitigation strategies are: (1) adopting on-farm activities such as changing land use and adopting production technologies, which lower GHG emissions and increase carbon sequestration; and (2) supplying bio-based substitutes for fossil fuel feedstocks to produce energy—either for transportation or for power and heat. The latter represents the intersection of agricultural/forestry and energy mitigation strategies.

Agriculture and forestry activities with the highest technical potential for GHG mitigation (which does not reflect the economic cost per ton of mitigation) are those that sequester carbon in biomass or the soil; afforestation of cropland and pasture land, and management of forest land are again at the top of the list. For land remaining in crop and pasture uses, activities with the highest technical potential—for which there is at least a medium level of confidence—include improved grazing management on rangeland and pasture, conversion of cropland to set-aside, adoption of no till on cropland, and land use change from cropland to perennial grasses (Eagle and Sifleet 2011). Though wetland restoration has also has been considered a potential GHG mitigation activity, a recent panel of experts concluded that its effect on net GHG emissions is not consistent across different locations, due to high variability in rates of carbon sequestration versus methane release (Eagle and Sifleet 2011). Also, little information exists on prior uses of restored wetlands, and the additional mitigation potential can vary significantly depending on the carbon sequestration that occurred in prior use. For example, restored wetlands on previously hayed land would generate little additional sequestration relative to previously row-cropped land.

Activities that primarily reduce N_2O or CH_4 emissions generally have lower technical mitigation potential than those sequestering carbon. Improved fertilizer management (e.g., reducing application rates and using slow-release fertilizer or nitrification inhibitors) reduces N_2O emissions

from soils.[3] Changes in livestock management that focus on the reduction of methane emissions and biogas capture (e.g., improved diet and improved manure management) also offer mitigation potential; however, some manure management approaches (such as handling manure in solid form, via composting) may increase N_2O emissions.

The second class of mitigation strategies—substituting bioenergy for fossil-fuel-based energy—can reduce GHG emissions under certain conditions, though quantifying life-cycle mitigation potential is the subject of substantial discussion and controversy (see Khanna and Zilberman 2012 for a summary of the issues). Part of the uncertainty in GHG accounting revolves around the availability of alternative technologies, and the implications for different types of feedstocks. The GHG implications of biomass-based energy will vary depending upon type of feedstock (i.e., residue/waste products vs. biomass grown for bioenergy; for the latter it will depend upon type of biomass, i.e., starch/sugar-based vs. cellulosic, annual vs. perennial crops, etc.). For feedstocks originating from additional crop production, uncertainties exist about the scale of land conversion and the resulting carbon releases associated with the production of feedstocks (direct land-use change) or crops displaced by producing feedstock crops (indirect land-use change).

Proposed Greenhouse Gas Cap-and-Trade Program

The most comprehensive federal climate mitigation policy seriously debated in the U.S. Congress was a program to cap-and-trade GHG emissions across the economy.[4] The U.S. House of Representatives passed the American Clean Energy and Security Act of 2009 (H.R. 2454);[5] however, parallel legislation stalled in the U.S. Senate in 2010.[6] In this section, we present the official U.S. government estimates of the potential for agricultural sector GHG mitigation projected for H.R. 2454, and then discuss qualifications to the estimates. The analysis incorporates the projected future path of mitigation attributable to current programs into the no-cap-and-trade baseline, including the biofuel blending mandate (RFS2) and the agricultural conservation and R&D programs (discussed in subsequent sections). The mitigation estimates for H.R. 2454 will provide a benchmark of the economic potential for additional

[3] *In some cases, a corollary benefit may be some additional carbon sequestration (Eagle and Sifleet 2011). Reduction in fertilizer use also reduces CO_2 emissions from the manufacture of fertilizer.*

[4] *A cap-and-trade program establishes a limit on total allowable emissions per unit of time for all sources covered by the cap. The total emissions cap is then allocated to covered firms in the form of allowances that can be freely exchanged among sources in a decentralized process, without approval at the program level. By permitting the trading of allowances, a cap-and-trade program can achieve cost-savings relative to traditional regulations by allowing high-cost sources to buy, and low-cost sources to sell allowances representing their allotted share of the cap.*

[5] *http://www.govtrack.us/congress/bills/111/hr2454/text. Accessed September 30, 2012.*

[6] *Currently, a range of multi-sector or energy-focused GHG mitigation policies are operating or are in development at the state and regional levels. For example, California's new multi-sector cap-and-trade program – the second largest in the world after the European Union Emission Trading System (ETS) - held its first auction in November 2012 and began operations in January 2013. In 2009, the northeastern states implemented the first mandatory cap-and-trade program in the United States (Regional Greenhouse Gas Initiative, or RGGI), targeting emission reduction in the electric generation sector. In addition, several voluntary markets are active (see http:/www.c2es.org/states-regions, and Kossoy and Ambrois 2012).*

mitigation in the sector for our discussions of bioenergy, conservation, and R&D programs below.

Economic Potential for Mitigation: Cap-and-Trade Estimates

The H.R. 2454 defined a federal cap-and-trade system for greenhouse gas emissions, where the cap would reduce covered greenhouse gas emissions 17% below 2005 levels by 2020, up to 83% below 2005 levels by 2050. Domestic agriculture and forestry mitigation activities, as well as international forestry mitigation, were included as a potential source for GHG emission offsets.[7]

The EPA's economic analysis of the program estimated that, with trading, the declining cap on allowable emissions over time could be met at a private cost of $17 per metric ton (mt) CO_2e from 2020-2029, rising to $72 per mt CO_2e from 2050-2059 (in 2008 U.S. dollars). As the total level of emissions allowed declines (the "cap") and the price rises over time, the ratio of total domestic offsets to allowable emissions increases from 3.6% to 62% over that period. The relative contributions to mitigation of different agricultural and forestry activities also change (Figure 2 and Appendix Table A1). Mitigation from forest management dominates at the lower prices in the early years, while afforestation matches forest management from 2030-2039 and dominates in the later years. The estimated emission reductions from agriculture, including cropland management (particularly N_2O from fertilizer and CH_4 from rice production) and livestock management (CH_4 and N_2O from manure and enteric fermentation) are relatively small, but increasing over time. Sequestration of additional soil carbon on cropland is limited. One contributing factor is that, even though the share of cropland in conservation tillage is increasing under cap-and-trade, the overall land area in crops is declining due to afforestation; in addition, some potential crop land mitigation activities are not included in the analysis (as discussed in the next section).

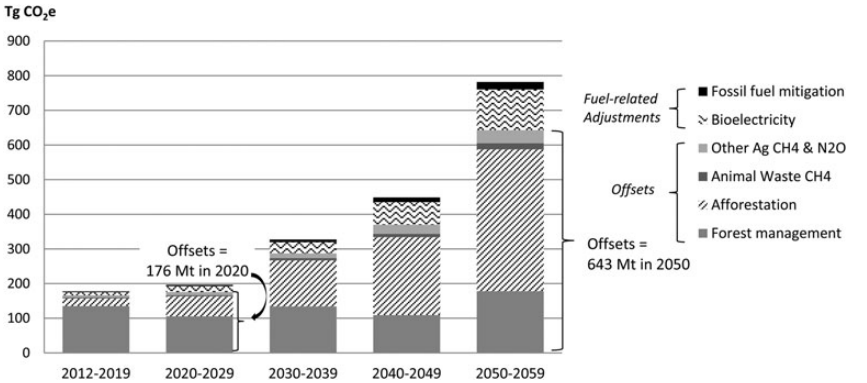
The EPA's economic analysis also estimated agricultural sector reductions in energy-based emissions. Because fossil fuel emissions fall under the cap, these emission reductions are not included in the offset program. The abatement of fossil fuel emissions (generated either on-farm by agriculture or forestry production, or upstream by input suppliers), and the substitution of bioelectricity for fossil fuel-based electricity add an additional 2% and 9%, respectively, to agriculture/forestry mitigation from 2020-2029, increasing the respective rates to 3% and 15% from 2050-2059.[8] Bioelectricity is consistently the third-most significant source of mitigation, and its share of mitigation increases over time as GHG prices rise and biomass feedstock yields increase.

In the cap-and-trade analysis, the GHG accounting for bioelectricity implicitly assumes that emissions from the combustion of biobased feedstock are fully offset by biological sequestration, so that emissions from bio-based fuel combustion are not counted. The accounting includes agricultural emissions due to crop production (i.e., emissions from any land use change that supports additional crop production, plus emissions from on-farm crop land

[7] *Introducing the opportunity to purchase offsets from entities in uncapped sectors provides those subject to the cap with additional options for lowering the costs of meeting their compliance obligations. In an offset program, unregulated firms voluntarily choose whether to earn offset credits for sale by adopting specific activities or projects that reduce emissions relative to a baseline level of emissions.*

[8] *U.S. EPA 2009. Data Appendix, HR 2454 June 23, 2009 analysis. (worksheet: HR2454 Data Annex\ ADAGE & IGEM v2.3.xls).*

Figure 2 Estimated U.S. agriculture and forestry sector GHG mitigation with American Clean Energy and Security Act of 2009 (H.R. 2454)



Note: In the proposed legislation's offset program, unregulated sectors, including agriculture, could reduce emissions and offer the reductions for sale to regulated firms to "offset" their emission reduction requirements. In addition, the energy sector's cap on allowable emissions would result in higher fossil fuel prices. In response to the higher fuel prices, agriculture could also reduce on-farm fuel use and increase the supply of agricultural feedstocks to produce bioelectricity. Offsets as a share of allowable emissions increased slowly until the final decade. In 2012, offsets were only 3.6% of allowable emissions. In 2020, they were 3.5%; in 2030, 8.1%; in 2040, 16.2%; and in 2050, 62.2%.
Source: U.S. EPA 2009.

management and fossil fuel use, upstream production of fertilizer and pesticides, as well as emissions from the one-time land use change required for additional production). The estimated emission reductions from avoiding fossil fuel combustion more than offset the estimated increase in emissions associated with land use change and crop production.

Caveats to Mitigation Estimates

Estimates from the EPA analysis are sensitive to several factors. On the one hand, the modeling may underestimate sector mitigation potential because it did not account for several categories of potential agricultural GHG reductions. The modeling does not capture the potential for any additional biofuel production beyond current policy mandates that would become economic with GHG pricing. Omitted on-farm activities include improvements in organic soil management, advances in the feed management of ruminants, changes in the timing, form, and method of fertilizer applications, and alternative manure management systems (other than anaerobic digesters, which are included).

On the other hand, the EPA analysis (2009) implicitly assumes that participation by agriculture and forestry is mandatory, which means that the incentive system embedded in the modeling includes both payments for GHG emission reductions/carbon sequestration and charges for GHG emissions for all farms. In reality, participation in the supply of offsets is voluntary, and there is no charge for GHG emissions from those not enrolled in the program. As a result, the analysis understates the potential for emissions leakage from non-enrolled suppliers in the market, or carbon sequestration reversals by enrolled landowners after their participation period ends (see the Box below for definitions of important terms and further discussion of

the environmental integrity issues that arise when designing a GHG mitigation program).

Investigating the extent of GHG leakage with a voluntary, rather than mandatory forestry offset program, Latta et al. (2011) found that the enrollment of private forested land would be substantially less than for a mandatory program, and the estimated quantities of carbon sequestration would be lower at all prices examined.[9] At $6 per ton CO_2e (in 2008 dollars), a voluntary program generated about one-third of the afforestation and one-sixth of the forest management mitigation generated by a mandatory program. At $35 per ton of CO_2e, a voluntary program generated just 40% of the afforestation and forest management mitigation level that a mandatory program did.[10]

Challenges to Environmental Integrity in Voluntary GHG Mitigation Programs

To ensure that GHG mitigation activities produce "real" reductions, several factors need to be considered when estimating the total net reduction in GHG—including leakage, non-additionality, and, for carbon sequestration, the potential for carbon sequestration reversals and carbon stock re-equilibration. These issues arise when accounting for the GHG emissions associated with voluntary GHG offsets, as well as the biofuel consumption mandate and voluntary conservation programs discussed below. If the mitigation credited under an offset program is not "real," then total emissions from the regulated community, in conjunction with offset supply, will exceed the mandatory cap specified in the program and the atmospheric GHG concentration targets will be compromised.

