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Greenhouse Gas Mitigation Opportunities in California Agriculture

Review of the Economics

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Greenhouse Gas Mitigation Opportunities in California Agriculture: Review of the Economics
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ABSTRACT

Although about three-quarters of California farm revenue derives from crop production, crops—mainly tree, vine, and vegetable crops—account for only about one-quarter of GHG emissions. Some studies indicate minimal yield loss from reducing nitrogen fertilizer use, and simulation results show significant percentage reductions in GHG emissions for payments of \$20/MTCO_2e. The economics of reducing emissions from enteric fermentation has been little studied. Manure management to reduce GHG emissions (mainly methane) can be as simple as covering manure lagoons and flaring methane. The more complex option of using manure-generated methane gas to replace fossil fuels has been investigated often. Most case studies and simulations suggest this option is costly. Its economic feasibility depends on specific local conditions, but there is no evidence of large-scale feasibility in California without large subsidies.

Acknowledgments

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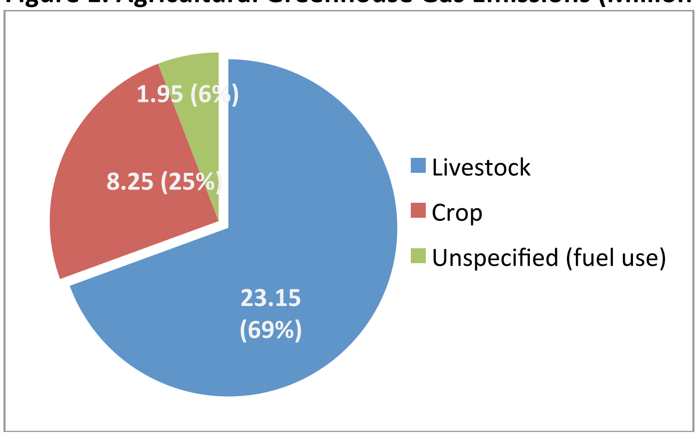
CONTENTS

INTRODUCTION

Farming generates each of the three main greenhouse gases: methane (CH_4), nitrous oxide, (N_2O), and carbon dioxide (CO_2). Major documented agricultural sources of GHG emissions in California include enteric fermentation and manure management in livestock operations, agricultural soil management and fertilizer use, rice cultivation, burning of agricultural residues, and on-farm energy use. Biological processes and input use on farms generate mostly CH_4 and N_2O, which are more potent greenhouse gases per ton than CO_2; CH_4 is 25 times and N_2O is 298 times more potent in carbon equivalent unit (ARB 2011; IPCC 2007). On-farm energy use generates mostly CO_2 emissions but accounts for a small portion of total agricultural emissions in California.

Farm-level greenhouse gas (GHG) emissions from California agriculture accounted for an estimated 7% of the state's total GHG emissions in 2009. On a carbon dioxide equivalent (CO_2e) basis, the California Air Resources Board (CARB) estimates that California farms generated about 33 million metric tons of CO_2e (CARB 2011). Of this total, 25% was from crop production and 69%, from livestock production (Figure 1). Farming accounts for about 1% of the state's gross state product, suggesting that it is more intensive than many other industries in GHG emissions per job or per unit of economic activity.

Figure 1. Agricultural Greenhouse Gas Emissions (Million MT CO_2e) by Source, 2009



Source: CARB (2011).

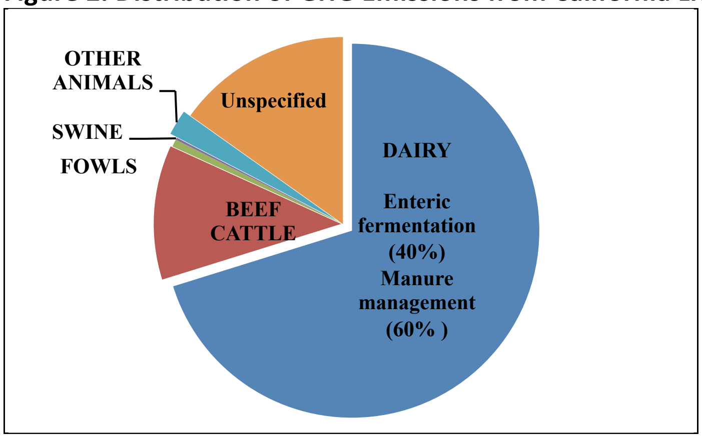
This report reviews and evaluates the current literature on the economics of potential (GHG) emissions mitigation strategies for California agriculture. The report's scope is modest in the sense that it focuses on the subset of GHG mitigation options that would have maximum impact given the current California agriculture practices and the feasibility of applying existing mitigation technologies. Further, this report, organized into crop and livestock parts, focuses mostly on annual crops and dairy. Even though perennial orchard crops are a major part of California crop agriculture, our discussion is limited only to annual crops because little research has been done on perennial crops.

Most attention is devoted to dairy emissions, which comprise about half of total agricultural emissions in California. California has more dairy cows and a larger dairy industry than any other state. Dairy represents about 60% of livestock revenues in California and 70% of GHG emissions from livestock (CDFA 2013) (Figure 2).

The two major sources of dairy emissions are enteric fermentation, accounting for 40% of the total, and manure management, accounting for 60% of the total (Figure 2). This report focuses on mitigation of GHG emissions from manure management, with an emphasis on anaerobic dairy digesters. This emphasis stems from the large and growing literature on digester economics, which is readily applicable to dairy farms in California, and from policy interest in digesters. This interest dates from December 2009, when the U.S. Secretary of Agriculture announced an agreement to reduce GHG emissions from dairy operations by 25% before 2020. As an approach for meeting this goal, anaerobic digestion has garnered the most attention (U.S. Department of Agriculture 2009).

Early studies on digester economics tended to focus on farm-level case studies in the East or Midwest dairy states, but more recent studies include aggregate, national-level analyses with a broader regional focus. The number of studies on California dairies specifically is also growing. Many have been facilitated by public or quasi-public state institutions for purpose of project or policy assessment.

Figure 2. Distribution of GHG Emissions from California Livestock Agriculture by Livestock Type, 2009



Source: ARB (2011).
Note: Unspecified is mostly fuel use and nitrogen in managed or unmanaged manure.

A huge literature investigates the potential for reducing GHG emissions through carbon sequestration in crops or forestry. Studies on carbon sequestration are mostly on a national or global scale, and their agricultural focus is on dryland field crops or forestland that have limited direct relevance for cropping systems in California, where irrigated crops including many tree, vine and vegetable crops predominate. California has less than 0.4 million acres of dryland crops compared, for example to about 24 million acres in Iowa, even though the value of crop production is much higher in California.

According to California GHG inventory data published by the Air Resource Board, soil management accounts for 68% of crop emissions, followed by 18% for agricultural residue burning, 8% for energy use, and 7% for rice production (ARB 2011). Within soil management, fertilizer nitrogen accounts for most emissions; application of synthetic fertilizer alone accounts for 60% of all emissions from crop agriculture (ARB 2011). Rosenstock, Liptzin, Six, and Tomich (2013) attempt to document fertilizer rates for California crops, but no hard data are available. The relationship between application of nitrogen fertilizer and GHG emissions is complex, depending on factors such as climate, soil type, crop, soil moisture content, and crop rotation (De Gryze, Catala-Luque, Howitt, and Six 2009b).

To consider economic relationships, the GHG emissions reduction potential of alternative soil management practices must be evaluated in the context of effects on crop yield and costs (Pendell et al. 2007). Studies of the economics of GHG mitigation in crop agriculture are relatively few, and even fewer are those that consider tree and vine crops or vegetables. This discussion is therefore limited to field crops, with the exception of processed tomatoes.

It begins with the economics of major GHG mitigation options in crop production and draws mostly on a recent report by ICF International (2013) that calculates these options' economic gains and losses in the absence of incentive schemes. Discussion of incentive payments for GHG mitigation, which entail changes in crop mix, references two papers—Mérel, Yi, Lee, and Six (2013) and Garnache, Mérel, Lee, and Six (2013)—that focus on how these payments would affect field crop emissions in California. Using crop growth simulation models for field crops in California as inputs into a positive mathematical programming (PMP) optimization model, these studies calculate how baseline emissions values would change if specific GHG policies were adopted. Related work on cropping conditions, important nationally, appeared much earlier in McCarl and Schneider (2001) and Murray et al. (2005).

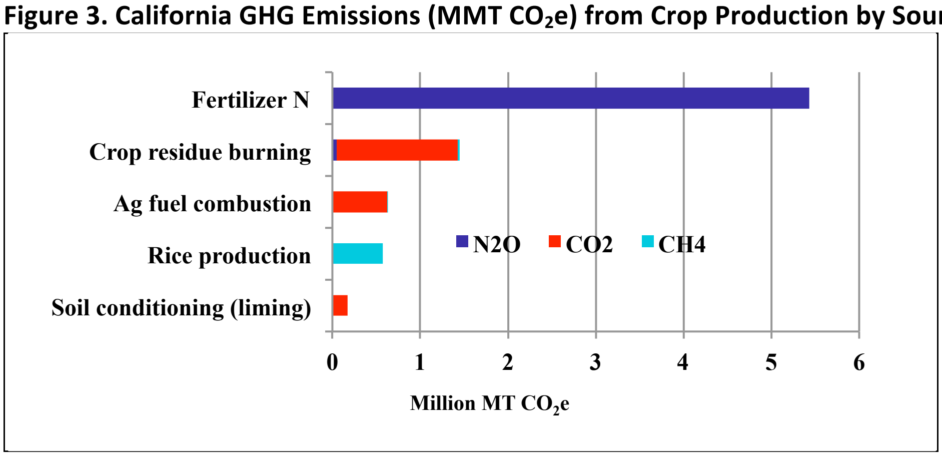
ECONOMICS OF GHG MITIGATION IN CALIFORNIA CROP AGRICULTURE

In crop agriculture, greenhouse gases are generated from soil management activities, including fertilizer N application or soil disturbance (tilling), crop residue burning, rice production, and energy use related to crop production. GHG emissions can be mitigated by reducing emissions from some practice or activity, adopting alternative practices with fewer GHG emissions, or shifting land use mix. Modified or alternative production practices or shifts in land use likely entail changes in output and production costs, which result in changes in net economic returns. GHG-mitigating strategies are often related to government policies that provide incentives for mitigation. Economists estimate the minimum compensation needed for a farm to adopt GHG mitigation options.

Overview of GHG Mitigation Practices in Crop Production

Quantification of GHG emissions from crop agriculture is essential to identifying mitigation practices. According to California GHG inventory data published by the California Air Resources Board (CARB), GHG emissions related to crop agriculture in 2009 totaled 8.25 million metric tons of CO_2 equivalent ($MMTCO_2e$). Of this total, nitrogen fertilizer accounts for 5.4 $MMTCO_2e$, followed by 1.5 $MMTCO_2e$ from agriculture residue burning, 0.6 $MMTCO_2e$ from energy use, 0.6 $MMTCO_2e$ from rice production, and 0.2 $MMTCO_2e$ from liming (Figure 3).[1]

Figure 3. California GHG Emissions (MMT CO_2e) from Crop Production by Source and Gas, 2009



Adding nitrogen to the soil stimulates the production of nitrous oxide (N_2O). Disturbing the soil releases stored carbon into the atmosphere. California's 2009 GHG inventory data indicate that application of nitrogen fertilizer is the most significant contributor of GHG emissions from crop agriculture. Of the 5.4

[1] CARB data do not include a specific category for tillage, which varies by crop and plot and in intensity. The exclusion of tillage may be due to lack of information.

MMT CO_2e related to nitrogen fertilization, 93% is due to synthetic nitrogen fertilizer. On average, approximately 50% of the nitrogen fertilizer applied in the field is lost through volatilization, leaching, and runoff (Burger and Horwath 2012). How much reducing nitrogen fertilization mitigates GHG emissions depends on factors such as climate, soil type, crop, and crop rotation (De Gryze, Catala-Luque, Howitt, and Six 2009b).

Greenhouse gas emissions can be mitigated by simply reducing their sources, such as fertilizer use. However, insufficient application of nitrogen can reduce yields and net farm income. A large amount of research based on field experiments or simulation methods has been devoted to generating scientific information about biophysical relationships among fertilizer applications, GHG emissions, and yields (Zhu, Burger, Doane, and Horwath 2013; Burger and Horwath 2012; De Gryze et al. 2009a; De Gryze, Catala-Luque, Howitt, and Six 2009b). The GHG emissions reduction potential of fertilizer use must be evaluated in the context of its yield relationship (Pendell et al. 2007).[2]

Changes in tillage can significantly affect soil carbon storage, mostly by changing the rate of residue decomposition and carbon loss from the soil (ICF International 2013). Studies indicate that the GHG emissions reduction from reduced tillage depends on crops and environmental conditions (such as soil type, climate) (De Gryze et al. 2009a). The relationship of soil N_2O emissions to tillage regime is not clear (ICF International 2013). Reduced tillage also leads to reduced fossil fuel combustion, resulting from fewer or less intense field operations, and hence to reduced GHG emissions.

Tillage may also affect yields. Even though reduced tillage is in general believed to reduce yields and GHG emissions (Heimlich 2003), the opposite effect is possible (Ogle, Swan, and Paustian 2012). Changes in yield under different tillage regimes also depend on crop type, soil, and climate conditions.

Management options for limiting per-acre CH_4 emissions from rice include water, crop residue, and nutrient management and, potentially, cultivation of improved rice cultivars. The GHG mitigation potential is not well understood for most of these management practices. For example, draining rice fields during the growing season has been shown to decrease CH_4 emissions, but in certain regions with high soil carbon, N_2O emissions rose significantly following drainage (Li, Frolking, and Butterbach-Bahl 2005). Water management during the non-growing season can also affect gaseous flux. With two years of monitoring, Fitzgerald, Scow, and Hill (2000) find that winter flooding increased annual CH_4 emissions from California rice fields.

Residue burning is used by farmers to control diseases, weeds, and insects as well as to reduce the need for tillage, suggesting that these production benefits should be taken into consideration when assessing farmers' response to any GHG mitigation incentive (ICF International 2013). In California, field burning used to be a common practice to dispose of rice straw. With the mandated phase-out of rice straw burning starting in 1992, this practice was reduced to 25% of total rice acreage by 2001. In 2009, only about 12% of rice fields were burned (Garnache, Rosen-Molina, and Sumner 2010). Currently, burning of tree crops remains a significant source of GHG emissions.

Recent studies have considered the effects on GHG emissions and crop yields of combining multiple GHG mitigation practices. De Gryze, Catala-Luque, Howitt, and Six (2009b) found that each individual practice produced modest emissions reductions but that combining practices produced larger reductions. Among soil management practices, fertilization influences yields most. However, in their experiment

[2] In addition, a crop that is under-fertilized and thus under-developed may also leave residual N in soil, which leads to the potential GHG releases. This consideration stresses the importance of identifying a fertilization rate reflecting yield impacts and the N uptake by the crop (Burger and Horwath 2012).

including seven California crops (alfalfa, corn, rice, safflower, sunflower, tomato and wheat), De Gryze, Catala-Luque, Howitt, and Six (2009b) found yield responses to fertilizer reductions were small.

Pricing of Major Mitigation Options in the Absence of Land Use Change

Most work focuses on generating scientific experimental information (Zhu, Burger, Doane, and Horwath 2013; Burger and Horwath 2012; De Gryze et al. 2009a; De Gryze, Catala-Luque, Howitt, and Six 2009b). Few studies incorporate economic assessment (Antle and Ogle 2012; ICF International 2013). The quantitative summary provided here is primarily drawn from a recent study by ICF International (2013). The selected measures, which vary depending on climatic, agronomic, and farm-specific conditions, allow cost comparisons of major mitigation options.

ICF International (2013) presents calculations of the costs of GHG reductions associated with various mitigation options. These costs also represent marginal incentive prices at which farmers would be willing to trade for one unit of GHG mitigation as a result of adopting GHG mitigation options. Three mitigation options considered here are a 10% reduction of fertilizer use, fertilizer reductions through variable rate technology, and changes in tillage, from conventional to no tillage and from reduced tillage to no tillage. This discussion focuses only on the Pacific region (including California, Oregon, and Washington) and excludes soybeans and sorghum, which are very minor crops in California.

Table 1 displays incentive prices per ton of CO_2e mitigation. Given the wide variance in GHG emissions related to fertilizer use, the first two fertilizer options are differentiated with high and low emissions scenarios. In the first option, incentive prices are not presented, except for wheat under the high emissions scenario, because the assumed 10% reduction in fertilizer application resulted in insignificant emissions reductions or yield losses. Under the variable rate technology, fertilizer reductions are not assumed to cause yield losses, because this technology allows farmers to precisely adjust the rate of fertilizer application.[3] However, the equipment to use variable rate technology has a capital cost of $22,000, implying that per-acre cost of this technology declines with farm size. The cases of two farm sizes are presented here. Finally, the tillage options do not include the switch from conventional to reduced tillage, which yields insignificant emissions reductions.

Table 1. Cost of GHG Mitigation ($/MTCO$_2$e) in Pacific Region on Selected Crops by Mitigation Option

	10% simple reduction in nitrogen		Nitrogen reductions under variable rate technology 1000 acres		550 acres		Conventional tillage to no tillage	Reduced tillage to no tillage
Emission	High	Low	High	Low	High	Low	No emissions scenario assumed	
Corn	n/e	n/e	<0	<0	<0	<0	20	16
Cotton	n/e	n/e	n/e		n/e		1,178	542
Wheat	2	n/e	13	30	50	113	106	63

Source: ICF International (2013).
Note: n/e = not estimated; none of these options caused significant emission reductions.

Table 1 indicates that under the option of simple reduction of fertilizer, the reduction has to be greater than 10% to achieve any significant emissions reduction; the only exception to tis finding is wheat under the high emissions scenario. Under the variable rate technology option, carbon prices are negative for corn farms, implying that their savings from fertilizer reductions outweigh their technology adoption

[3] The ICF study considered only corn and wheat; assumed fertilizer reductions are 21% for corn and 10% for wheat.

costs. Variable rate technology is already practiced relatively widely in the Pacific region (ICF International 2013). Reduced tillage or no-till options are found to be expensive, especially for cotton. According to Mitchell, Klonsky, and Shrestha (2007), conventional tillage dominates California agriculture, accounting for about 98% of California annual crop acreage in 2003. Cotton has been a tillage-intensive crop in California (Mitchell et al. 2012). Even though recent data on tillage are not available, such a high share of intensely tilled acreage is consistent with the high carbon prices in Table 1. Given substantial acreage of cotton and wheat in California (534,000 and 454,000 acres, respectively, in 2012), tillage options for cotton and wheat could offer large mitigation potential, but at a prohibitively high cost. Based on per-acre emissions data used by ICF (2013), the maximum mitigation potential in California from tillage options is 48,000 $MTCO_2e$ for cotton and 41,000 $MTCO_2e$ for wheat (at 0.08 $MTCO_2e$/acre). Mitigation from reduced tillage of corn is available at much lower prices.