- **Leakage.** This occurs when a GHG-mitigation activity (such as setting aside cropland under a conservation program) displaces GHG-emitting activities to other geographic locations not within the scope of the program (e.g., outside of the United States for a national program), or in the case of a voluntary program, to sources that have chosen not to enroll. For example, a program that compensates farmers for converting highly erodible crop land to grass land, or induces the use of crops for energy feedstocks, may induce the clearing of forest land for agriculture. In this case, the GHG emissions from the additional cropland conversion would offset, at least in part, the gains from emission reductions by program participants. When calculating the net GHG impacts of a program, a full accounting of impacts would include emission increases associated with any expansion of activities, and/or shifts in their location as a result of the program. The limited empirical studies on the topic suggest that: voluntary participation in forest land preservation (removing it from pro-

[9] *Lewandrowski et al. (2004) found a similar result in their study comparing voluntary and mandatory offset programs with a 15-year commitment period for agricultural mitigation. Using a comparative static model of the agricultural sector, the analysis was able to capture afforestation, but not the full set of forestry mitigation activities.*

[10] *The analysis is not directly comparable because the EPA analysis of the legislation assumed GHG prices increase over time at 5% per year, whereas this study assumed constant prices over time.*

duction) is most likely to induce compensatory planting elsewhere; taking cropland out of production is likely to generate less leakage because conversion tends to occur on lower-productivity land; and finally, adopting land management practices such as conservation tillage or reducing fallow crops is least likely to reduce crop supply on participating land and generate a compensatory response (Murray et al. 2007).

- **Non-additionality**. The additionality of net GHG emission reductions signifies that the reductions are beyond what would have occurred without the program. For example, if a farmer would have adopted no-till cultivation without being compensated, then the GHG emission reductions from no-till adoption as a conservation program participant are not additional. Conversely, if a farmer would have abandoned conservation tillage without the program, then continued conservation tillage would be additional. Mitigation actions mandated by policies already in place would not be considered additional.
- **Carbon Sequestration Reversals (non-permanence)**. With the termination of carbon-sequestering activities such as shifting from conservation tillage to intensive tillage, or from forest to crop land use, not only does the sequestration stop (as occurs when an energy-efficiency technology is terminated), but also the carbon sequestered during an earlier time period will be released. A full description of the GHG impacts of a program would include the carbon releases resulting from future reversals of the sequestering activities after exit from the program.
- **Carbon-stock re-equilibration.** Over time and under relatively constant environmental and management conditions, rates of carbon additions and emissions tend to equilibrate and the amount of organic carbon in soils stabilizes at a constant or steady-state level (i.e., the carbon-stock equilibrium). If the relationship between additions and losses subsequently changes due to a change in soil management or land use, the soil will gradually move to a new carbon-stock equilibrium, at which point additional sequestration (or emissions) will essentially cease (Paustian et al. 2006). For set-aside lands in grass land use, sequestration generally is highest in the first decade, declines in the second decade, and is negligible in subsequent years.

Bio-energy Policies

In this section we first outline the estimated future contribution of the current biofuels mandate (which reaches its full potential in 2022) in the absence of cap-and-trade. We then highlight the new proposals for federal renewable/clean energy standards, and discuss factors affecting the extent to which they could achieve the agriculture and forestry sector energy-related mitigation potential estimated in the cap-and-trade analysis. Addressing this question is complicated by the fact that the methodology for calculating life-cycle GHG mitigation from bioenergy is being re-evaluated since analysis of the cap-and-trade legislation (and the biofuel blending mandate) was completed. Analysis of the proposed cap-and-trade legislation estimated that energy-related mitigation would represent a small share of future agriculture and forestry sector mitigation (11% from 2020-2029), primarily from

bioelectricity, with a small additional contribution from reduction in on-farm energy use.[11] Bioelectricity from agricultural and forestry feedstocks sources was consistently the third-largest source of mitigation—though a distant third after afforestation and forest management across the four decades analyzed (2020-2059). No expansion of biofuel production beyond that specified in the current Renewable Fuel Standards (RFS2) was modeled.

U.S. Renewable Fuels Standard (RFS2)

Numerous policies introduced over the last four decades have stimulated the development of the biofuel sector, with high energy prices in tandem with biofuel consumption mandates providing a particularly strong lift during the last decade. The Energy Policy Act of 2005 (EPACT 2005) created the first national Renewable Fuel Standard (RFS), which mandated domestic use of 7.5 billion gallons of biofuels by 2012.[12] Two years later, the Energy Independence and Security Act of 2007 (EISA) greatly expanded the biofuels blending mandate to 36 billion gallons of total renewable fuels per year by 2022, with a set of nested volume requirements for sub-categories of renewable fuels, each with its own sustainability provisions to ensure "real" reductions in GHG emissions.[13] Though both policies and market forces are important drivers, RFS2 is viewed as setting a floor for demand, as other policies and market economics change over time, thus reducing uncertainty for investors (Schmit, Luo, and Conrad 2011).

For RFS2's total renewable-fuel requirement each year, a maximum is imposed on the volume that can be supplied with conventional biofuel (but only if they reduce GHG emissions at least 20% relative to the reference technology. In the United States, the predominant conventional biofuel is corn-based ethanol produced using different fuel sources and technologies). The remainder of the total renewable fuel volume mandate must be met by "advanced biofuels", which reduce emissions at least 50% relative to the reference technology. Nested within the advanced biofuel volume requirement are specified minimum volumes for cellulosic biofuel and biomass-based biodiesel; the residual may come from "other advanced biofuels."

However, the mandated levels of the total and the individual components of advanced biofuels can be adjusted each year by the EPA, based on its assessment of available commercial production capacity. Under this authority, the EPA has lowered the cellulosic mandates each year. However, the EPA has approved as advanced biofuels several fuels made from conventional feedstocks (ethanol from sugar cane, biodiesel from soybean oil, and renewable diesel from waste oil, fats and grease), and consequently retained

[11] *We do not assess the potential mitigation associated with reducing farm sector energy-related emissions. For one, it represents only 2% of the sector mitigation with cap-and-trade. In addition, it would be hard to predict—absent a quantitative study—how much potential CES would realize, because its effects on different GHG-emitting energy sources will vary: though the EIA CES study forecasts that electricity prices will increase, electricity represents only 1/3 of on-farm energy use.*

[12] *Ethanol, the most prominent biofuel, began commercial–scale production in large part due to a blenders tax credit created under the Energy Policy Act of 1978. The blenders' tax credit and the complementary tariff on foreign imports of ethanol (introduced in the 1990s) were terminated at the end of 2011. Another more recent policy stimulus to ethanol production was the effective ban on methyl tertiary butyl ether (MBTE) as an oxygenate in gasoline, based on air quality grounds from EPACT 2005 (Babcock 2008).*

[13] *See U.S. EPA 2011 for an informative explanation of the legislation and the subsequent EPA rules implementing the legislation.*

the full volume mandate for total advanced biofuels and for total renewable fuels (U.S. EPA 2011).

According to the EPA's regulatory impact analysis estimates, use of RFS2-mandated renewable fuel quantities in 2022 will displace about 13.6 billion gallons of petroleum-based gasoline and diesel fuel—about 7% of the expected annual gasoline and diesel consumption in 2022, relative to U.S. Department of Energy (2007) market projections for 2022 without the mandate. In the RFS2 scenario, cellulosic biodiesel and cellulosic ethanol are assumed to provide 6.5 and 4.7 billion gallons, respectively, of the total of 36 billion gallons mandated renewable biofuel use in 2022.

With the projected portfolio of biofuel products that will supply the mandated levels for 2022, the year of full implementation of the mandate, the EPA estimates a net annual reduction of 138 Tg CO_2e[14] relative to a no-mandate reference case, based on a 30-year time period (Appendix Table A1). To develop this estimate, the EPA employed a lifecycle framework, which compares emissions with and without the mandate, from feedstock production (farm production or petroleum recovery from wells), feedstock processing, fuel transportation and blending, and fuel combustion during vehicle use. As in the cap-and-trade economic analysis, the GHG accounting for biofuels implicitly assumes that emissions from the combustion of biobased feedstock are fully offset by biological sequestration: as a consequence, combustion emissions from biobased feedstock combustion are not counted. With biobased feedstocks, the estimated emission reductions from direct fuel use combustion and other segments of the fuel lifecycle more than offset the increase in emissions from land use change—annualized over 30 years at a zero percent discount rate—and from annual crop production.

The feasibility of meeting the mandate in 2022 is uncertain for several reasons. Whether new technologies (including cellulosic) that are incorporated in the mandate can reach commercialization in time is uncertain. It may be possible to meet a greater share of the advanced biofuel volume mandate by substituting imports of sugar-based ethanol from Brazil, combined with domestic production of soybean-based biodiesel for cellulosic biofuels. For most advanced fuels, reaching the full mandate also will depend upon new investments in infrastructure and the vehicle fleet; as the mandate volumes increase over time, it will become increasingly difficult for the transportation sector to absorb them with its current technologies, which can accommodate no more than 10% or 15% ethanol in blended fuels. The exception would be for new "drop-in" fuels—those that are near-perfect substitutes for gasoline or diesel; however, progress toward commercialization is slow for these fuels as well (Coyle 2010).

Proposed Clean Electricity Standards

High fossil fuel prices, as well as federal tax incentives and grants and an extensive set of state renewable/clean energy programs,[15] have been driving the growth of renewable electricity, including bioelectricity. Major proposals for federal policy exist on two fronts,[16] though the role of

[14] *1 Teragram (Tg) is equal to 1 million metric tons (mt).*

[15] *Incentives include the federal production tax credit and energy tax credits. See the Database of State Incentives for Renewables and Efficiency (DSIRE) for information about state-level policies. Available at www.dsireusa.org. Accessed May 1, 2013.*

[16] *See Linn and Richardson (2013) for further discussion of these policy proposals.*

bioelectricity remains uncertain for both. First, on March 27, 2012, the U.S. EPA proposed a carbon pollution standard for new electric power plants, as required under the Clean Air Act (CAA). However, in order to seek input from expert panels, the EPA has delayed provisions identifying under what circumstances biogenic feedstocks for co-firing or dedicated biomass electricity generation may qualify a plant as being the best available technology under the CAA.

Second, both the U.S. President and members of Congress have put forth concepts for a federal clean energy standard (CES) that would increase the share of electricity generated from "clean" energy sources to 80% by 2035, approximately doubling current levels. The definitions of renewable and "clean" energy vary, where the broader category of clean energy may include partial credit for fossil fuels with carbon capture and storage (CCS), as well as for efficient natural gas.[17] The scale of bioelectricity will depend upon policy choices regarding the stringency, timing, and scope of clean energy mandates, where scope refers not only to which technologies qualify, but also to the relative weighting of GHG mitigation contributions from the different "clean" energy sources (Linn and Richardson 2013). Official proposals for that component of the policies have not yet been released.

The first bill to be introduced in 2012, Senator Bingaman's Clean Energy Standard (CES) Act,[18] defers decision-making about crediting for biogenic feedstocks to an expert panel. In the Bingaman proposal, annual clean energy targets[19] ramp up linearly from the current level to 84% in 2035.