Policy-Induced GHG Mitigation in California Crop Agriculture

Two closely related studies examine GHG abatement potential in California crops. Mérel, Yi, Lee, and Six (2013) analyze how a tax on fertilizer application affects N losses through both N leaching and N_2O emissions in California's Yolo County. Garnache, Mérel, Lee, and Six (2013) examine the effects of a carbon credit on GHG emissions generated from crop production in California's Central Valley.

These are the only recent California studies that attempt to examine economic implications for GHG mitigation policies in crop agriculture. Both use the same basic approach and have many features in common. As with any simulation model, results depend crucially on approach and assumptions, which are examined here in some detail. Technical issues are not considered here; instead, the focus is on empirical application.

Both Mérel, Yi, Lee, and Six (2013) and Garnache, Mérel, Lee, and Six (2013), which is based on Garnache (2013), use a mathematical programming optimization method to assess the effects of hypothetical environmental policies—a tax on N fertilizer or a tax on carbon emissions—on agricultural production and the environment. The two studies examine adjustments in input use and area allocation across crops. These studies find that GHG mitigation incentives (taxes or subsidies) can lead to sizable reductions in GHG emissions from crop agriculture using data calibrated to California conditions. Garnache, Mérel, Lee, and Six (2013) further finds that policies targeting a single input, such as nitrogen fertilizer, can lead to a higher cost per unit of emissions abatement than using an array of policies.

Both papers model regional field crop choice derived from the maximization of aggregate farm net returns, given input and crop prices. This work is subject to a land constraint in Mérel, Yi, Lee, and Six (2013) and to both a land constraint and a water constraint in Garnache, Mérel, Lee, and Six (2013). That is, Garnache, Mérel, Lee, and Six (2013) assume that land and irrigation water used for the studied crops is fixed, and the model allocates resources among the crops accordingly.

The programming model used in both studies is calibrated with economic and agronomic information using positive mathematical programming (PMP). Agronomic information used in Mérel, Yi, Lee, and Six (2013) and Garnache, Mérel, Lee, and Six et al. (2013) is generated by a biophysical model—the DAYCENT model. Given that the DAYCENT model is not calibrated for tree or vine or for most vegetable crops, both studies focus only on field crops (including processing tomatoes).[4]

[4] De Gryze et al. (2009b) provide more details about the DAYCENT model. This model calculates yield responses to irrigation water, nitrogen fertilizer, and tillage practices. Data from the DAYCENT model show little response to tillage for all crops except corn and processing tomatoes.

Mérel, Yi, Lee, and Six (2013) considers alfalfa, corn, irrigated pasture, rice, safflower, sunflower, processing tomatoes, and wheat, which together represent about 51% of the total agricultural land in Yolo County. Mérel, Yi, Lee, and Six (2013) include essentially all the irrigated land over which farmers allocate annual crops in Yolo County. Cropping activities analyzed in Garnache, Mérel, Lee, and Six (2013) include alfalfa, corn, cotton, wheat, sunflower, processing tomatoes, and safflower, but not rice or irrigated pasture. Wheat and sunflower are used to represent disparate areas of bundles of crops assumed to be similar in responses to incentives. The crop area covered in Garnache, Mérel, Lee, and Six (2013) represents 70% of the non-perennial crop area in the Central Valley, but a much smaller percentage of total crop area or total agricultural area in the diverse Central Valley.

The base period used in both studies for their model calibrations is around 2005. Agricultural technology and economic conditions have changed since then. For example, drip irrigation for production of processing tomatoes has expanded rapidly. Grain and oilseed prices more than doubled after 2007 and remain more than double, in real terms, compared with those in 2005. Fertilizer prices rose rapidly with farm commodity prices.

Mérel et al. (2013) simulate the effects of nitrogen taxes ranging from 4 cents to 16 cents per pound of nitrogen. The highest tax considered is equivalent to a 50% increase in the price of urea during the calibration period (2002–2008). Their simulation output includes crop-specific changes in behavioral and environmental variables such as the intensity of fertilizer use, area planted to each crop, and nitrate oxide emissions. Effects are decomposed into those achieved by changes only in input use holding crop areas constant and those achieved by changes in crop mix holding the intensity of input use constant.

Simulation results from Mérel, Yi, Lee, and Six (2013) indicate that all crops experience a reduction in fertilizer and water application intensities due to the imposition of a fertilizer tax. Fertilizer application intensities tend to decline less with high revenue per acre crops, while the crop mix tends to shift toward crops that use less nitrogen per unit of land. Mérel, Yi, Lee, and Six (2013) also find that nitrate-leaching effect is important; the relationship between nitrate leaching and fertilizer use varies significantly by crop.

Table 2 summarizes environmental effects—those that occur by changes in input use (intensive margin) and those due to changes in crop mix (extensive margin). Intensive margin effects dominate in both nitrate leaching and N_2O flux, implying that the bulk of environmental effects are achieved by changes in fertilizer and irrigation water use rather than changes in crop mix. The bottom line of Table 2 shows that a 50% increase in nitrogen fertilizer prices would achieve less than a 6% reduction in N_2O flux.

Table 2. Environmental Effects Induced by Fertilizer Taxes: Aggregate, Intensive Margin, and Extensive Margin Effects

	Nitrate leaching (base = 4,573 ton N/yr)			N_2O flux (base = 318 ton N/yr)		
	% change from the base case					
Scenarios	Total change	Intensive margin	Extensive margin	Total change	Intensive margin	Extensive margin
¢4/lb. N	−3.78	-3.32	−0.46	−1.88	-1.53	−0.35
¢8/lb. N	−6.01	-5.11	−0.90	−3.30	-2.61	−0.69
¢12/lb. N	−7.78	-6.47	−1.31	−4.51	-3.48	−1.03
¢16/lb. N	−9.42	-7.72	−1.70	−5.70	-4.34	−1.36

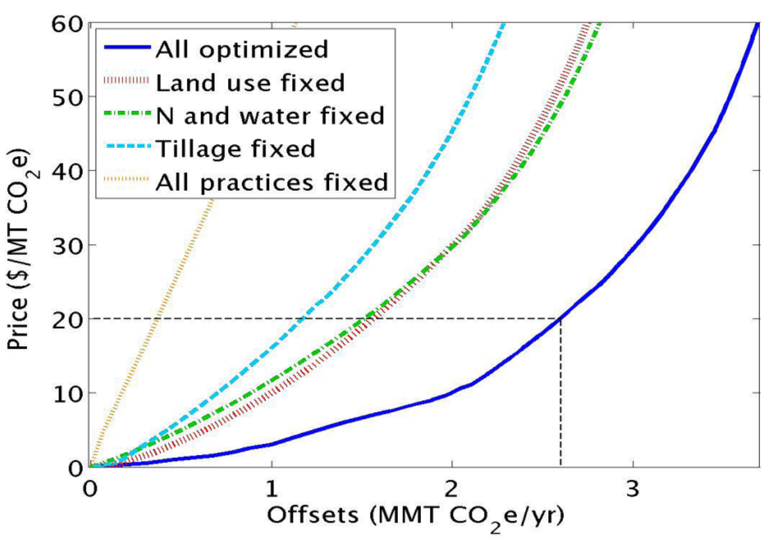
Source: Mérel, Yi, Lee, and Six (2013).

Garnache, Mérel, Lee, and Six (2013) simulate marginal cost curves for GHG emissions abatement for Central Valley field crops. The marginal abatement cost curve depicts the quantity of GHG offsets that would be supplied at various offset prices. Simulation results for the \$20/MTCO$_2$e scenario indicate that the modeled cropping activities would supply 2.6 million metric tons of carbon offsets—a 58% reduction in GHG emissions. In the Garnache, Mérel, Lee, and Six (2013) model, emissions reductions come from changes in fertilizer use, irrigation water use, tillage, and crop mix. GHG abatements are from N_2O emissions reduction and increased soil carbon sequestration. The researchers find that N_2O emissions reduction constitutes about 60% of the total abatement at \$20/MTCO$_2$e.[5] Figure 4 presents the decomposition of carbon offset supply attributable to input use and crop mix changes. Consistent with the results obtained by Mérel, Yi, Lee, and Six (2013), the majority of emissions reductions are due to changes in input use rather than changes in crop mix.

Both Mérel, Yi, Lee, and Six (2013) and Garnache, Mérel, Lee, and Six (2013) show the relative importance of changes in input use to GHG mitigation. Mérel Yi, Lee, and Six (2013) suggested that the efficiency of a GHG mitigation policy would be improved by incorporating emissions effects of interactions between fertilizer use and water use.

Mérel, Yi, Lee, and Six (2013) and Garnache, Mérel, Lee, and Six (2013) assume exogenous output prices that are not affected by local production. This simplification raises concerns for alfalfa hay and corn silage that have local markets and for processing tomatoes for which Central Valley output is a large share of world production and exports. This exogenous output price assumption affects allocation of area across crops and input intensity. Ignoring the market equilibrium price effect in this case may lead to an overestimated GHG abatement for crops for which the demand function is not perfectly elastic. Both studies consider only some of the annual crops in the Central Valley. Therefore, crop substitution outside the set of crops represented is ignored. Shifts to tree and vine crops, irrigated pasture, and vegetables and melons are all relevant options.

Figure 4. Decomposition of Carbon Offset Supply Curves under Four Scenarios in Which Land Use and Either or Both Tillage Intensity and N Fertilizer and Water Application Rates Are Restricted to Their Baseline Levels



Source: Garnache, Mérel, Lee, and Six (2013).

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[5] These results may be mostly dictated by study frameworks, including region and crop mix. McCarl and Schneider (2001) found that most emissions reductions in U.S. agriculture were from increased soil carbon sequestration.