To assess whether the Bingaman CES would achieve a scale of bioelectricity comparable to the cap-and-trade legislation (H.R. 2454), we compare the differences between the reference and policy scenarios for the two policies; for greater comparability of analytical tools and approach, we compare the EIA analyses of the two policies. For 2025, the projected increase in biomass-based electricity generation is somewhat higher with H.R. 2454 (83% increase beyond the reference case) than with the Bingaman clean energy standard (62% increase beyond the reference case). The difference is attributable in part to EIA revisions between 2009 and 2011 in the cost structure for biomass generation, so that dedicated energy crop plants are no longer economically feasible compared to other clean energy alternatives: all of the growth in biomass use relative to the reference case is attributable to co-fired generation, which begins to decline at the end of the period, as coal-fired plants that co-fire biomass are retired.[20] Between 2030 and 2035, the difference widens slightly. As time goes on, an increasingly tighter emissions cap under cap-and-trade, compared to flat CES energy targets past 2035, would generate an increasing divergence in mitigation.[21]

[17] *Page 135, Economic Report of President, 2011.*

[18] *http://energy.senate.gov/public/index.cfm/files/serve?File_id=b3580f37-ec8c-4698-a635-3e19f9815b9a. Accessed May 31, 2013.*

[19] *All generation from existing and new wind, solar, geothermal biomass, municipal solid waste, and landfill gas earns full credits; hydroelectric and nuclear generation placed into service after 1991 earn full credits. Partial credits are earned for generation using specific technologies fueled by natural gas or coal, based on a crediting factor that reflects the carbon intensity of each technology.*

[20] *Personal communication, Christopher Namovicz, EIA, January 2012.*

[21] *We do not assess the potential mitigation associated with reducing farm sector energy-related emissions. For one, it represents only 2% of the sector mitigation with cap-and-trade. In addition it would be hard to predict—absent a quantitative study—how much potential CES would realize, because its effects on different GHG-emitting energy sources will vary: though the EIA CES study forecasts that electricity prices will increase, electricity represents only 1/3 of on-farm energy use.*

Agricultural and Forestry Conservation Programs

In this section we discuss estimates of the current contribution to GHG mitigation of federal conservation programs administered by the USDA, and then consider what opportunities and challenges may exist to increase mitigation by expanding and/or retargeting the current programs. With projected total net outlays of $24 billion, conservation programs collectively comprised about 8% of Farm Bill outlays from 2008 to 2012 (Monke and Johnson 2010).[22] Most USDA conservation program budgets promote conservation on cropland and pasture or range lands (including afforestation), rather than on existing forested lands. Though GHG mitigation represents only one of several stated environmental goals of the conservation programs, many of the practices supported by the programs tend to promote GHG mitigation. These programs incentivize the predominant sources of potential future mitigation in the sector, based on analysis of the cap-and-trade legislation, for example sequestering soil and biomass carbon, or reducing non-energy-related GHG emissions from crop and livestock management.

In addition to considering the reported GHG impacts on currently enrolled land, we also consider the additional factors relevant to a more complete GHG accounting of cumulative program impact: non-additionality, leakage, and—for carbon sequestering activities—carbon stock re-equilibration and post-contract carbon reversals.

Before addressing specific conservation programs, we note that the USDA administers a number of agricultural commodity support and crop insurance subsidy programs that are designed to increase the returns, or reduce the downside risks of low returns, from agricultural production. These programs have the potential to induce producers to bring additional land into crop production when they make cropping profitable where it would otherwise not be. In other words, these programs counteract GHG mitigation provided by the USDA's conservation programs by releasing carbon sequestered in the soil and increasing GHG emissions from fertilizer and other inputs. Studies examining the aggregate acreage response of agricultural income support programs have found mixed land use effects across the different policies studied (e.g., Gardner, Hardie, and Parks 2010; Bhaskar and Beghin 2009). Several studies found that increases in crop insurance subsidies are associated with very modest expansions of cultivated cropland area (Claassen et al. 2011; Lubowski et al. 2006; Goodwin et al. 2004). In general, quantifying the land use impacts of these programs is difficult due to challenges in separating the impacts of program payments from those of other factors that affect land use decisions.

Conservation Reserve Program

About three-quarters of the estimated mitigation attributable to federal conservation programs stems from lands enrolled in the Conservation Reserve Program (CRP) (Appendix Table A2). The CRP is by far the largest U.S. conservation program in terms of hectares enrolled and budget; it provides annual rental payments to farmers who voluntarily retire environmentally-sensitive cropland from production over a 10-15 year

[22]*Most USDA funding for federal conservation, commodity and farm support policies, as well as other rural, food, and farm-related provisions, derives from multi-year, omnibus laws "farm bills", which must be renewed every 5 years.*

contract period, as well as cost-share assistance for establishing approved grassland or tree cover on the enrolled land. After the program's initial implementation period (1986-1990), the enrolled land area fluctuated between 13.1-14.9 million hectares through 2008 (with the exception of 1998-2000, during which time many contracts expired and enrollment dipped to 12.1 million hectares). In 2008, 14 million hectares were enrolled in the CRP at an approximate annual rental cost of $1.8 billion (averaging $125.4/ha), with 88% planted to grasses, and 10.5% planted to trees, and the small residual dedicated to wetland restoration (USDA FSA 2008).

The conversions of cropland to grassland or forest lands are two land use/land management activities that scientists agree result in net GHG mitigation on a given site (Eagle and Sifleet 2011). From 1997 to 2008, the USDA OCE (2011) estimated that CRP grass lands and afforested lands increased carbon sequestration annually by 31 Tg of CO_2e (split about evenly between grass and tree contracts), and avoided 9 Tg CO_2e of emissions due to reduced fuel and fertilizer use (see Appendix Table A2). With the exception of the first set of contracts that expired in 1997, a majority of the enrolled lands have re-enrolled in the program when their contract expired. Consequently, as the fairly constant scale of long-term enrolled land approaches a new carbon stock equilibrium, annual soil sequestration is declining (see Box above); as a result, by 2008, the annual sequestration rate is lower than the 1997-2008 average.[23]

Lubowski, Plantinga, and Stavins (2008) estimate that 15% of CRP land enrolled through 1997 would have been converted from crops to pasture, range, or forests even in the absence of CRP due to economic considerations, and consequently did not contribute any additional mitigation. Assessments of land leakage—the share of cropland enrolled in the CRP that was offset by land conversion to crop use elsewhere—have been inconclusive: estimates range from 20% (Wu 2000, 2005) to 53% (Leathers and Harrington 2000), depending on the estimation method and the geographic and temporal scope of the analysis. However, Roberts and Bucholtz (2006) raise questions about the effectiveness of both studies in statistically identifying land leakage.[24]

When CRP land is returned to cropping after a contract expires, much of the stock of carbon sequestered in the soil and in forest biomass during program participation in prior periods may be released,[25] though the extent of release will depend upon choices of rotations, tillage, and other management practices before and after participation.[26] For CRP planted in trees, some portion of the aboveground forest biomass will also be removed when land use is changed to hay, pasture, or range uses. Consequently, for a more complete accounting of the GHG impacts of CRP, these carbon releases after a contract expires should be deducted from the carbon gains calculated for the contract period.

[23] *Personal communication, Stephen Ogle, Colorado State University, September 2012.*

[24] *We also note that estimating the GHG emissions that result from land leakage requires an additional step, since the land entering and leaving production may sequester carbon and emit nitrous oxide from fertilizers at different rates.*

[25] *In contrast, the emission reductions in prior periods due to lower fertilizer use when land is converted to grass are not subject to reversals; consequently, when the land is returned to cropping, the credits for reduced N_2O stop, but no deductions against past credits are needed.*

[26] *Carbon releases from land use change generally occur over a shorter time period than it takes to sequester the carbon (Paustian 2006).*

Table 1 For CRP Land in Grass or in Trees: Post Contract Land Uses by Period of Contract Expiration (1993-97; 1998-2002; 2003-07)

CRP lands in contracts in year: with contract expirations during:	**1 1992 1993-1997**	**2 1997 1998-2002**	**3 2002 2003-2007**	**All cohorts[2]**
CRP lands in grass				
Land (mi. ha.)	1.3	11.7	2.9	15.9
Land uses in post-contract periods				
Cropped in first period	43%	18%	3%	17%
Cropped in first or second period[2]	51%	21%	nd[3]	24%
Cropped post-contract (any period)	**53%**	**21%**	**3%**	**20%**
Only crop	37%	17%	3%	16%
Mixed crop and (CRP, hay, pasture, range)[2,4]	15%	5%	nd[3]	6%
Never cropped post-contract	**44%**	**79%**	**97%**	**79%**
Only CRP	13%	61%	89%	62%
Only hay, pasture, or range	28%	16%	7%	15%
Only forest	–	–	1%	–
Mixed CRP & (hay, pasture, or range)[2]	3%	1%	nd[3]	1%
Development/other	**3%**	**1%**	**–**	**1%**
CRP lands in trees				
Land (mi. ha.)	0.12	0.79	0.34	1.25
Land uses in post-contract periods				
Some non-CRP, non-forest use	**34%**	**12%**	**5%**	**12%**
Only CRP, forest use	**66%**	**88%**	**95%**	**88%**

Source: Authors calculations using NRI data (USDA NRCS).
1. For cohort 1 (which the NRI reports was enrolled in CRP in 1992, and contract expired before 1997), the NRI provides 3 observations after expiration of1992 contract (1997, 2002, and 2007); for cohort 2, NRI provides 2 observations after 1997 contract (2002, 2007) and for cohort 3, NRI provides 1 post-contract observation (2007). Post-contract land use options are: new CRP contract, forest, hay, pasture, range, or crop.
2. Mixed land uses are only defined where at least 2 observations are available; therefore the rates for mixed land uses for "all cohorts" are calculated across hectares in cohorts 1 and 2 only, and all-cohort totals do not necessarily add up.
3. Abbreviation "nd" = not defined.
4. The shares with mixed crop and pasture/hay/range (no CRP) were 13% for cohort 1 and 3% for cohort 2, and 4% for the two cohorts combined.

Though cropland use is a condition of eligibility for enrollment in CRP, the analysis of NRI panel data examining post-contract land use for CRP contracts expiring from 1993 through 2007 (the last year for which NRI data is available) indicates that 20% of expiring CRP grass lands had converted back to cropland at some point by 2007; with some lands moving in and out of crop use, 16% were cropped throughout the post-contract period (see last column, Table 1). Of the nearly 80% of land that did not convert to cropland post-contract, most land re-enrolled in CRP at some point, with 62% reporting continuous CRP enrollment through 2007. The remainder adopted post-contract land uses that sequester additional carbon (hay, pasture and range, and forest) for either the entire time (15%), or part of the time jointly with CRP (1%).

The pattern varies across cohorts,[27] likely due to a combination of program and market conditions at the time of contract expiration (columns 1-3, Table 1). The overall pattern primarily reflects the experience of the second NRI cohort of CRP-grass contracts (expiring 1998-2002), which represents about three-quarters of the expiring contracts recorded in the NRI (the second largest wave of contract expirations continued through 2010, beyond the period of available NRI data). For the much smaller first NRI cohort with contracts expiring from 1993-1997, the rate of crop land conversion (plus hay, pasture, or range conversion for forested CRP lands) at the time of contract expiration was substantially higher,[28] and the CRP re-enrollment rate was substantially lower, than for the later cohorts. This cohort represents the first CRP contracts written—before the program instituted ranking procedures to prioritize land characteristics that provide multiple environmental benefits; as a result, they were not originally selected based on the same set of ranking criteria introduced in 1997. Further, commodity prices were relatively high from 1996-1997, when the contract expirations occurred for this cohort. With the three NRI observations post-contract for this cohort (representing 15 years), we can see that the higher share of lands from this cohort that were in cropland use at some point post-contract expiration primarily reflects a high initial rate of cropland conversion rate, rather than substantial additional conversions later in the post-contract period.

For the third cohort (with contracts expiring 2003-2007), a USDA program initiated in 2006 offered current contract holders with land expiring between 2007 and 2010 priority to re-enroll with 10-15 year contracts, or to extend their contracts for 2-5 years.[29] Despite rising commodity prices, in part due to increasing demands for ethanol feedstocks, the re-enrollment program was very successful; as a result, the CRP re-enrollment rate (89%) was much higher in this cohort, and the cropland conversion was much lower (3%) than the other cohorts.

For the much smaller set of CRP forested lands, the post-contract conversion rates are lower, with similar patterns of variations across cohorts to those for CRP grass lands. An estimated 12% of expiring CRP forested lands converted back to crop, hay, pasture, or range at some point during the post-contract period, including lands that spent part of the post-contract period in CRP or (non-CRP) forest use.

Jones, Nickerson, and Sperow (2013) provide preliminary estimates—for the largest cohort of grassland contracts (exiting between 1998-2002)—of annual carbon fluxes during CRP enrollment, and for post-contract land use across 10 farm production regions by employing an IPCC-based carbon accounting methodology. Only for the lands that were planted in crops in both observed post-contract periods did they estimate a net release of the carbon sequestered: an estimated 90% of carbon sequestered in 10 years of CRP participation on these lands (2.7 Tg CO_2e yr^{-1}) was released during the 10-year post-contract period. Adjusting the average annual mitigation

[27] *NRI data are released every five years, so are grouped into three cohorts spanning 5-year periods during which contracts ended, as indicated in Table 1.*

[28] *Consistent with Roberts and Lubowski (2007), we calculate that 66% of grass contracts exiting CRP after 1992 (i.e., not re-enrolling) converted to either cropland or hay by 1997. However, in the cropland conversion rates we report, we separate out hay from other crops and include all lands with expiring contracts in the denominator.*

[29] *A stated goal of the program was to achieve continuity in CRP's benefits, given that a large share of total land in contracts (11 million hectares) was expiring from 2007-2010 (USDA FSA 2008).*

estimate for grasslands for post-contract reversals would reduce the rate from 16 to 13 Tg CO_2e per year. Among all lands with mixed crop and other uses (4%), the other uses (including hay, pasture, range, or CRP re-enrollment) sequestered more carbon than was lost during the crop use. It is interesting to note that for the 79% of expiring grass lands that did not convert to cropping during either post-contract period, Jones, Nickerson, and Sperow (2013) estimate that the level of carbon sequestration during the 10-year post-contract period was roughly comparable to sequestration during the 10-year CRP contract period.