Final Remarks on the Economics of GHG Mitigation for Crop Agriculture in California

Beyond issues that arise specifically in the two studies examined in detail, several points are worth reinforcing.
First, fertilizer nitrogen inputs are major contributors to field crop GHG emissions in California. A permanent reduction in nitrogen application per unit of land would likely lead to permanent reductions in NO emissions per unit of land. But depending on the relationship between fertilizer and crop yield per hectare, the impact on GHG emissions per unit of output or per value of output is less obvious. Research on actual rather than simulated farmer behavior could be a significant contribution. Nitrogen application rates actually applied on farms for specific crops are not well documented. Data on the distribution of farm fertilizer practices would be more enlightening than recommendations, benchmarks, or economic simulations that assume the accuracy of hypothetical crop simulation models. Similarly, useful research would develop statistical evidence on responses of fertilizer application rates to changes in relative prices and other incentives. Such empirical research is lacking for the irrigated cropping conditions in California.

Second, the effects of some GHG mitigation practices on GHG emissions and agricultural production are not well understood. For example, even though straw incorporation leads to an increase in CH_4 emissions compared to burning, it may sequester soil carbon and preserve soil nitrogen (Linquist, Brouder, and Hill 2006). Rice in particular has complex interactions with methane as well as carbon and nitrogen important to GHG emissions. Further examination is required to understand the tradeoffs of residue burning on disease and other factors in costs and crop yields (ICF International 2013).

Third, in addition to the effects of GHG mitigation practices on emissions and agricultural production, other economic and behavioral factors should be reflected in the design of GHG mitigation incentives. For example, switching from conventional to no tillage involves changes in capital, labor, and fuel costs (ICF International 2013). Antle and Ogle (2012) find that adoption of conservation tillage influences changes in soil NO emissions and CO_2 emissions attributed to changes in fuel use. Six et al. (2004) had previously suggested that no-till adoption influences not only soil carbon stocks, but also NO emissions and fuel-related GHG emissions. The simulations in McCarl and Schneider (2001) and Antle and Ogle (2012) indicate that recognition of NO emissions and CO_2 reduction substantially shifts GHG offset supply curves. These results suggest that subsidies from government programs or market-based emissions trading based on soil carbon alone will not accurately reflect changes in GHG emissions or the true opportunity cost of supplying offsets.

Fourth, as indicated by Cacho, Wise, and MacDicken (2004) and Mooney, Antle, Capalbo, and Paustian (2004), incorporating tillage-based GHG emissions reduction in GHG emissions policy likely entails high transactions costs and much imprecision. Like the cost and difficulty of accounting for soil carbon, the issues of permanence and additionality complicate the effectiveness of carbon offset policy. For any emissions reduction to be permanent, GHG mitigation practices under which carbon offsets are supplied must remain in place forever. The requirement for permanence presents obvious difficulties for policy implementation. To minimize the problems associated with additionality, well-developed baseline definitions and measurement are needed (Gramig 2011; Murray, Sohngen, and Ross 2007).

ECONOMICS OF GHG MITIGATION IN LIVESTOCK PRODUCTION IN CALIFORNIA

Greenhouse gases produced from livestock production, mainly methane, are derived from two primary sources—manure management and enteric fermentation. Manure management is an issue for all animals, whereas enteric emissions concern only ruminants. In California, where the dairy industry accounts for the lion's share of livestock agriculture, both enteric fermentation and manure management are significant GHG emissions sources.

Strategies to mitigate manure-related GHG emissions include converting liquid manure systems to solid systems and converting anaerobic lagoons to anaerobic digesters (Owen, Kebreab, and Silver 2013).[6] Emissions from anaerobic lagoons have more than 10 times the global warming potential of emissions from solid manure piles (Owen, Kebreab, and Silver 2013). This paper's focus on anaerobic dairy digesters stems from several California-specific conditions. First, most livestock revenue is from dairy, and dairy's share of emissions is even higher than its revenue share. Second, substantial research informs understanding of manure-related issues. Third, researchers have highlighted the GHG mitigation potential of anaerobic lagoons—the most GHG intensive and most typical manure management system in California.

Digesters have also garnered wide attention from policy makers as an important GHG mitigation strategy in agriculture. In December 2009, the U.S. Secretary of Agriculture announced an agreement to reduce GHG emissions from dairy operations by 25% before 2020. It cited anaerobic digestion as the primary method for meeting this goal (USDA 2009). Finally, CARB has identified digesters as having the largest potential to reduce agriculturally related GHG emissions in California.

Basic anaerobic digester technology is old, but economic studies on dairy digesters are relatively new. This literature has accumulated rapidly over about a decade, spurred by rising concern over GHG emissions and interest in renewable energy. Reviewed below are peer-reviewed literature and quasi–peer-reviewed reports published or coordinated by public institutions. Government reports are mostly California-specific studies, which provide in-depth details of the covered digester projects. Starting with a brief discussion on enteric emissions, the report proceeds to an examination of anaerobic digestion, first with a broad national focus and then with a specific California focus.

Enteric Emissions and Covering Anaerobic Lagoons

Enteric Emissions

Although it may be a significant source of livestock emissions in California, enteric fermentation has received relatively little attention from economists. Therefore, the discussion of this source here is brief.

Animal science research indicates that GHG emissions from enteric fermentation can be mitigated by modifying the ruminant diet (ICF International 2013; Moraes, Fadel, Castillo, and Kebreab 2013). Studies have found that feed with a relatively low concentration of nutrients per unit of volume causes enteric methane emissions per unit of feed value higher than those of high-quality feed. Therefore, methods to reduce enteric emissions include increases in supplemental fat, protein content, or the amount of feed concentrates rather than low-calorie roughages in the diet. However, the potential of these methods to feasibly reduce enteric emissions is generally believed to be limited because rations in California already have high protein and fat contents and adding more fat is either not economical or could cause rumen disorders.[7] Pharmaceutical supplements have been considered, but their long-term effects are unknown

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[6] ICF International (2013) provides a broad analysis of GHG mitigation options related to manure management and identifies technologies and practices that can be adopted in response to policy incentives. The mitigation options considered in ICF International (2013) are using anaerobic digesters, covering existing manure storage facilities (i.e., ponds, tanks, or lagoons), improving separators, and improving nitrification–denitrification systems.

[7] Moraes, Fadel, Castillo, and Kebreab (2013) presents detailed research on how various mixes of dairy feed may affect GHG emissions from enteric fermentation and feed costs.

(ICF International 2013). Scientific research in enteric emissions is still developing, and there has been little economic evaluation of the options.

Moraes, Fadel, Castillo, and Kebreab (2013) reviewed technical options for mitigating enteric CH_4 emissions by changing dietary compositions. Based on the relationship between enteric emissions and dietary compositions that are available from previous studies, they presented a linear programming model that minimized the increase in dietary cost subject to a given reduction of enteric emissions. By progressively increasing the assumed reductions in enteric emissions, their study calculates the carbon prices associated with each emissions reduction. Their results show that carbon prices associated with enteric emissions are relatively high: \$244/MTCO_2e at a 3% reduction of enteric emissions from the baseline scenario, \$544/MTCO_2e at a 20% reduction, and \$2,270/MTCO_2e at a 24% reduction.

Covering Anaerobic Lagoons

GHG mitigation options developed below focus primarily on reducing CH_4 emissions from anaerobic lagoons. Most economic analysis has concerned the economic returns to anaerobic digesters with electricity generation. However, before turning to that option, this discussion considers the simple procedure of covering an existing lagoon to avoid emissions.[8]

Installation of an air-tight cover over an existing lagoon, tank, or pond can reduce emissions by allowing for the capture and destruction of methane gas. After methane is captured, it can be flared using a combustion device. Manure handling practices compatible with this covering system include the liquid/slurry practice as well as the anaerobic lagoon system (ICF International 2013). Covering manure and flaring the gas requires relatively simple technology.

In California, candidates for lagoon covering are only those farms with existing anaerobic lagoons, because new lagoon construction or significant modifications would likely require stringent permit requirements from local authorities (ESA 2011; ICF International 2013), which entail substantial costs. Thus, the cost calculations presented here do not include the cost of lagoon construction. ICF (2013) finds that lagoon covering costs per unit of manure decline with the amount of manure handled. Total costs for a covered lagoon system ranges from \$0.1 million for a herd of 300 cows (\$333 per cow) to \$0.9 million for a herd of 5,000 cows (\$180 per cow) (ICF International 2013). Based on these costs, ICF (2013) calculates the break-even prices of carbon offsets per MTCO_2e as follows: \$5 per MTCO_2e for 5,000 cows; \$7 per MTCO_2e for 1,000 cows, \$8 per MTCO_2e for 600 cows, and \$9 per MTCO_2e for 300 cows.

The ICF calculation of 17,336 MTCO_2e of emissions reduction for covering the lagoon of a 5,000-cow dairy implied an average reduction per cow of 3.5 MTCO_2e. That average is used here to calculate the potential emissions reduction in California for covering lagoons. According to CARB, California dairies generated 9.7 MMTCO_2e of manure-related emissions in 2009 (Table 3), and anaerobic lagoons generated the lion's share of manure-related GHG emissions in California dairies. The percentages of GHG reductions provided in Table 4 are calculated on the basis of the total emissions inventory of 9.7 MMTCO_2e. These percentages allow calculation of the carbon offset supply potential from dairy lagoon covering. Based on the data and assumptions in Table 4, covered lagoons would eliminate 26% of total dairy-manure-related emissions at \$6/MTCO_2e, 50% at \$6.5/MTCO_2e, 63% at \$8/MTCO_2e, and 69% at \$9/MTCO_2e. These break-even prices are considerably low compared with the prices for enteric emissions mitigation options.

[8] Two other mitigation options include the installation of a solids separator, which removes solids from manure, and nitrification and denitrification systems that remove solids and nitrogen from effluent streams.

Table 3. Dairy-Manure-Related GHG Emissions (Million MTCO2e) in California from Farms with Anaerobic Lagoon Systems and All Farms, 2009

	Total GHG	CH_4 emissions	N_2O emissions
	(Million MT CO2 equivalent)		
Anaerobic lagoon	7.6	7.2	0.4
Total	9.7	8.5	1.2

Source: CARB (2011).

Table 4. GHG Emissions Mitigation Potential for Covered Anaerobic Lagoons in California

		Herd size			
	Total in CA	200–500	500–1000	1000–2500	> 2500
Number of farms with dairy cows[a]	13,544	962	681	498	180
Number of dairy cows (thousands)[a]	2,503	298	467	745	725
Assumed % of cows in lagoon systems[b]		60%	80%	90%	100%
Emission reduction potential ($MMTCO_2e$)[c]		0.6	1.3	2.3	2.5
Share of total dairy manure GHG reductions[d]		6%	13%	24%	26%
Break-even price used ($\$/MTCO_2e$)[e]		9	8	6.5	6
Total offset value ($Mil)		5.6	10.4	15.1	15.1

Sources: USDA and NASS (2009), ICF International (2012), and authors' calculations.
[a] Data from the 2007 Census of Agriculture; total (dairy cows and heifers) exceeds number of lactating cows.
[b] No detailed data are available on lagoon use by size. Estimates of the shares of cows in lagoon systems are based on two facts: (1) large-scale dairy farms in California are known to rely almost exclusively on anaerobic lagoon systems to manage dairy manure and (2) California GHG inventory data indicate that for the industry as a whole almost 80% of dairy-manure-related GHG emissions are from anaerobic lagoon systems.
[c] Emissions reductions are calculated on the basis of the reduction of 3.5 $MTCO_2e$ per cow derived from the emissions reduction calculation by the ICF. This average is multiplied by the number of cows in a lagoon system in each size category.
[d] These percentages are based on the total dairy-manure-related emissions of 9.7 $MMTCO_2e$.
[e] The farm size categories used in ICF International (2012) differ from the categories used in the Census of Agriculture. Break-even prices by ICF International (2012) have been adjusted using median values.

Background on Dairy Digesters, Digester Policy, and Digester Economics

To facilitate economic evaluation of dairy digesters that generate bio-energy, some basic information on digester system operation and policy is presented here.

Anaerobic Digestion

Biogas recovery systems are often referred to as anaerobic digesters, because they use anaerobic digestion during which manure is decomposed in an oxygen-free environment.[9] Anaerobic digestion of animal manure has multiple benefits, including biogas generation, odor control, and reductions in pathogenic properties of manure (Yiridoe, Gondon, and Brown 2009). Once biogas, comprised mostly of methane, is produced, the liquid effluent can be used as a fertilizer, and digested solids, removed from the digester effluent by means of a solids separator, can be used as livestock bedding or mulch. The collected biogas is most often used to generate electricity for on-farm use or sale to the local electric utility.

Digester Types

Three primary types of anaerobic digester systems are commonly used to treat dairy manure and capture the biogas. A covered lagoon system is the simplest type. This system involves two lagoons, one with an impermeable cover for methane generation and another for effluent storage. This system is not heated and thus is used in warmer climates. It is compatible with flushed manure (Marsh, LaMendola, Schiffler, and Sousa 2009).[10] The second type is a complete-mix system and involves a tank into which manure and water are added and mixed. This technology is compatible with slurry manure. To maintain an optimal temperature for methane production, the digester has to be heated. Complete-mix systems are compatible with slurry manure and expensive to construct and require applied energy. The third type, the plug flow digester, consists of a long lined tank, often built below ground, which receives manure in batches (or plugs). Because this system requires semi-solid manure with a consistency solid enough for the formation of "plugs," it is compatible with scrape manure management systems. Plug-flow digesters can be heated or unheated and are generally used in colder climates.

The three types of systems differ in cost and energy production intensity. Complete-mix and plug-flow systems are the most costly, and their energy production intensity is highest, which tends to equalize the unit cost of energy for all systems. The choice of digester type depends on factors such as existing manure collection method, climate, farm size, possibility of digester feedstock other than manure, and use (on-farm or sale) of generated energy.

Co-Digestion

Total biogas production from a dairy manure digester can be greatly increased by adding other non-manure organic feedstock, a process referred to as *co-digestion*. Organic materials with higher energy content usually produce higher amounts of biogas. Most food wastes produce a much higher level of biogas per unit of input than dairy manure. Further, the biogas facility may receive a tipping fee for food-processing wastes that are otherwise processed in municipal waste treatment facilities. By contrast, co-digestion requires more capital and entails intensive oversight to comply with water quality and solid waste regulations (ESA 2011). The compatibility of feedstock differs by digester system. Covered lagoon systems are well suited for co-digestion of whey and vegetables or similar agricultural wastes but not for co-digestion of heavier, more concentrated feedstock (e.g., grease) (ESA 2011).

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[9] The term *digester* refers to a biogas-producing system such as a lagoon or a tank where biogas is generated. However, this term is also used, in a broader sense, to denote the entire bio-energy producing system, including the engine generator.

[10] Excreted dairy manure has about 14% solid content. *Solid* is defined as more than 20% solid content, *semi-solid* as 10–20% solid content, *slurry* as 5–10% solid content, and *flushed* as less than 5% solid content (U.S. EPA 2011).

Many previous studies on dairy digesters incorporated co-digestion in their framework (ESA 2011; Bachewe et al. 2008; Bishop and Shumway 2009; Bishop, Frear, Shumway, and Chen 2010). Bishop and Shumway (2009) especially emphasized the economic value of co-digestion through high biogas production, tipping fees, and revenue from receiving food waste. In California, almost half of the dairy digesters operating in 2013 co-digested food (creamery) waste.

Output of Dairy Digesters

Anaerobic digestion produces two main types of products: biogas and digestate. The biogas can be either flared or burned in a generator to produce renewable electricity, natural gas, or compressed natural gas (CNG). Digestate is the material remaining after the anaerobic digestion of a biodegradable feedstock. Digested fiber can be used as soil amendments or bedding material for livestock (Leuer, Hyde, and Richards 2008). The value of digestate as a soil amendment is limited in California, because concerns about salt and nitrate loading limit land application rates (ESA 2011). Cow manure used as fertilizer can have a negative impact on ground and surface water quality if not sufficiently disbursed over enough land (Informa Economics 2013). Some studies on dairy digesters explicitly incorporate digestate value in their revenue stream (Bishop and Shumway 2009; Bishop, Frear, Shumway, and Chen 2010; Leuer, Hyde, and Richards 2008; Lazarus and Rumdstrom 2007).

Carbon Credits or Renewable Energy Credits

In addition to producing a tangible output, dairy digesters remove a potent greenhouse gas by capturing and using methane. This activity may create so-called carbon credits or carbon offsets, when measured in carbon-equivalent units. Further, when dairy digester projects use captured biogas to generate electricity, natural gas, or vehicle fuel, they can receive renewable energy credits. The type of renewable credits available depends on the final form of energy output and the regulatory regime (Table 5).

California dairy digester projects such as the Cottonwood dairy reportedly received carbon credits through the Chicago Climate Exchange (CCX). The CCX was North America's only voluntary GHG trading system, but it ceased to operate in 2010 (Gronewold 2011). Starting in 2013, the state of California began to operate a carbon offset program through the cap-and-trade program required by AB 32. Dairy digesters are eligible for carbon credits, which can be traded under the program. Thus far, none of the dairy digester projects in California has either applied for or received carbon credits.[11] The effectiveness of most renewable credits is also limited for dairy digesters, because most renewable energy credit programs are designed mainly for small producers. Table 5 lists credit programs currently available in California. CDC (2013) and ESA (2011) provide information on these programs.

Table 5. Credits Available to Dairy Digester Projects in California

Program	Eligible energy type	Authority	Tradability
For carbon offsets			
Cap-and-trade program	GHG reduction	CA AB 32	Tradable
For generation of renewable energy			
Renewable feed-in tariff	Electricity	CA SB 32, CA SB 1122	
Renewable action mechanism	Electricity	CA Renewable Portfolio Standard	
Renewable energy credit	Electricity	CA Renewable Portfolio Standard	Tradable
Renewable identification numbers	Vehicle fuel	Federal Renewable Fuels Standard	Tradable
Low-carbon fuel standard	Vehicle fuel	CA AB 32	

[11] According to J.P. Cativiela of the California Dairy Campaign (pers. comm.), CARB has approved "listing" of 24 livestock projects. However, none are in California. See http://www.arb.ca.gov/cc/capandtrade/offsets/earlyaction/projects.htm.

Economies of Scale

The construction of an anaerobic digester incurs a large capital investment, which includes cost components that are fixed or do not vary proportionally with amount of material digested. Some operating and regulatory compliance costs are also not proportional to scale. For example, one-time or annual permit fees or utility interconnection fees are mostly invariant to amount of material digested or amount of energy generated. The possibility of economies of scale in anaerobic digester projects was suggested as early as 2002 (U.S. EPA 2002), and almost all studies on dairy digesters verify that per unit costs decline with the scale of the project.

Much attention has been paid to identifying the minimum threshold dairy herd size that makes a digester project viable. Gloy and Dressler (2010) consider this size to be between 500 and 1,000 cows, and this view is widely accepted (U.S. EPA 2010b, 2011; Key and Sneeringer 2011; Gloy 2011). Several recent case studies analyze relatively large operations. Gloy (2008) considers an 11,000-cow dairy operation, and most California studies analyze systems of 10,000 or more cows (ESA 2011; CDC 2013). Centralized digester systems are explicitly designed to gain economies in digester operation by using the manure from a cluster of dairy farms.