Estimates of carbon reversal are not available for the 12% of CRP forest lands that shifted out of CRP or forest use after their contract expired. We can say that the amount of carbon removed from forested CRP lands upon conversion is higher than for grasslands due to the substantially greater above-ground biomass accumulation, particularly during the second decade of growth. On the other hand, most timber harvested from U.S. forests is used in wood products, and most waste goes to solid waste facilities, which store carbon long-term, rather than releasing it rapidly to the atmosphere (as would occur, for example, with wood burning).[30]

Thus far, the discussion has focused on estimated CRP mitigation from 1991 through 2007/2008, during which period enrollments fairly consistently ranged from 12.1-14.9 million hectares. However, from peak enrollment in October 2007 (including extensive re-enrollments through active USDA recruitment), enrolled lands have fallen 3.9 million ha. (25%) through October 2012, to 11.1 million hectares. Several factors have contributed to changing enrollments. To meet the 2008 Farm Bill[31] enrollment cap of 13 million hectares by October 2009, the USDA allowed 1 million hectares to leave the program without an offer to renew or extend. Rising commodity prices in the latter part of the decade—in part due to increasing demand for crops for ethanol production—have made cropping a more attractive alternative for some landowners, and fewer lands have been offered for enrollment.

Higher prices may also result in greater cropland conversion rates among lands exiting the program due to the abovementioned downscaling. The majority of lands with long-lived participation in grassland CRP are no longer adding substantial additional carbon to the soil. Nonetheless, they embody a substantial store of carbon sequestered as a result of the CRP taking land out of crop production, which will be released if lands convert back to cropland use.

The decline in enrolled land in CRP affects incremental GHG mitigation from the program in two ways. First, the level of carbon sequestration on enrolled lands will decline. Second, cropland conversions are likely to increase, with the attendant carbon reversals, as more land exits the program in a time of high commodity prices, with limited opportunities to reenroll in CRP.

An estimate of the first effect is not yet available in U.S. GHG inventory reporting. However, the combination of lower hectares enrolled and the fact that many of the enrolled grass lands are approaching the new carbon stock equilibrium means that the annual sequestration from 2008-2012 on grass

[30] *Earles, Yeh, and Skog (2012) estimate that 36% of carbon from above-ground biomass harvested from U.S. forested lands remains in long-term storage after 30 years.*

[31] *The 2008 Farm Bill is formally known as the Food, Conservation, and Energy Act of 2008.*

lands declined further relative to the 1997-2008 average.[32] In contrast, for CRP lands planted in trees (10.5% of total), the rate of sequestration of above ground biomass increases in the second decade of enrollment (Smith et al. 2006), which will moderate the impact of the decline in enrollments on annual sequestration in CRP lands planted in trees.

In estimating the second effect, the U.S. Greenhouse Gas Inventory (U.S. EPA 2013) assumes full cropland conversion[33] (with annual releases of 0.5 Tg ha^{-1}), which results in carbon releases of 1.4, 2.0, 3.6, and 3.7 Tg CO_2e yr^{-1} from 2008-2011, respectively. Applying the same approach to the 2012 enrollment (a further reduction by 1.4 mi. ha), we estimate an additional release of 7 Tg CO_2e yr^{-1} in 2012 as the number of enrolled hectares declined further. To the extent that some of the land is not being converted back to cropland, this estimate is an upper bound.[34]

Other Land Retirement/Preservation Programs

Two other conservation programs remove cropland from production, either permanently, or for an extended period; however, their small sizes limit their current potential for GHG mitigation. The Wetlands Reserve Program (WRP) offers financial assistance to restore, enhance and protect wetlands on land retired from agriculture, and on some lands, purchases permanent or 30-year easements for the wetlands. The WRP had over .8 million hectares enrolled through 2008 and is capped at a total enrollment of about 1.2 million hectares (USDA NRCS 2008). As noted above, the direction of the net GHG impacts of wetland restoration activities can vary across different types of soil, past uses, and wetland types (Eagle and Sifleet 2011). Further, a recent study did not find statistically significant increases in carbon stocks associated with wetland restoration projects funded by WRP (and, to a lesser extent, CRP) (Gleason et al. 2008). For this reason, we do not include mitigation estimates for wetland restoration through either WRP or CRP.

The Grassland Reserve Program (GRP) purchases contracts or easements on grazing lands that otherwise could be converted to cropland or developed land to retain the lands in grazing use. The GRP also supports the restoration and enhancement of grassland, including rangeland, pastureland, shrubland and certain other lands. As of 2008, GRP protected about 43,300 grassland hectares from conversion to crop land using permanent easements, and another 253,000 hectares using 0–30 year term contracts.[35]

[32]*Personal communication, Stephen Ogle, NREL, Colorado State U., September 2012. The last year for which the data in the USDA inventory, which does break out CRP sequestration, are available is 2008 (USDA OCE 2011). The U.S. Greenhouse Gas Inventory does not explicitly break out sequestration on CRP lands; however, as noted below, it does include an accounting for carbon reversals from 2008-2011 (U.S. EPA 2013).*

[33]*NRI data are not yet available for the years after 2007 to identify the longevity of CRP participation of exiting lands, nor the post-CRP land use; however, we note that land in crops has increased by 2.3 million ha between 2007 and 2012, though the quantities have fluctuated greatly during the period (USDA NASS, Crop Production Annual Summary 2007, 2012). Available at: http://usda.mannlib.cornell.edu/MannUsda/viewDocumentInfo.do?documentID=1047.*

[34]*Temporary carbon removal from the atmosphere and storage in the biosphere has the potential to mitigate climate change by avoiding some radiative forcing over the period of storage. In a review of the literature, Brandão, Levasseur, et al. (2013) find that there appears to be no consensus on the most appropriate ways of considering and quantifying the gains, and illustrate that the benefits depend on the time horizon of analysis selected.*

[35]*USDA OBPA 2011.*

The NRCS estimates annual carbon sequestration on GRP grass lands was .007 Tg of CO_2e yr^{-1} in 2007, increasing to .033 CO_2e yr^{-1} in 2010 (Appendix Table A2).

The additionality and leakage of sequestration under GRP has not been studied; given that GRP and CRP both keep land out of cropping uses, the CRP estimate may represent reasonable best guesses of the scale of additionality and leakage. Relative to CRP, the permanence of carbon sequestration will be enhanced through GRP to the extent that some of the easements (85% for WRP, 17% for GRP) are permanent, relative to the CRP 10-15 year contract period.

Agricultural Working Lands Programs

Agricultural "working lands" conservation programs such as the Environmental Quality Incentive Program (EQIP) and Conservation Stewardship Program (CStP) pay participants to voluntarily adopt or to maintain and enhance conservation practices on farmland that remains in production, including conservation tillage, precision use of fertilizers and pesticides, and anaerobic digesters on dairy farms. In 2008, of payments that totaled $943.4 million (about half the scale of CRP), EQIP provided $42.5 million to assist farmers with adopting conservation tillage on 1.1 million hectares, $35.7 million to adopt improved nutrient management on 1.6 million hectares, and $592.4 million to adopt improved livestock practices (Horowitz and Gottlieb 2010, USDA NRCS 2009). The CStP, authorized by the 2008 Farm Bill to pay participants for conservation actions that enhance conservation performance beyond a stewardship threshold, targeted enrollment of about 5.2 million hectares each year (with continuing enrollment in following years) over the 2009-2012 period, at an average cost (for all practices funded) of $44.5 $ha.^{-1}$ yr^{-1} (CStP replaced the similar Conservation Security Program, which funded conservation enhancements from 2002-2008 period). A third program is Conservation Technical Assistance, which provides technical assistance to individuals and entities seeking to adopt resource-conserving practices.

The technical GHG mitigation potential for many activities funded through these programs has been well studied: scientific agreement exists that no-till and improved grazing management sequesters carbon, but there is less scientific certainty about estimating the mitigation potential of other types of conservation tillage, and practices that reduce fertilizer N rates (Eagle et al. 2012).[36] The NRCS estimated GHG mitigation in 2010 to be 3.97 Tg CO_2e yr^{-1} through EQIP, and less than .01 Tg CO_2e yr^{-1} through CStP, primarily from producers adopting wildlife habitat management, prescribed grazing, windbreaks, and shelter breaks (Appendix Table A2).

However, it is uncertain how much of the mitigation associated with these working lands programs is additional. Indeed, recent research suggests that a substantial percentage of some practices may not be additional. Estimates of additionality for conservation tillage ranged from 51% in a study using data from 26 States (Claassen et al. 2013), to less than 20% in a study on Ohio (Mezzatesta et al. 2013).[37] Higher levels of additionality were

[36]*Drought conditions have been found to turn rangelands into a carbon source, if at least two-thirds of the area is in drought conditions (Zhang et al. 2010).*

[37]*Additionality is defined somewhat differently in these two studies. Mezzatesta et al. (2013) use matching estimators to quantify additionality and define it as the average increase in the proportion of the land enrolled farmers who adopt a conservation practice relative to the proportion they would have adopted*

found for structural and vegetative practices: 79-80% for buffer practices and soil conservation structures (Claassen et al. 2013), and above 80% for filter strips and cover crops (Mezzatesta et al., 2013). Additionality may be higher for practices that are more expensive to install, and are least likely to provide operators with on-farm benefits in the short run (Claassen et al. 2013). Also, EQIP provides funding to operations that are subject to federal, tribal, state, and local environmental regulations, and (through its air quality initiative) to operations in counties that are designated as non-attainment according to Clean Air Act requirements (USDA NRCS 2013). While some of these funded practices may mitigate GHGs, the mitigation would be attributable to the regulations rather than to EQIP. Similarly, some mitigation from Conservation Technical Assistance, which is estimated to provide 8.2 Tg CO_2e yr^{-1} in mitigation benefits annually, may not be additional to conservation programs, since technical assistance is provided to producers and entities to help them comply with various environmental regulations.

As with the land retirement/preservation programs, the mitigation estimates developed by the program agency only take into account potential mitigation on currently funded contracts: they do not consider whether farmers continue the adopted activities post-contract. For the working land practices that sequester carbon, such as improved grazing management, shelter belts, and conservation tillage, terminating the practices post-contract will result in carbon releases, reversing the benefits of past carbon sequestration; this is similar to the land retirement activities discussed above. In contrast, terminating working land practices that reduce GHG emissions, such as adopting anaerobic digesters or changing livestock feed, will not reverse past mitigation benefits, though potential future emission reductions that would occur if the practices were continued will be foregone.

While the EQIP and CStP contracts cover a maximum of 10 years, the program payments are intended to offset adoption costs; because the practices are expected to be profitable over the long term, the expectation is that farm operators will continue the practices after contract termination. However, we are not aware of any studies that examine practice continuation after EQIP or CStP contract termination, so the permanence of these practices in mitigating GHG is uncertain.

Forestry Conservation Programs

Although about two-thirds of the 271 million hectares of forestland in the United States is privately owned, after 2008 only two USDA conservation programs focused on forestlands. In the Forest Legacy Program (FLP), the USDA Forest Service cooperates with states to purchase permanent conservation easements on private forestlands (and then take them out of production). By January 2012, about .91 million hectares had been protected through the FLP, which is almost 60% of the currently enrolled CRP hectares planted in trees. The Healthy Forests Reserve Program (HFRP), which aims to promote the recovery of endangered and threatened species, improve biodiversity,

without funding. Additionality is defined in Claassen et al. (2013) as the difference between the probability of adoption on farms receiving a conservation payment and the weighted average proportion of similar farms that adopted the practice without a conservation payment. Both studies use matching techniques to quantify additionality.

and enhance carbon sequestration, restores and protects forestland through permanent easements, 30-year contracts, and 10-year cost-share agreements with the USDA NRCS. Enrollment in HFRP is capped at about .81 million hectares.