Centralized Digesters

A cluster of dairy farms may be connected through a manure distribution system to a centralized digester. The four main benefits of centralized systems are economies of scale, better leverage in marketing of energy output, additional financing opportunities, and third-party management (Leuer, Hyde, and Richards 2008; Bachewe et al. 2008). Centralized systems have been studied often (Gooch et al. 2010; Gloy 2010; Dusault 2008; Ghafoori and Flynn 2006; ESA 2011; CDC 2013). Bachewe et al. (2008) noted that centralized systems are common in Europe but rare in the United States. Dairy farms are much smaller in Europe than in California, and a centralized system may be the only feasible option for most European dairy farms. Two recent California studies (ESA 2011; CDC 2013) examine cases with more than 10,000 cows and explore the possibility of producing pipeline-injected biogas, which involves substantial interconnection costs.

The Port of Tillamook Bay in Oregon operates a well-publicized centralized digester, using manure from 4,000 cows. This facility began operating in 2003 using manure trucked from 7 to 11 nearby farms. A centralized system operated in the Chino Valley, California, has ceased operation (Cheremisinoff, George, and Cohen 2009; Marsh, LaMendola, Schiffler, and Sousa 2009).

Regulatory Policies

Dairy digesters are regulated by federal, state, and local agencies concerned with air, energy, water, climate, and transportation. In 2005, new regulatory programs were enacted for the construction of digesters at dairies (CalEPA 2011). New regulations applied to nitrogen oxides (NO_x) emissions of the digesters located in the San Joaquin Valley and the South Coast. In 2007, the Central Valley Regional Water Board began to apply more stringent requirements for waste discharge from dairies and dairy digesters (CalEPA 2011), and in December 2010, it developed complete waste discharge requirements for dairy manure digesters and co-digesters. In addition, the California Department of Resources Recycling and Recovery requires dairy farms to have a permit to handle solid wastes generated from digester operation. The California Public Utilities Commission (CPUC) regulates distribution of electricity and pipeline bio-methane through its regulated utilities, and the Federal Department of Transportation, the

California Department of Transportation, and the California Highway Patrol oversee over-the-road movement of biofuels (CalEPA 2011).

Additional Factors Affecting Economic Outcomes

Digester location greatly influences the economic payoff of a dairy digester. Differences in regional environmental regulations entail differences in the costs of a digester system. For example, the NO_x requirements for electrical generators used in California's Central Valley result in additional development costs. Lazarus, Goodkind, Gallagher, and Conway (2011) estimated additional capital costs of \$50/cow and additional operating costs of \$25/cow/year from this regulation. Further, depending the location, retail and wholesale prices offered to digester operators vary considerably (Cheremisinoff, George, and Cohen 2009; Marsh, LaMendola, Schiffler, and Sousa 2009). Distance from the digester to a hookup to the power grid also determines the feasibility of interconnection for power export (Black and Veatch 2013).[12]

Federal and state government support in the form of grants, favorable loan arrangements, and tax incentives has been vital to digester development. Grants have averaged more than 40% of capital costs in projects involved in case studies. Many case studies investigated the effects of grants on the economic payoff of a specific digester system. Lazarus and Rudstrom (2007) calculate that without government grants their 8% internal rate of return drops to -13%.

Digester operating life, risk premiums, and applicable discount rates also affect economic outcomes (Enahoro and Gloy 2008). Assumptions about discount rates have ranged from 3% to 10% and, given the long expected life of a digester (20 years on average), a change in the discount rate can easily change the sign of the net returns of the digester investment.

Finally, the pre-existing manure management system affects payoff in two ways. First, GHG emissions vary substantially across manure management practices, and the existing emissions form a baseline on which the carbon offsets are calculated. Second, initial construction costs depend on the existing manure system. A concentrated supply of methane can be only captured by storing manure in anaerobic conditions, and lagoon or pit types are the only suitable manure management systems that can be incorporated cost-effectively into a methane digester system (Gloy 2010; Key and Sneeringer 2011).

National Case Studies

The predominant subject for studies of digester economics has been the economic viability of digester construction and operation. While the majority of studies have focused on identifying conditions favorable for adoption of digesters in the context of individual operations, recent research has considered how digesters contribute to a supply curve of GHG mitigation at a national or regional level. Mitigation supply curves may identify mitigation subsidies needed to induce degrees of digester adoption and aggregate GHG mitigation. The discussion below covers case studies at the farm level and then aggregate studies focused on deriving the mitigation supply curve.

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[12] Black and Veatch (2013) indicate that the high interconnection-cost areas in California are mostly located in the north. They include Humboldt, Mendocino, Glenn, Plumas, and Sierra counties but also some Central Valley locations.

Farm-Level Case Studies

Most studies on digester economics have analyzed potential cost and revenue streams in a case study format. Based on either actual data or "best guess" data, in the context of a hypothetical project, these studies develop estimates of an internal rate of return or net present value of an investment in a digester. These assessments of economic performance differ considerably across individual studies, depending on specific operational situations and assumptions. The literature focuses primarily on accounting of costs and revenues to assess economic viability and identify potential to improve the economic performance of digester systems.

As noted above, herd size, which determines the amount of material to be digested, has long been recognized as crucial. Early studies tend to analyze herds with fewer than 500 cows (Mehta 2002; Goodrich 2005), whereas recent studies tend to consider much larger herd sizes. To control factors other than herd size, Lazarus, Goodkind, Gallagher, and Conway (2011) used the same approach to investigate six progressively larger herd sizes: 100, 200, 500, 1,000, 2,500, and more than 2,500 cows. Leuer, Hyde, and Richards (2008) investigate digesters for herds of 500, 1,000, and 2,000 cows. These studies find strong positive relationships between herd size and profitability; the major cost advantage stems from declining capital cost per cow with size.[13]

Gloy (2008) emphasized the importance of co-digestion using the example of an 11,000-cow dairy operation. He demonstrated that, despite the large herd size, manure alone could not make the digester system viable, and co-digestion of high-energy feedstock was crucial to produce sufficient electricity to be profitable.[14] Bishop and Shumway (2009) also employed the framework of co-digestion.

Many studies analyzed profitability by investigating revenue streams other than those from electricity generation. Dairy manure contains fibrous solids that can serve as mulch or livestock bedding as well as food wastes, which when used as additional feedstock may generate tipping fees. Bishop and Shumway (2009) and Bishop, Frear, Shumway, and Chen (2010) investigated these revenue possibilities in a project in which food wastes accounted for 16% of total influent but produced about half of gas production. Bishop and Shumway (2009) investigated how much net present value (NPV) and internal rate of returns (IRR) rise with an additional revenue stream. Their study found that tipping fees raised the IRR from a negative value to 17%, but co-product sales add little new revenue.[15] Inclusion of carbon credits at \$4/MTCO_2 increased the IRR to 20%, and further raising the carbon price to \$20/MTCO_2 increased the IRR to 27%.

Leuer, Hyde, and Richards (2008) investigated the economic performance of additional revenue streams in a stochastic framework that assigned probability distributions to some parameters. For example, they assume that a system's economic life has a triangular distribution with three points: 10, 15, and 20 years. About half of the parameters were specified as stochastic. Based on distributional assumptions on parameters, the study employed Monte Carlo simulations and derived the probability of NPV being

[13] Leuer, Hyde, and Richards (2008) estimated that the capital cost per cow drops almost by half, from \$1,608 to \$887, when farm size increases from 500 cows to 2,000 cows. Per cow capital costs found in other studies include \$530 for 800 cows (Lazarus and Rudstrom 2007) and \$940 for 1,000 cows (Enahoro and Gloy 2008). Based on 37 vendor quotes, Key and Sneeringer (2011) noted that the capital cost per cow ranged from \$238 to \$1,672, with an average of \$937.

[14] However, to receive additional feedstock such as food wastes, the feedstock source must be located within an economically feasible distance, which is determined by site-specific conditions.

[15] The market for fiber products is well developed, but in 2007, only 6 of 92 dairy digesters in the United States had marketed their fiber products for purposes other than bedding (Bishop et al. 2010).

greater than or equal to zero. Their results highlight the importance of a solid separator and the production of bedding material.[16]

Following Leuer, Hyde, and Richards (2008), Stokes, Rajagopolan, and Stefanou (2008) used the theory of real options to analyze the economic feasibility of installing a digester on a dairy farm in Pennsylvania. They calculated NPVs and option values under many different parameter settings and found that positive option values exist under most scenarios. Their general conclusion is that when the possibility of option value is considered, project deployment may require higher projected net returns to compensate for the option value of waiting to see if technology or pricing improve.

Gloy and Dressler (2010) identify financing difficulties as well as lack of information as major barriers for digester adoption. They indicate that lenders are reluctant to finance digester projects because of uncertainty stemming from lack of information regarding initial capital investment, predicted biogas production, expected lifetime, future electricity prices, and operating costs.

To provide a general evaluation of a financial payoff for potential digester projects, a tool kit called FarmWare, was made available by the U.S. Environmental Protection Agency's AgSTAR Program. FarmWare produces the IRR and other associated figures on the basis of a capital budgeting approach. Enahoro and Gloy (2008) applied their own model and FarmWare to a hypothetical project. Their results indicate that under compatible assumptions, FarmWare produced a higher performance estimate than their model. Finally, Faulhaber and Raman (2011) developed a case study tool to evaluate potential plug-flow digester projects. They evaluate these projects' non-market environmental attributes such as odor reduction in addition to GHG mitigation (Faulhaber, Raman, and Burns 2012) using the ratio of the net cost of biogas production to the market price of natural gas as the evaluation criterion.

Aggregate Studies

The recent literature has focused on assessing the potential adoption of dairy digesters throughout the United States (U.S EPA 2010b; Gloy 2011; Key and Sneeringer 2011; Key and Sneeringer 2012; Zaks et al. 2011). Consistent with the widespread notion that government incentives are needed to capture the environmental benefits of dairy digesters, these studies compute the minimum carbon price that induces adoption. By aggregating those farms estimated to profitably adopt digester technology at each carbon price, they construct a potential supply of carbon equivalent offsets. The national-level assessment requires distributional information about dairy farms; two sources of information are used: the Census of Agriculture and the Agricultural Resource and Management Survey (ARMS), both of which are provided by the U.S. Department of Agriculture (USDA). Both sources provide nationwide farm-level information. The Census of Agriculture is conducted every five years and the ARMS is conducted annually.

U.S EPA (2010b) investigated the potential for bio-energy production from confined commercial livestock facilities in the United States. Dividing the nation into 11 regions (the top 10 dairy states and the rest) and based on farms with a minimum of 500 cows, the study identified 2,645 dairy farms nationwide as potential digester adopters. Together, these farms have the potential to eliminate more than 85% of total dairy manure-related emissions (U.S EPA 2010b, 2011). The study also identifies California as the state with the highest methane emissions reduction potential–38% of the national total. Although these estimates provide some guideline for cross-region comparison, they rely on simplifications that limit their applicability. Selection of candidate farms was based only on size, and estimates of avoided CO_2e emissions likely over-represent the replacement of electricity produced from fossil fuel. For example, in

[16] A solid separator separates the liquid and solid portions of the manure before the manure is transported to the digester. The liquid portion is stored, but the digested solids have value as bedding or soil amendments.

California, coal represented only 1.7% of electricity generated in 2010 (California Energy Commission 2013) compared with the national average of 30% from coal (EIA 2013).

Gloy (2011) estimates an aggregate supply curve for CO_2 offsets based on ARMS data and state electricity prices obtained from the U.S. Energy Information Administration (EIA). He takes account of location-specific electricity prices in the calculation of electricity savings, sales revenue, or both. Given the condition of at least zero profit for digester adoption, the supply curve for carbon offsets is calculated by balancing cost and revenue. The supply curve is approximated to be linear up to \$30/MTCO$_2$e; it becomes very nonlinear above \$30. Carbon offsets are about 0.8 million MTCO$_2$e at no payment for offsets and about 13 million MTCO$_2$e at \$20/MTCO$_2$e, implying that each dollar increase in price induces a reduction of 0.6 million MTCO$_2$e (or 2.7% of total emissions) up to \$30.

In his subsequent work, Gloy (2012) emphasized the importance of more stringent criteria in selecting candidate farms. He argues the investing farm has to be financially sound, anticipating staying in business for a reasonable period, and large enough to take advantage of economies of scale. He suggests that candidate farms have a debt/asset ratio below 20%, expect at least 20 more years in business, use anaerobic manure systems, and milk at least 500 cows. The 2005 ARMS data identified only 318 such farms nationally. Gloy (2012) also indicates the importance of targeting large farms, noting that the largest 1% of farms could reduce emissions by nearly 25%.

Key and Sneeringer (2011) estimate a carbon supply curve for U.S. dairy farms using more detailed cost data based on 14 case studies and 31 vendor quotes (U.S. EPA 2009). The capital cost, estimated as a function of herd size, increases with herd size at a decreasing rate; operation and maintenance costs (O&M) were also specified as a function of electricity generation. To reflect the effect of climate on lagoon systems, state adjusters are incorporated into the O&M costs. Key and Sneeringer (2011) report the results for only two carbon prices, \$13 and \$26/MTCO$_2$e, and emphasize their effects on adoption rate. At \$13/MTCO$_2$e, 934 farms, representing 48% of farms with more than 1,000 cows, adopt digesters. At \$26/MTCO$_2$e, more than 70% of the farms that have more than 1,000 cows adopt digesters. At the price of \$13, carbon offset revenue represents 62% of the present value of gross returns. With no carbon credit, no farms under 2,500 cows would adopt digesters. Key and Sneeringer (2012) also incorporate subsidy policies, including loan guarantees, accelerated depreciation, cost-share programs, and policies enhancing demand for digester electricity. Their results indicate that government grants would most affect adoption.

ICF International (2013) conducted a similar analysis based on AgSTAR's cost formulae. The ICF study developed costs and emissions reductions specific to digester types, existing manure management practices, and farm size. The study's revenue estimation is conservative in the sense that it assumes farms sell no electricity, given uncertainty about the availability of required infrastructure. Break-even carbon prices are calculated as a price that balances cost and revenue. Break-even prices differ by region due to region-specific retail electricity prices that affect the savings accrued from on-farm use.

The Pacific region (which includes California) shows greater potential for GHG reductions at a lower break-even price than other regions. This result follows from the higher emissions baseline of the Pacific region due to its warmer climate and higher proportion of anaerobic lagoons. The region's break-even carbon prices for transition to a lagoon-based digester are \$2/MTCO$_2$e for a farm with 5,000 cows, \$17/MTCO$_2$e for a farm with 1,000 cows, and \$30/MTCO$_2$e for a farm with 600 cows. Clearly, herd size is the dominant factor in digester adoption.

These prices, especially \$2 for the 5,000-cow category, are relatively low compared with those found in other studies. These low break-even prices may be partly due to the overestimation of savings related to on-farm use of electricity. The ICF study assumes that all farms use 70% of generated electricity for on-farm use. But, this share likely overestimates the amount of electricity used on large Pacific region dairies. Data shows that electricity production per cow from digesters is roughly constant but that on-farm electricity use per cow decreases as herd size increases. This finding implies that the electricity share of on-farm use falls with number of cows. Therefore, the constant 70% share results in overestimated savings from on-farm use of electricity that are credited at the retail price, which usually is much higher than the wholesale price. Therefore, the realistic break-even price for large farms, particularly farms with 5,000 cows, would be higher than the estimated \$2/$MTCO_2e$. This higher break-even price is also more consistent with the observation that although farms are relatively large in California (the average herd size exceeds 1,100), digesters are uncommon—even where sale of digester-produced energy is considered crucial.

Comparison of Assumptions and Implications

Cost assumptions are critical in evaluating the financial performance of digester projects and the marginal cost of GHG mitigation potential. Table 6 summarizes the cost assumptions used in the studies reviewed above. Most studies use three approaches to arrive at cost estimates: accounting information obtained from actual operations, "best guesses" from the previous literature, and statistical estimation based on a variety of information, including vendor quotes.

Table 6. Cost Assumptions Used in Previous Studies

Study	Cost data source	Location	Herd size	Digester type	Co-digestion	Electricity generation (kWh)/ cow/ year	Capital cost ($)/ cow or total capital cost	O&M cost as % of capital cost
Enahoro and Gloy (2008)	Previous studies	NY	1,000	Not specified	No	1,115	$940/cow	4.20%
Lazarus and Rudstrom (2007)	Actual data for 1999–2004	MN	800	Plug-flow	No	1,253	$530/cow	8.70%
Leuer, Hyde, and Richard (2008)	Previous studies	PA	500	Not known	No	n/a	$1,608/cow	1.50%
			1,000		No	n/a	$1,073/cow	1.50%
			2,000		No	n/a	$887/cow	1.50%
Bishop and Shumway (2009)	Actual data for 2005–2007	WN	750	Plug-flow	Yes	2,590		11%
Lazarus, Goodkind, Gallagher, and Conway (2011)	Estimated based on AgSTAR database	National	Any farm size with profit≥0 (b)	Mixed and plug-flow	No	2,286	516,465+$589 *(no. of cows)[a]	3%
				Covered lagoon	No	1,489	$471,533+$538*(no. of cows)[c]	3%
				For CA			extra $50/kW[d]	
AgSTAR (2010a)	Estimated using 40 vendor quotes	National	All farms with herd size (>500)	Complete mix	No		320,864+$563 *(no. of cows)	8.40%
				Plug-flow			566,006+$617 *(no. of cows)	
				Covered lagoon			599,556+$400 *(no. of cows)	
Key and Sneeringer (2011)	Estimated using case studies and vendor quotes	National	Any farm size with NPV>0	Mixed and plug-flow	No	729	$17,654*(\text{no. of cows})^{0.596}$ [e]	function of electricity generated
				Covered lagoon	No	450[f]	$39,020*(\text{no. of cows})^{0.454}$	
Gloy (2011)		National	Any farm size with profit≥0[b]	All types	No	1,100	$10,000*(\text{no. of cows})^{0.7}$	3.50%

[a] The ancillary cost of $110,000+$125*(no. of cows) is included.
[b] Annualized capital costs are used.
[c] Lazarus assumed that covered lagoons cost 8.7% less than heated systems.
[d] The extra costs for California are due to extra capital cost investment to meet the tougher California NO_x standard.
[e] This equation was converted from the original log-log model to facilitate comparison with other estimates presented here.
[f] State factors are applied to this base value, which can be considered an average.

Capital per-cow costs range from \$530 to \$1,608. Lazarus, Goodkind, Gallagher, and Conway (2011) assume an extra capital cost of \$50 per Kw of generator capacity for the covered lagoon system in California to account for meeting the NO_x requirement in the Central Valley. A California study, ESA (2011), also indicates that meeting the NO_x requirement implies additional costs, but it provides no estimates.

Assumptions about O&M cost also vary widely from 1.5% to 11% of capital costs. The high O&M cost reported in Bishop and Shumway (2009) is due to the co-digestion system, which causes additional feedstock handling costs. In Lazarus and Rudstrom (2007), the low capital cost is associated with a relatively high O&M cost. Different studies *may* use different accounting methods, some of which shift some cost items into the O&M cost category.

Aggregate national-level studies embed assumptions about declining unit cost with herd size into their cost formulae. Based on these formulae, Table 7 illustrates how per-cow capital cost declines as herd size increases. The effect of economies of scale seems to be larger for the lagoon system than for other systems.