No estimate is available for the GHG mitigation benefits arising from the FLP. We do note that the limited empirical studies on leakage suggest that voluntary participation in forest land preservation is more likely to induce compensatory planting elsewhere, and therefore to generate substantial leakage compared to converting cropland to grass land or adopting carbon-sequestering activities on working lands (see Box above). The GHG mitigation benefits arising from HFRP are also uncertain, but few would be attributable to HFRP if most HFRP contracts assist landowners in meeting existing requirements for species protection.

Future Mitigation Potential with Conservation Programs

In the absence of cap-and-trade, the ability of conservation programs as currently configured to provide GHG mitigation at levels projected in the cap-and-trade analysis is limited for several reasons. First, even achieving the level of mitigation attributed to current conservation programs included in the baseline of the cap-and-trade analysis (52Tg CO_2e yr^{-1}, Appendix Table A2) is not likely to be feasible, given that the estimate does not consider post-contract land use changes, additionality or leakage. Indeed, preliminary estimates suggest that accounting for just the former on CRP grassland contracts would reduce the 1997–2008 baseline for grassland mitigation from 16Tg CO_2e yr^{-1} to 13Tg CO_2e yr^{-1}.

Second, the preceding review of conservation programs highlights that across all programs, priorities diverge from being in a program to generate the greatest GHG mitigation per dollar spent. In particular, relatively little emphasis is currently placed on forest activities — afforestation of agricultural lands and forest management activities—which the cap-and-trade analyses identified as having the greatest economic potential. Conservation programs supporting forest management activities have existed in the past, but are no longer funded. At present, conservation programs are targeted to achieve a wide range of agri-environmental goals. Re-targeting program enrollment priorities could entail tradeoffs with the programs' other environmental objectives (Cattaneo et al. 2006). Some studies have examined whether carbon sequestration through afforestation and other environmental services are complementary: complementarities exist with erosion control and water quality improvements; the results are mixed for wildlife benefits, varying with species (e.g., Nelson et al. 2008; Haufler, ed. 2005; Plantinga and Wu 2003).

Further, among working lands programs, much of the programs' expenditures support compliance with regulations or reward good stewards (farmers who have already adopted good practices without public financial support) for which the GHG benefits are non-additional. Yet a recent study sponsored by the USDA (USDA OCE 2013) identified various best management practices that could supply substantial mitigation at relatively low costs. Among land management activities, low-cost practices include retiring organic soils from commodity production, restoring riparian forests (particularly in the Southeast and Delta states), and restoring forest

wetlands. Also a number of manure management practices represent substantial opportunities for adoption at low breakeven prices.

Third, incentivizing the additional mitigation by increasing the size of conservation program payments to farmland owners if they adopt GHG-mitigating practices could in theory induce additional mitigation via conservation programs beyond that included in the baseline of the cap-and-trade analysis. However, the offset program expenditures required to induce the additional mitigation in the agriculture/forestry sector are $3B from 2020-2029 (176 Tg CO_2e mitigation purchased at $17 mt^{-1}, Appendix Table A1 and Figure 2), which is comparable in scale to the current level of conservation program expenditures (Appendix Table A2). As the emissions cap becomes more restrictive over time, offset program payments increase to $11B from 2050-2059 (for 643 Tg CO_2e supplied at $72 mt^{-1}, Appendix Table A1). Yet the current budget reality is one of fiscal restraint, which leads to declining federal conservation budgets and program sizes. As noted above, an alternative source of finance would be to re-program funding in conservation programs to place a greater priority on GHG mitigation.

Agricultural R&D Programs

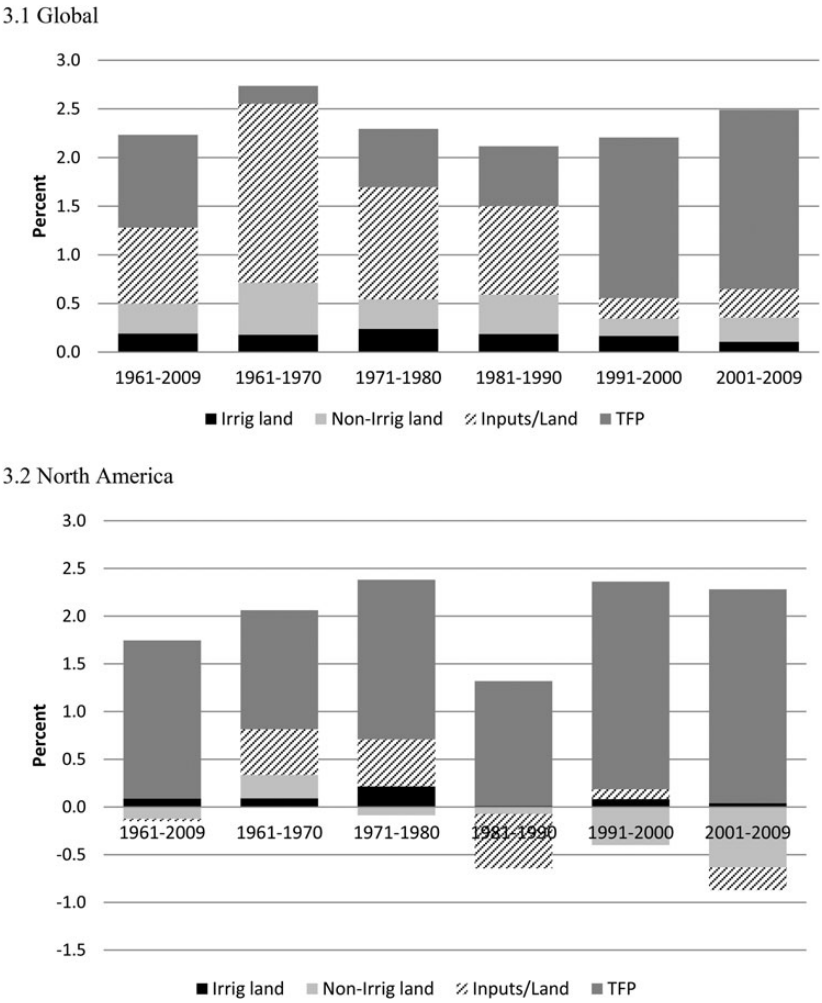
The prior two sections focused on bioenergy and conservation policies. The technical and economic mitigation potentials of the activities they incentivize—supply of bioenergy and the adoption of various land use change, land, and animal management practices—have been widely studied, as highlighted in the discussion above. In contrast, the economics of GHG mitigation from R&D-related productivity improvements has not yet been extensively studied.

Consequently, before discussing estimates of the mitigation potential of R&D investments in this section, we provide some context. We first highlight the trends over the past five decades in agricultural R&D funding for productivity improvements, and the contribution of productivity growth (compared to input growth) to the growth in agricultural output; we then provide a brief overview of the literature on the market dynamics of productivity improvements and GHG mitigation. Subsequently, we focus on the three studies that specifically tie the relationship between productivity gains and GHG mitigation to investments in agricultural R&D. Two of these studies address historical R&D investments, and one addresses future additional investments beyond their current levels. Because both historical studies were conducted on a global scale, we highlight how the costs of GHG mitigation from U.S. investments in R&D might differ from global costs; the study on future investments does provide a U.S. estimate.

Public and Private Agricultural Research Programs and Productivity Growth

Public and private sector investments in agricultural R&D have increased global and U.S. agricultural productivity dramatically over the past six decades or more (Fuglie, Wang, and Ball 2012; Alston et al. 2010). From 1961-2009, agricultural production increased more than 150% globally, and 115% in the United States (Fuglie 2012). The sources of output growth can be decomposed into (1) growth in inputs, and (2) growth in total factor productivity (TFP). Measured as output per unit of all inputs, total-factor

Figure 3. Sources of growth in agricultural output, 1961-2009



Source: Fuglie 2012.

productivity captures the extent to which fewer resources are required to produce a given level of output.

At the global scale, TFP grew from a small source to the predominant source of output growth over the five examined decades (Figure 3.1). Global land in agriculture (irrigated and non-irrigated) increased modestly throughout the period; other resource inputs (such as fertilizer and machinery) declined from the predominant source to a relatively small source by the end of the five decades. In contrast, virtually all of output growth in the United States over the last 50 years has been driven by increases in TFP: the overall size of the resource base in agriculture has barely increased, and land in agriculture has declined (Figure 3.2).

Growth in TFP is strongly associated with the adoption of new technologies that raise yields or lower costs.[38] Public investment in agricultural R&D

[38]*Total factor productivity will also reflect economies of scale and changing composition of output, as well as changes over time in key environmental conditions affecting yields, including local growing*

is a major source of new agricultural technology; it also complements other productivity-enhancing activities like extension, education, infrastructure, and private R&D (Heisey, Wang, and Fuglie 2011). Research and development investments typically begin boosting TFP within 3-5 years, with benefits peaking in 10-20 years, followed by the depreciation of the "knowledge capital" as technologies become ineffective or obsolete (Huffman and Evenson 2006; Alston et al. 2010).

Over the past five decades, total real spending on productivity-related agricultural research by U.S. federal and state agricultural research institutes[39] rose from about $1.5 billion (2008 dollars) in 1961 to about $2.5 billion in 2008 (Appendix Table A3). However, real growth in these public expenditures slowed dramatically after the early 1980s and has declined by more than 20% since peaking in 1994. Over the same period, private agricultural inputs research in the United States, that is, private research, excluding food research, grew from about $1.3 billion to over $4 billion (Fuglie et al. 2011).

Historical R&D-driven Technological Change

Though very little R&D investment has been specifically targeted for GHG mitigation, a growing body of literature has suggested that agricultural R&D that increases the productivity of the sector has been a powerful global GHG mitigation strategy (Borlaug 2007; Burney, Davis and Lobell 2010; Stevenson et al. 2013), or could be in the future (Wise et al. 2009; Choi et al. 2011; Havlik et al. 2013; Lobell, Baldos, and Hertel 2013; and Jones and Sands 2013). Most of the literature linking agricultural productivity to GHG mitigation focuses on the role of exogenous improvements in crop productivity to reduce crop land expansion, and thereby avoid the associated carbon dioxide (CO_2) emissions. However, due to market feedback effects across regions and between crop and livestock commodities, the relationship between productivity and land use change is more complex and will vary across heterogeneous regions.

As prices fall with declining costs of production, demand for agricultural products will increase (sometimes referred to as a "rebound" effect), thus promoting the expansion of agricultural production, and therefore further cropland conversion and associated GHG emissions. In a single agricultural commodity model, Hertel (2012) has recently shown global cropland expansion necessarily declines with higher productivity only if agricultural demand is inelastic and the productivity changes are globally uniform. Simulations illustrate how the patterns of cropland conversion can vary across regions: the ones that are more likely to experience higher GHG emissions with increasing productivity are those with low yields, a high supply-elasticity of land, and high emission-intensity per unit of output, which characterizes Sub-Saharan Africa and Latin America, but not North America (Lobell, Baldos, and Hertel 2013).

Further complexity is added when the livestock sector (whose costs are directly affected by crop as well as livestock productivity through its

conditions (length of growing season and precipitation), soil quality, and the availability of water for irrigation (Fuglie, MacDonald, and Ball 2007).

[39]*Research and development on post-harvest, environmental, and rural development issues is excluded. To develop the data series for the real value of R&D, expenditures are adjusted by the costs of doing research, which increased at a faster rate than the CPI (Heisey, Wang, and Fuglie 2011).*

demand for feed) is explicitly incorporated, and when the GHG accounting is expanded to include non-CO_2 emissions from crop and livestock. When lower costs lead to a greater scale of crop and livestock production, non-CO_2 emissions from crop and livestock production will increase, and GHG tradeoffs will occur when the intensification of fertilizer per unit of land is substituted for additional land on the input side. Among the studies that include non-CO_2 emissions and livestock feedback effects, the results are mixed. The one study using a U.S.-only model (Baker et al. 2013) finds that decreases in crop emissions from projected future crop and livestock yield increases are essentially offset by increases in livestock emissions. In contrast, the studies using global models (Havlik et al. 2013; Jones and Sands 2013) found that net GHG emissions decline in the United States (or the industrialized countries), as well as globally. In global models, the increase in livestock production due to lower costs occurs in developing countries (particularly Latin America and Asia), where livestock productivity is growing faster than in the United States. The difference in results may be due to the fact that the U.S. model does not capture this global market dynamic.

We are aware of two studies that link historical R&D investments in the crop sector to productivity improvements and their implications for net GHG emissions. We focus on comparing the estimated costs of mitigation in the studies; because they use different methodologies and also examine different scales of productivity improvements, their estimates of mitigation are not comparable. Burney, Davis, and Lobell (2010) estimate the GHG savings associated with total global crop productivity growth from 1961-2005. In their ("Borlaug") approach, they specify two alternative frozen technology (i.e., no yield growth) counterfactual scenarios—one assumes that as population grows, the standard of living (crop production/person) grows following historical trends over time (with yield growth); the other scenario assumes the standard of living is fixed at 1961 levels for the growing population. The authors then calculate the increase in cropland area required to meet these food production goals, with technology and input use per unit of land frozen at 1961 levels, and calculate associated GHG emissions from land use change, rice production, fertilizer production, and fertilizer application to cropland. Burney, Davis, and Lobell (2010) estimate that global investments in agriculture have helped to avoid GHG emissions at an effective cost of \$5 and \$9 per mt CO_2e for the two scenarios.[40]

However, Burney, Davis, and Lobell do not take into account the responsiveness of market supply and demand to price and income changes. Consequently, estimates of cost per mt avoided are based on what appear to be unrealistic estimates of land use conversion over the period for the two frozen technology scenarios: global cropland would double and triple, respectively, for the frozen and growing standard of living scenarios.

In contrast, the second study develops scenarios of population and income growth and employs a global computable general equilibrium model (GTAP-AEZ) to endogenously determine the changes in food prices

[40]*The estimated ranges for emissions avoided in BDL are: 9.9 - 21.6 Gt CO_2e per year (historical standard of living growth rates) and 5.2 - 8.8 Gt CO_2e per year (standard of living frozen at the 1961 level). These estimates are based on the assumptions that 34% of yield growth is due to R&D and 70% of global R&D targets productivity improvements. We converted the units of BDL's estimates from Gt C to Gt CO_2e and from 2000 international (PPP) dollars to 2008 U.S. dollars, to be consistent with the other estimates in the article.*

and land use in response to the scenarios (Stevenson et al. 2013). The authors consider the economic, land use and GHG implications of a more narrow productivity scenario by removing the gains in cereal productivity attributed to the widespread adoption of improved crop varieties in developing countries. Estimated emission reductions are more than an order of magnitude smaller than in Burney, Davis, and Lobell (2010).[41] However, it is not possible to determine how much of this difference is due to the different scope of productivity change modeled vs. the different methodology. We develop an estimate of the R&D costs of improved crop germplasm in developing countries,[42] and calculate the range of mitigation costs for their base case counterfactual to be $19-27 per mt of CO_2e avoided. This scenario allows prices for crop staples to rise much higher than observed historically, thereby cutting food consumption substantially below the levels observed over time. Cost estimates for this scenario, which in some ways parallels Burney, Davis, and Lobell's frozen standard of living scenario, are 2-3 times higher than those authors' cost estimates for their frozen standard of living scenario ($9 per mt CO_2e).[43]

Given that our focus is on U.S. policies, the question arises as to what insights do the global studies provide regarding the costs of emissions avoided from U.S. R&D investments? We sketch an exercise downscaling the Burney, Davis, and Lobell estimates to the United States (the R&D scenarios in Stevenson et al. (2013) only cover developing country investments in crop genetic improvements, and thus are not suited for downscaling to the United States). We focus on differences between the United States and global values for two key parameters, avoided emissions from land use change and R&D per hectare. Both Stevenson et al. (2013) and Burney, Davis, and Lobell (2010) found that most of the estimated emissions avoided were due to avoided land use change: the role of non-CO2 emissions from crop or livestock production was small.

Estimated U.S. aggregate yield growth between 1961 and 2005 is comparable to estimated global yield growth, which means that in the frozen technology counterfactuals, the simulated U.S. rate of cropland conversion would be comparable to global rates. However, because average carbon stocks in biomass and soils are lower in the temperate zone than in the globe as a whole, we estimate that, on average, avoided land-use-change emissions per hectare of cropland in the United States—taking into account sequestration for perennial crops—would be 76% of the global emission rate. For the estimates of global R&D costs per mt of CO_2e mitigation, we estimate that U.S. productivity-oriented research expenditures per hectare of cropland (arable plus permanent crops) from 1961-2005 were 1.9 times the global average (Appendix Table A3).[44]

[41] *These authors include spillover effects in developed countries.*

[42] *The estimate is the sum of the international research costs incurred by crop-oriented centers in the Consultative Group for International Agricultural Research (CGIAR) system, plus half the costs of developing country productivity-oriented research, following the rule of thumb that half observed increases in crop yields may be attributed to improved varieties, and the other half to improved crop management (Day-Rubenstein et al. 2005).*

[43] *Stevenson et al. (2013) do not provide as many numerical details about their "maintained consumption" model, but very roughly it might imply costs per mt ranging from $12-$23.*

[44] *We maintain the conservative assumptions employed in the global scenarios that indicate 34% of yield growth is due to R&D and 70% of R&D targets productivity improvements.*

With these adjustments to the Burney, Davis, and Lobell (2010) global analysis, a preliminary estimate is that U.S. R&D investments in agriculture have helped to avoid GHG emissions at an effective cost of $8-13 per mt CO_2e yr^{-1}. This is comparable to the cost per mt associated with past afforestation in the Conservation Reserve Program (Appendix Table A2).

Future R&D-driven Technological Change

Stagnant real funding for public agricultural research since the 1980s may be causing agricultural TFP growth in the United States to slow down, although statistical analyses of productivity growth trends are inconclusive. The USDA ERS simulations indicate that, if public research spending is held constant at the 2005-2009 average, TFP growth will stabilize at 1.4% per year (Heisey, Wang, and Fuglie 2011).

Lobell, Baldos, and Hertel (2013) consider the global mitigation implications of additional R&D funding beyond current baseline levels. These authors frame their scenario as increasing R&D to achieve full adaptation to climate change—that is, increasing R&D sufficiently to offset the negative yield impacts of temperature and precipitation changes and return TFP to no-climate change levels. Lobell, Baldos, and Hertel's (2013) approach to calculating the necessary R&D expenditures required is to employ an estimated R&D elasticity of TFP of .3 across all regions, based on estimates from the literature.

Lobell, Baldos, and Hertel (2013) estimate that additional future global investments in agriculture R&D could avoid GHG emissions at an effective average cost of $11-22 per mt CO_2e.[45] For future R&D investments in the United States and Canada only, these authors estimate a cost of $25 per mt CO_2e (Appendix Table A3).[46] The literature suggests that the responsiveness of TFP to R&D—a key parameter affecting the scale of R&D investment required to mitigate climate change impacts—is higher for developed countries than for developing countries (and consequently for the global average). If this difference were taken into account, the estimated U.S. cost would be lower than $25 per mt CO_2e.[47] As a final note, we observe that none of the mitigation cost estimates takes into account the lag between research investments and observable impacts on productivity.

Conclusions

Following the failure of legislative proposals for a multi-sector GHG cap-and-trade policy, the shift in focus to energy sector policies ignores the perhaps substantial potential for GHG mitigation from agriculture. In this article we have compiled government estimates of the current agriculture/forestry contributions to mitigation from bioenergy, conservation programs, and R&D, and highlighted potential opportunities and challenges to increase mitigation through the current programs, or in the case of energy-related mitigation, by adopting proposed new programs.

[45] *They estimate that global investments in agriculture to adapt to climate change could avoid .25-.43Gt CO_2 per year on the global scale, where 1 Gigaton (Gt) = 1000 Tg.*

[46] *Personal communication, T.W. Hertel, April 2013.*

[47] *Compare R&D elasticity of TFP estimates in Wang, Schimmelpfennig, and Fuglie 2012 (.29) and Huffman and Evenson 2006 (.47) for the United States, with estimates for developing countries in Nin Pratt and Fan (2010), which for the most part are substantially lower.*

Among current programs, fully implementing the biofuel consumption mandate in the Renewable Fuel Standard of 2007 (RFS2) would result in more than 2.5 times the estimated mitigation currently achieved from the next highest source, USDA conservation programs. However, this level of estimated annual RFS2 gains does not accrue until 2022, the year in which full implementation of the mandate is scheduled. Also, the estimated gains critically depend on the assumption that—in the interim—the industry will surmount the technological and economic challenges of developing and commercializing low-emission advanced biofuels, and will also make needed investments in new fuel distribution infrastructure and vehicle fleets.

Voluntary USDA conservation programs are estimated to have generated reductions in net annual emissions, on average, equivalent to 11% of the average annual agricultural emissions from 1997-2008. Taking into account non-additionality, leakage and carbon sequestration reversals lowers estimated mitigation, as does the recent 25% decline in enrollments in the Conservation Reserve Program, which is the source of the majority of estimated conservation program benefits. No estimate exists of mitigation achieved with current R&D programs, but research suggests global investments in R&D to increase agricultural productivity have been a powerful, low-cost tool for helping to achieve greenhouse gas mitigation.

In contrast, with the implementation of a cap-and-trade program, estimates indicate that the dominant source of mitigation would be on-site agriculture and forestry activities—with the primary contribution coming from afforestation and forest management. In the absence of cap-and-trade, opportunities exist to achieve this potential with the current policy framework, but various challenges exist.

The USDA has a tradition of conservation programs that pay for environmental services on agricultural lands, and to a lesser extent on non-industrial forest lands. Conceptually, programs that pay for environmental services function similarly to cap-and-trade offset programs that provide incentives to supply environmental services. Achieving additional mitigation through conservation programs would require substantial additional funding: indeed, the offset program payments from 2020-2029 estimated to be needed in a cap-and-trade program would be equivalent to the current budget of USDA conservation programs. As the emissions cap becomes more restrictive over time, offset program payments increase, more than tripling in size by 2050-2059.

However, current conservation programs target different priorities than would be implied by a program solely targeting GHG mitigation. One gap in coverage is the lack of current programs focusing on forest management; further, as evidenced by CRP enrollment patterns, cropland retirement incentives disproportionately concentrate on conversion to grass lands rather than afforestation. Additional mitigation could be achieved by expanding the scope of programs to include forest management (which has been covered by programs in the past), and re-targeting funds among the land retirement and working lands programs. However, re-targeting program enrollment priorities could entail tradeoffs with the programs' other environmental objectives.

A promising, complementary option for promoting mitigation from soil/biomass and animal management is investing into productivity-enhancing R&D. Developing new technologies with low or no GHG emissions—particularly for energy—has long been recognized as a source of GHG

mitigation. The new insight highlighted by this paper is that investing in R&D to increase agricultural productivity—for example, to adapt to productivity shocks from climate change—can serve the dual purposes of promoting mitigation as well as productivity. However, allocating scarce budgets to support R&D is a matter of priorities.

We conclude by noting that the budget-driven nature of conservation and R&D programs contrasts with regulatory programs, such as cap-and-trade, in which the scale of mitigation incentives depends on how tightly the cap is set, and how many offsets the agriculture and forestry sector is allowed to sell to firms covered by the cap. Given the current context of declining federal budgets, the self-financing feature of cap-and-trade programs (as well as other regulatory programs, such as clean energy standards) provides a distinct advantage relative to programs requiring incentive payments to change behavior.

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References

Alston, J.M., M.A. Anderson, J.S. James, and P.G. Pardey. 2010. *Persistence Pays: U.S. Agricultural Productivity Growth and the Benefits from Public R&D Spending*. New York: Springer.

Babcock, B. 2008. Distributional Implications of U.S. Ethanol Policy. *Applied Economic Perspectives and Policy* 30 (3): 533–542.

Baker, J.S., B.C. Murray, B.A. McCarl, S. Feng, and R. Johansson. 2013. Implications of Alternative Agricultural Productivity Growth Assumptions on Land Management, Greenhouse Gas Emissions, and Mitigation Potential. *American Journal of Agricultural Economics* 95 (2): 435–441.

Beckman, J., A. Borchers, and C.A. Jones. 2013. *Agriculture's Supply and Demand for Energy and Energy Products*. Washington, DC: U.S. Department of Agriculture, Economic Research Service, Economic Information Bulletin No. 112.

Bhaskar, A., and J.C. Beghin. 2009. How Coupled Are Decoupled Farm Payments? A Review of the Evidence. *Journal for Agricultural and Resource Economics* 34 (1): 130–153.

Borlaug, N. 2007. Feeding a hungry world. *Science* 318 (5849): 359.

Brandão, M., A. Levasseur, et al. 2013. Key Issues and Options in Accounting for Carbon Sequestration and Temporary Storage in Life Cycle Assessment and Carbon Footprinting. *The Journal of Life Cycle Assessment* 18 (1): 230–240.

Burney, J.A., S.J. Davis, and D.B. Lobell. 2010. Greenhouse Gas Mitigation by Agricultural Intensification. *Proceedings of the National Academy of Sciences* 107 (26): 12052–12057.

Cattaneo, A., D. Hellerstein, C. Nickerson, and C. Myers. 2006. *Balancing the Multiple Objectives in Conservation Programs*. Washington, DC: U.S. Department of Agriculture, Economic Research Service, Economic Research Report No. 19.

Chambers, A.S. 2011. Encouraging Carbon Sequestration on Private Agricultural Lands in the United States. Abstract #35. Presentation at the Air & Waste Management Association meeting, Greenhouse Gas Strategies in a Changing Climate, San Francisco, CA.

Choi, S., B. Sohngen, S. Rose, T. Hertel, and A. Golub. 2011. Total Factor Productivity Change in Agriculture and Emissions from Deforestation. *American Journal of Agricultural Economics* 93 (2): 349–355.

Claassen, R., F. Carriazo, J.C. Cooper, D. Hellerstein, and K. Ueda. 2011. *Grassland to Cropland Conversion in the Northern Plains: The Role of Crop Insurance, Commodity, and Disaster Programs.* Washington, DC: U.S. Department of Agriculture, Economic Research Service, Economic Research Report No. 120.

Claassen, R., E. Duquette, and J. Horowitz. 2013. Additionality in Agricultural Conservation Payment Programs. *Journal of Soil and Water Conservation* 68 (3): 74A–77A.

Coyle, W.T. 2010. *Next-Generation Biofuels: Near-Term Challenges and Implications for Agriculture.* Washington, DC: U.S. Department of Agriculture, Economic Research Service, Outlook Report No. (BIO-01-01).

Day-Rubenstein, K., P. Heisey, R. Shoemaker, J. Sullivan, and G. Frisvold. 2005. *Crop Genetic Resources: An Economic Appraisal*. Washington, DC: U.S. Department of Agriculture, Economic Research Service, Economic Information Bulletin No. 2.

Eagle, A.J., L.R. Henry, L.P. Olander, K. Haugen-Kozrya, N. Millar, and G.P. Robertson. 2012. *Greenhouse Gas Mitigation Potential of Agricultural Land management in the United States: A Synthesis of the Literature*. Nicholas Institute for Environmental Policy Solutions Report NIR 10-04, Third Edition. Durham, NC: Duke University.

Eagle, A.J., and S.D. Sifleet. 2011. *T-AGG Survey of Experts Scientific Certainty Associated with GHG Mitigation Potential of Agricultural Land Management Practices*. Nicholas Institute for Environmental Policy Solutions Report NIR 11-05. Durham, NC: Duke University.

Earles, J.M., S. Yeh, and K.E. Skog. 2012. Timing of Carbon Emissions from Global Forest Clearance. *Nature Climate Change* 2: 682–685 (September).

Follett, R.F., G.E. Varvel, J.M. Kimble, and K.P. Vogel 2009. No-Till Corn after Bromegrass: Effect on Soil Carbon and Soil Aggregates. *Agronomy Journal* 101 (2): 261–268.

Fuglie, K.O. 2012. Productivity Growth and Technology Capital in the Global Agricultural Economy. *In* Productivity Growth in Agriculture: An International Perspective, ed. K.O. Fuglie, S.L. Wang, and V. Eldon Ball. Wallingford, Oxfordshire, UK: CAB International, pp. 335–368.

Fuglie, K.O., P.W. Heisey, J.L. King, C.E. Pray, K. Day-Rubenstein, D. Schimmelpfennig, and S.L. Wang, and R. Karmarkar-Deshmukh. 2011. *Research Investments and Market Structure in the Food Processing, Agricultural Input, and Biofuel Industries Worldwide*. Washington, DC: U.S. Department of Agriculture, Economic Research Service, Economic Research Report No. 130.

Fuglie, K.O., J.M. MacDonald, and E. Ball. 2007. *Productivity Growth in U.S. Agriculture*. Washington, DC: U.S. Department of Agriculture, Economic Research Service, Economic Brief No. 9.

Fuglie, K.O., S.L. Wang, and V.E. Ball. 2012. Introduction to Productivity Growth in Agriculture. *In* Productivity Growth in Agriculture: An International Perspective, ed. K.O. Fuglie, S.L. Wang, and V.E. Ball. Wallingford, Oxfordshire, UK: CAB International, pp. 1–11.

Gardner, B., I. Hardie, and P.J. Parks. 2010. United States Farm Commodity Programs and Land Use. *American Journal of Agricultural Economics* 92 (3): 803–820.

Gilley, J.E., and J.W. Doran. 1997. Tillage Effects on Soil Erosion Potential and Soil Quality of a Former Conservation Reserve Program Site. *Journal of Soil and Water Conservation* 52 (3): 184–188.

Gleason, R.A., B.A. Tangen, and M.K. Laubhan. 2008. Carbon Sequestration. *In* Ecosystem Services Derived from Wetland Conservation Practices in the United States Prairie Pothole Region with an emphasis on the U.S. Department of Agriculture Conservation Reserve and Wetlands Reserve Programs, ed. R.A. Gleason, and M.K. Laubhan, and N.H. Euliss, Jr., Chapter C. Washington, DC: U.S. Department of the Interior, U.S. Geological Survey, Professional Paper 1745. http://www.fsa.usda.gov/Internet/FSA_File/pp1745.pdf

Goodwin, B.K., M. Vandeveer, and J. Deal. 2004. An Empirical Analysis of Acreage Distortions and Participation in the Federal Crop Insurance Program. *American Journal of Agricultural Economics* 86 (4): 1058–1077.

Haufler, J.B., (ed.) 2005. *Fish and Wildlife Benefits of Farm Bill Conservation Programs: 2000-2005 Update*. The Wildlife Society Technical Review 05-2.

Havlik, P., H. Valin, A. Mosnier, M. Obersteiner, J.S. Baker, M. Herrero, M.C. Rufino, and E. Schmid. 2013. Crop Productivity and the Global Livestock Sector: Implications for Land Use Change and Greenhouse Gas Emissions. *American Journal of Agricultural Economics* 95 (2): 442–448.

Heisey, P., S.L. Wang, and K. Fuglie. 2011. *Public Agricultural Research Spending and Future Agricultural Productivity Growth: Scenarios for 2010-2050.* Washington, DC: U.S. Department of Agriculture, Economic Research Service, Economic Bulletin No. 7.

Hellerstein, D., and S. Malcolm. 2011. *The Influence of Rising Commodity Prices on the Conservation Research Programs*. Washington, DC: U.S. Department of Agriculture, Economic Research Service, Economic Research Report No. 110.

Hertel, T.W. 2012. *Implications of Agricultural Productivity for Global Cropland Use and GHG Emissions: Borlaug vs. Jevons*, Working Paper 69, Global Trade Analysis Program, Purdue University.

Horowitz, J., and J. Gottlieb. 2010. *The Role of Agriculture in Reducing Greenhouse Gas Emissions.* Washington, DC: U.S. Department of Agriculture, Economic Research Service, Economic Bulletin No. 15.

Huffman, W.E., and R.E. Evenson. 2006. *Science for Agriculture: A Long Term Perspective*. Second edition. Ames, IA: Blackwell Publishing.

Intergovernmental Panel on Climate Change. 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Prepared by the National Greenhouse Gas Inventories Programme. Ed. H.S. Eggleston, L. Buendia, K. Miwa, T. Ngara, and K. Tanabe. Hayama, Japan: Institute for Global Environmental Strategies.

IPCC. 2007. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, 2007. Ed. B. Metz, O.R. Davidson, P.R. Bosch, R. Dave, and L.A. Meyer. Cambridge, UK, and New York, NY, USA: Cambridge University Press.

Jones, C.A., C. Nickerson, and N. Cavallaro. 2013. U.S. Policies and GHG Mitigation in the Agriculture Sector. *In* Land Use and the Carbon Cycle, ed. D.G. Brown, and D.T. Robinson, and B.C. Reed. Cambridge, UK, and New York, NY, USA: Cambridge University Press, pp. 403–430.

Jones, C.A., C. Nickerson, and M. Sperow. 2013. Greenhouse Gas Mitigation from the Conservation Reserve Program: The Contribution of Post-Contract Land Use Change. Selected Paper prepared for presentation at 2013 AAEA & CAES Joint Annual Meeting, Washington, DC. http://ageconsearch.umn.edu/bitstream/150501/2/Jones%20Nickerson%20Sperow%20CRP%20060313.pdf

Jones, C.A., and R.D. Sands. 2013. Impact of Future Agricultural Productivity Gains on Greenhouse Gas Emissions: a Global Analysis. *American Journal of Agricultural Economics*; doi: 10.1093/ajae/aat035.

Khanna, M., and D. Zilberman. 2012. Modeling the Land-Use and Greenhouse-Gas Implications of Biofuels. *Climate Change Economics* 3 (3): 1250016-1–1250016-15.

Kossoy, A., and P. Ambrois. 2012. *State and Trends of the Carbon Market 2012*. Washington, DC: World Bank.

Latta, G.S., D.M. Adams, R.J. Alig, E. White, and E. 2011. Simulated Effect of Mandatory versus Voluntary Forest Carbon Offset Markets in the United States. *Journal of Forest Economics* 17 (2): 127–141.

Leathers, N., and L.M.B. Harrington 2000. Effectiveness of Conservation Reserve Programs and Land "Slippage" in Southwestern Kansas. *Professional Geographer* 52 (1): 83–93.

Lewandrowski, J., M. Peters, C. Jones, R. House, M. Sperow, M. Eve, K. Paustian, and K. 2004. *Economics of Sequestering Carbon in the U.S. Agricultural Sector*. Washington, DC: U.S. Department of Agriculture, Economic Research Service, Technical Bulletin No. 1909.

Linn, J., and N. Richardson. 2013. Cutting Carbon, Take Two: A Brief Guide to Federal Electricity-Sector Climate Policy without Cap-and-Trade. *Applied Economic Perspectives and Policy* 35 (3): 377–397.

Lobell, D.B., U.R.C. Baldos, and T.W. Hertel. 2013. Climate Adaptation as Mitigation: The Case of Agricultural Investments. *Environmental Research Letters* 8 (1): 015012.

Lubowski, R.N., A.J. Plantinga, and R.N. Stavins. 2006. Land-Use Change and Carbon Sinks: Econometric Estimation of the Carbon Sequestration Supply Function. *Journal of Environmental Economics and Management* 51 (2): 135–152.

——. 2008. What Drives Land-Use Changes in the United States? A National Analysis of Landowner Decisions. *Land Economics* 84 (4): 529–550.

Mezzatesta, M., D.A. Newburn, and R.T. Woodward. 2013. Additionality and the Adoption of Farm Conservation Practices. *Land Economics*.

Monke, J., and R. Johnson. 2010. *Actual Farm Bill Spending and Cost Estimates*. Washington, DC: Congressional Research Service, R41195. Available at: http://cnie.org/NLE/CRS/abstract.cfm?NLEid=2272

Murray, B.C., B. Sohngen, and M.T. Ross. 2007. Economic Consequences of Consideration of Permanence, Leakage and Additionality for Soil Carbon Sequestration Projects. *Climatic Change* 80 (1–2): 127–143.

Nelson, E., S. Polasky, D.J. Lewis, A.J. Plantinga, E. Lonsdorf, D. White, D. Bael, and J.L. Lawler. 2008. Efficiency of Incentives to Jointly Increase Carbon Sequestration and Species Conservation on a Landscape. *Proceedings of the National Academy of Sciences* 105 (28): 9471–9476.

Nickerson, C., R. Ebel, A. Borchers, and F. Carriazo. 2011. *Major Uses of Land in the United States, 2007*. Washington, DC: U.S. Department of Agriculture, Economic Research Service, Economic Information Bulletin No. 89.

Nin Pratt, A., and S. Fan. 2010. R&D Investment in National and International Agricultural Research: An Ex-ante Analysis of Productivity and Poverty. Discussion Paper 00986, International Food Policy Research Institute.

Paustian, K., J.M. Antle, J. Sheehan, and E.A. Paul. 2006. *Agriculture's Role in Greenhouse Gas Mitigation*. Washington, DC: Pew Center on Global Climate Change.

Plantinga, A.J., and J-J Wu. 2003. Co-Benefits from Carbon Sequestration in Forests: Evaluating Reductions in Agricultural Externalities from an Afforestation Policy in Wisconsin. *Land Economics* 79 (1): 74–85.

Roberts, M.J., and S. Bucholtz. 2006. Slippage in the Conservation Reserve Program or Spurious Correlation? A Rejoinder. A*merican Journal of Agricultural Economics* 88 (2): 512–514.

Roberts, M.J., and R.N. Lubowski. 2007. Enduring Impacts of Land Retirement Policies: Evidence from the Conservation Reserve Program. *Land Economics* 83 (4): 516–538.

Schmit, T.M., J. Luo, and J.M. Conrad. 2011. Estimating the Influence of Ethanol Policy on Plant Investment Decisions: A Real Options Analysis with Two Stochastic Variables. Energy Economics 33 (6): 1194–1205.

Searchinger, T., R. Heimlich, R.A. Houghton, F. Dong, A. Elobeid, J. Fabiosa, S. Tokgoz, and D. Hayes, and T. Yu. 2008. Use of U.S. Croplands for Biofuels Increases Greenhouse Gases Through Emissions from Land-Use Change. *Science* 319: 1238–1240.

Smith, J.E., L.S. Heath, K.E. Skog, and R.A. Birdsey. 2006. *Methods for Calculating Forest Ecosystem and Harvested Carbon, with Standard Estimates for Forest Types of the United States*. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northeastern Research Station, General Technical Report NE-343.

Stevenson, J.R., N. Villoria, D. Byerlee, T. Kelley, and M. Maredia. 2013. Green Revolution Research Saved an Estimated 18 to 27 Million Hectares from Being Brought into Agricultural Production. *Proceedings of the National Academy of Sciences* 110 (21): 8363–8368.

U.S. Department of Agriculture, Economic Research Service. 2012a. *Agricultural Productivity in the United States: Overview*. http://www.ers.usda.gov/Data/AgProductivity/

——. 2012b. *Agricultural Research Funding in the Public and Private Sectors*. http://www.ers.usda.gov/data-products/agricultural-research-funding-in-the-public-and-private-sectors.aspx

U.S. Department of Agriculture, Farm Service Agency. 2008. *Conservation Reserve Program: Summary and Enrollment Statistics*, 2008. http://www.fsa.usda.gov/Internet/FSA_File/annualsummary2008.pdf

U.S. Department of Agriculture, Forest Service. 2013. *Forest Legacy Program Funded and Completed Projects*. http://www.fs.fed.us/spf/coop/programs/loa/flp_projects.shtml

U.S. Department of Agriculture, Natural Resources Conservation Service. 2008. *Farm Bill 2008 At a Glance: Wetlands Reserve Program*. http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs143_008151.pdf.

——. 2009. Protracts Database. Washington, DC.

——. 2013. Environmental Quality Incentives Program website. http://www.nrcs.usda.gov/wps/portal/nrcs/main/national/programs/financial/eqip/

U.S. Department of Agriculture, Office of Budget and Program Analysis. 2010. 2012 USDA Budget Explanatory Notes for Committee on Appropriations. Washington, DC. http://www.obpa.usda.gov/25nrcs2012notes.pdf

U.S. Department of Agriculture, Office of the Chief Economist. 2011. *USDA Agricultural and Forestry Greenhouse Gas Inventory: 1990-2008*. Washington, DC: Technical Bulletin No. 1930.

——. 2013. *Greenhouse Gas Mitigation Options and Costs for Agricultural Land and Animal Production with the United States*. Prepared by ICF International. Washington, DC: U.S. Department of Agriculture, Climate Change Program Office under USDA Contract No. AG-3142-P-10-0214.

U.S. Department of Energy, Energy Information Administration. 2009. *Energy Market and Economic Impacts of H.R. 2454, the American Clean Energy and Security Act of 2009*. Washington, DC: Report # SR-OIAF/2009-05.

——. 2007. *Annual Energy Outlook 2007, with Projections to 2030*. Washington, DC: DOE/EIA-0383(2007).

——. 2012. *Analysis of the Clean Energy Standard Act of 2012*. Washington, DC.

U.S. Environmental Protection Agency. 2009. EPA Analysis of the American Clean Energy and Security Act of 2009 H.R. 2454 in the *111th* Congress, including Appendix and Data Annex. http://www.epa.gov/climatechange/economics/economicanalyses.html

——. 2010a. *Renewable Fuel Standard Program (RFS2) Regulatory Impact Analysis*. Washington, DC: USEPA-420-R-10-006.

——. 2010b. *Regulation of Fuels and Fuel Additives: Modifications to Renewable Fuel Standard Program; Final Rule*. Federal Register Volume 75, Number 89, pp. 26025–26048.

——. 2011. *Biofuels and the Environment: First Triennial Report to Congress*. Washington, DC: EPA/600/R-10/183F.

——. 2013. *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2011*. Washington DC: USEPA-430-R-13-001.

Wang, S.L., D. Schimmelpfennig, and K. Fuglie. 2012. Is Agricultural Productivity Growth Slowing in Western Europe? In *Productivity Growth in Agriculture: An International Perspective*, ed. O. Keith Fuglie, Sun Ling Wang, and V. Eldon Ball, 109–125. Wallingford, Oxfordshire, UK: CAB International.

Wise, M., K. Calvin, A. Thomson, L. Clarke, E. Bond-Lamberty, R. Sands, S.J. Smith, A. Janetos, and J.A. Edmonds. 2009. Implications of Limiting CO_2 Concentration for Land Use and Energy. *Science* 324: 1183–1186.

Wu, J. 2000. Slippage Effects of the Conservation Reserve Program. *American Journal ofAgricultural Economics* 82 (4): 979–992.

——. 2005. Slippage Effects of the Conservation Reserve Program: Reply. *American Journal of Agricultural Economics* 87 (1): 251–254.

Zhang, L., B.K. Wylie, L. Ji, T.G. Gilmanov, and L.L. Tieszen. 2010. Climate-Driven Interannual Variability in Net Ecosystem Exchange in the Northern Great Plains Grasslands. *Rangeland Ecology and Management* 63 (1): 40–50.

Appendix

Table A1 Contributions to U.S. Greenhouse Mitigation from Proposed Cap and Trade and Clean Energy Programs

	Estimated annual mitigation, CO2e		CO_2e price
	Period	**Tg yr^{-1}**	**2008$**
Proposed Programs			
Multi-sector GHG cap & trade (H.R. 2454)[1]			
Agriculture offsets	2020-29	12	
Forestry offsets			
Afforestation	2020-29	58	
Forest management	2020-29	106	
Fossil fuel energy use (ag&f)	2020-29	4	
Bioelectricity (ag&f feedstocks)	2020-29	17	
Total	2020-29	**197**	\$17 mt^{-1} CO_2e
Agriculture offsets	2050-59	55	
Forestry offsets			
Afforestation	2050-59	410	
Forest management	2050-59	178	
Fossil fuel energy use (ag&f)	2050-59	22	
Bioelectricity (ag&f feedstocks)	2050-59	117	
Total	2050-59	**782**	\$72 mt^{-1} CO_2e
Clean Energy Standard (CES)	2025	na[3]	na[4]
(Bingaman 2012 proposal)[2]	2035	na[3]	na[4]
Current Programs			
Renewable Fuel Standard (RFS2)[5]	2022	**138**	na[4]

Notes: 1 million metric tons (mt) = 1 Teragram (Tg).
na = not available.
1. Source: U.S. EPA 2009. Mitigation from policies currently in place is included in the reference case (no cap-and-trade). Allowance price increases 5% per year over the period. By imposing a price on forestry and agriculture emissions, it effectively models a mandatory program. See text discussion for caveats regarding use of these estimates to infer potential mitigation from a voluntary offset program.
2. Source: U.S. DOE EIA 2012. Bioelectricity beyond reference levels generated through the CES is about 25% less than for H.R. 2454 GHG cap and trade policy in 2025, and divergence widens for 2035 (based on cap-and-trade analysis in U.S. DOE EIA 2009).
3. EIA does not estimate life cycle emissions.
4. Costs are borne privately; no estimate is available.
5. Full mandate level of blending is scheduled for 2022.

Table A2 Contributions to U.S. Greenhouse Mitigation from Current USDA Conservation Programs[1]

	Estimated annual mitigation, CO_2e		Effective CO_2e price; program costs	Hectares enrolled
	Period	Tg yr^{-1}	2008$	mi. ha.
Current Programs				
Conservation Reserve Program[2]				
Grassland soil	1997-08	16	$55/mt[3]	12.4
Afforestation	1999-08	15	$14/mt[3]	1.5
Wetlands	1997-08	[4]		0.9
Reduced fuel, fertilizer (onfarm & upstream)	1997-08	9		
Total		~39	$1.9 billion	14.8
Wetlands Reserve Program[4]				
Wetland restoration	2008		na	0.8
Grassland Reserve Program[5,7]				
Grassland preservation	2007	0.007	na	
	2010	0.033	na	
Forest Legacy Program				
Permanent forest easements	Jan. 2012		na	0.9[6]
Working Lands Programs - EQIP, CStP, WHIP, Conservation Technical Assistance[7]				
Includes improved livestock practices, conservation tillage (no-till), prescribed grazing, upland wildlife habitat management, etc.	2010	12.5	$.943 billion	

Notes:

1. Estimates do not take into account whether GHG benefits are additional or permanent, or if leakage has occurred. Costs reported for conservation programs with term contracts (CRP, working lands) are annual program costs for 2008.

2. For 2008-2011, US EPA (2013) estimates cumulative releases due to declining enrolled ha. as 10.7 Tg CO2e; but does not report separately estimated lower annual sequestration on enrolled lands relative to 1997-2008. Sources: Grassland soil: USDA OCE, 2011, p. 61; Afforestation: Supplementary Data (supplied by S. Hyberg), USDA FSA 2008; Reduced fuel and fertilizer, hectares enrolled and program costs: USDA FSA 2008.

3. This calculation attributes all costs of this multi-objective program to GHG mitigation (Horowitz and Gottlieb 2010).

4. We do not report an estimate of wetland GHG mitigation, based on (Eagle and Sifleet 2011), who found that the direction of net GHG impacts varies by soil type, past uses and wetland types and on Gleason et al. 2008, who did not find statistically significant increases in carbon stocks associated with WRP projects. Source for enrolled ha: USDA NRCS 2008.

5. Of enrolled lands, 85% are in permanent easements and the remainder are in contracts of up to 30 years (OBPA 2011).

6. Cumulative total for permanent easements (which take forests out of production, and prevent conversion out of forest use). Source: USDA USFS 2012.

7. Source for mitigation estimates: Chambers 2011 (data on mitigation benefits from current contracts compiled by Greg Zwicke and Greg Johnson).

Table A3 Contributions to U.S. Greenhouse Mitigation from Public and Private R&D Programs

	Estimated annual mitigation, CO_2e		Effective CO_2e price; program costs
	Period	**Tg yr^{-1}**	**2008$**
Past Investments			
Average annual U.S. expenditures on productivity-oriented public (federal and state) and private R&D.[1]	1961-2005	na[3]	$4.2 billion
Annual U.S. expenditures on productivity-oriented public (federal and state) and private R&D.[1]	2008	na[3]	$2.5 billion (public) + $4.0 billion (private)
Potential Future Investments			
Simulation scenario: U.S. makes sufficient R&D investments to increase U.S. productivity enough to offset climate change productivity shocks.[2]	2005-2050	na[3]	$25 mt^{-1}

Notes: na = not available.
1. Source: Authors' calculations of R&D expenditures are based on USDA ERS 2012b, Fuglie et al. 2011, and Alston et al. 2010.
2. Source: Lobell, Baldos, and Hertel 2013; T.W. Hertel, personal communication, April 2013. This calculation attributes all R&D costs for improving productivity to GHG mitigation.
3. See discussion in text.