Table 7. Per-Cow Capital Cost of Three Herd Sizes

Study	Digester type	Capital cost (\$)/cow with herd size		
		1,000 cows	2,000 cows	5,000 cows
Lazarus, Goodkind, Gallagher, and Conway (2011)	Mixed and plug-flow	1,105	847	692
	Covered lagoon	1,010	774	632
	California	1,025	781	639
AgSTAR (2010a)	Complete mix	884	723	627
	Plug-flow	1183	900	730
	Covered lagoon	1000	700	520
Key and Sneeringer (2011)	Mixed and plug-flow	1,084	819	566
	Covered lagoon	898	615	373
Gloy (2011)	All types	1,259	1,023	777

Source: Calculation by authors based on formulae in Table 6.

Table 8 summarizes economic performance indicators and selected non-capital cost assumptions for several studies. Discount rates range from 3% to 10%. Further, assumed system life varies from 10 to 40 years. Subsidies are reported only for the cases using actual operational data. Most farm-level studies use the internal rate of return (IRR) or net present value (NPV) as an indicator for economic performance. In the aggregate studies (Key and Sneeringer 2011; Gloy 2011), the condition of non-negative NPV is used to identify the farms that adopt digesters.

Digester systems examined in *ex post* case studies represent generally successful cases, otherwise they would not be in a sample of farms that actually had operating digesters. These cases are not representative of performance on a typical hypothetical farm or, especially, of the vast majority of farms that chose not to adopt the technology.

Table 8. Assumptions Used for Key Parameter Values Other Than Cost Figures

	Discount rate or inflation rate	Lifetime (years)	Subsidy (as % of capital cost)[a]	Economic performance indicators[b]
Enahoro and Gloy (2008)	10%	20 and 7[c]	None	IRR, NPV (IRR=4% in base scenario)
Lazarus and Rudstrom (2007)	3%	10	36% grant and 6 yr no interest loan on $150,000	IRR, NPV (IRR=8% in base scenario)
Leuer, Hyde, and Richard (2008)	8%	10, 15, 20	None	Probability of NPV≥0
Bishop and Shumway (2009)	4%	20, 30, 40	38%	IRR, NPV, MIRR[d]
Key and Sneeringer (2011)	5%	15	None	NPV≥0
Gloy (2011)	5%	20	None	NPV≥0

[a] Subsidy rates are reported only for the actual data.
[b] When the base scenario is plausible, the value of IRR is reported.
[c] 20 years for digester and 7 for generator.
[d] Not made each period.

Studies with a California Focus

Research specific to dairy digesters in California began as early as 1996 (Morse, Guthrie, and Mutters 1996), but with one exception (Camarillo, Stringfellow, Jue, and Hanlon 2012), studies with an economic focus have not been published as peer-reviewed academic articles. However, a number of reports have been commissioned or initiated by state government agencies or quasi-public organizations, and this review includes some of those reports. The California Energy Commission (CEC) commissioned Marsh, LaMendola, Schiffler, and Sousa (2009); Cheremisinoff, George, and Cohen (2009); and Dusault (2008). The California Public Utility Commission (CPUC) commissioned Black and Veatch (2013). The California Regional Water Quality Control Board, Central Valley Region, commissioned ESA (2011). A California Dairy Campaign report (CDC 2013) was sponsored by the U.S. Department of Agriculture.

Marsh, LaMendola, Schiffler, and Sousa (2009) and Cheremisinoff, George, and Cohen (2009) were prepared as part of a program assessment and provide extensive information about the performance of each operating dairy digester project that received state funding.[17] The other reports either provide policy analysis or examine hypothetical digester projects with California-specific features. The review here summarizes information about individual dairy digesters in California and examines hypothetical cases that are considered in other reports.

Performance Evaluation of California Dairy Digesters

[17] The California Energy Commission had distributed $5.8 million to 10 California dairies by early 2005.

Table 9 summarizes key characteristics of and performance information for operation of dairy digesters on 10 California dairy farms. The operation periods are 2004–2005 for nine farms and 2009 for one farm. Information about nine of the farms is primarily from Cheremisinoff, George, and Cohen (2009) and supplemented by other sources, including Marsh, LaMendola, Schiffler, and Sousa (2009), U.S. EPA (2013), and CARB (2013). Information on the Fiscalini farm is based on Camarillo, Stringfellow, Jue, and Hanlon (2012).

Using financial data, Cheremisinoff, George, and Cohen (2009) evaluated the economic performance of nine operations (1 of 10 awarded farms later withdrew its contract). Key assumptions were as follows: the lifetime of the system is 20 years; the inflation rate is 2.5%; a net metering system is in place, and net metering credits are sold back to utilities; the value of energy captured from heat or steam is credited back to revenue; and carbon credits or renewable credits, if any, are included.

The largest project was formed by the Inland Empire Utilities Agency (IEUA) as a six-farm cluster. This project incorporates food waste as feedstock for dairy digester operation. Two other co-digestion projects use creamery wastes.[18] To facilitate comparison, capital costs and operation/maintenance (O&M) costs are calculated on the basis of a unit of electricity production in kWh. Costs differ significantly across the projects; the most expensive is the IEUA project. Co-digestion systems incur O&M expenses almost four times per kWh higher than systems that use dairy manure alone. The unit cost of electricity generation ranges mainly from \$0.06 to \$0.09; the highest is \$0.14/kWh. Four of ten projects have negative IRR; the after-tax rate is not available for the Fiscalini project.

The Fiscalini farm received relatively large grants (57% of capital costs). Camarillo, Stringfellow, Jue, and Hanlon (2012) found that in the absence of these grants, the farm's NPV would have been negative. By the time this project began operation, NO_x emission controls had been placed on systems in the Central Valley, which added to capital and variable costs (ESA 2011). Even though Camarillo, Stringfellow, Jue, and Hanlon (2012) state that the cost of incorporating emissions control technology for NO_x removal was not large enough to reverse the Fiscalini farm's economic viability, other reports indicate that the farm incurred a substantial amount of additional costs (about \$4 million) to meet NO_x air quality requirements (Huffstutter 2010; Merlo 2010; ESA 2011).

Marsh, LaMendola, Schiffler, and Sousa (2009) conducted further analysis and more economic performance information is added to the information in Cheremisinoff, George and Cohen (2009). Table 10 summarizes additional information. Grants received are presented in the table and the payback years are also calculated under the assumption of no grant. Government grants account for over 40% of capital cost, on average.

[18] These two also own cheese factories at the same site.

Table 9. Economic Performance of California Dairy Digesters

Facility	Fiscalini	Castela-nelli Bro. Dairy	Cotton-wood	Hilarides	Blakes Landing	Eden-Vale	Koetsier	Van Ommering Dairy	Meadow-brook Dairy	IEUA (6-farm cluster)
City (county)	Modesto (Stani-slaus)	Lodi (San Joaquin)	Atwater Merced)	Linsay (Tulare)	Marshall (Marin)	Lemoore (Kings)	Visalia (Tulare)	Lakeside (San Diego)	El Mirage (San Bern-ardino)	Chino (San Bern-ardino)
Digester type	Complete mix	Covered lagoon	Covered lagoon	Covered lagoon	Covered lagoon	Plug-flow	Plug-flow	Plug-flow	Plug-flow	Plug-flow
Co-digestion	Grass & waste from cheese plant		Waste from cheese plant							Food wastes
Herd size	1,500	3,601	5,616	6,000	447	1,100	2,285	717	3,194	9,843
Year started	2009	2004	2004	2005	2004	2005	2005	2005	2004	2005
Generator capacity	710kW	160kW	300kW*	500kW	75kW	180kW	260kW	130kW	160kW	943kW
Electricity gen./ year	3,560 MWh	1,135 MWh	2,133 MWh	3,383 MWh	253 MWh	457 MWh	540 MWh	489 MWh	1,100 MWh	7,572 MWh
Capital cost ($/kW)	5,662	6,043	8,993	2,480	4,504	4,471	5,264	6,668	6,379	13,734
O&M cost (cents/ kWh)	4.35	0.94	4.34	0.45	1.16	1.16	1.15	1.61	2.71	10.20
Benefits ($/yr) from electricity gen. (share of on-farm use and sale @$/kWh)	420,000 (100% sale @ $0.1095)	92,730 (50% on farm @ $0.091 & 50% sale @ $0.0724)	159,548 (100% on-farm @ $0.0748)	217,493 (62% on-farm @ $0.0736, 38% sale @ $0.0491)	35,627 (60% on-farm @ $0.1509, 40% sale @ $0.1257)	16,817 (17% on-farm @ $0.0700, 83% sale @ $0.0300)	28,512 (76% on-farm @ $0.0600, 24% sale @ $0.0300)	24,450 (10% on-farm @ $0.0500, 90% sale @ $0.0500)	58,960 (68% on-farm @ $0.0600, 32% sale @ $0.0400)	605,760 (100% on-site use @ $0.0800)
Unit electricity cost ($/kWh)	NA	0.0817	0.094	0.0643	0.141	0.0449	0.0648	0.0613	0.0673	0.098
After tax IRR (%)	8.6 (pretax rate)	21.27	8.64	22.82	19.02	-13.95	-13.25	-0.12	4.76	-13.780

Sources: Information about Fiscalini is from Camarillo, Stringfellow, Jue, and Hanlon (2012); information for the rest is primarily from Cheremisinoff, George, and Cohen (2009), supplemented by Marsh, LaMendola, Schiffler, and Sousa (2009); U.S. EPA (2013); and CARB (2013).

Table 10. Economic Performance of California Dairy Digesters

	Estimated total capital cost	Total grants	Grants as % of total capital cost	Simple payback years (no grants)	Simple payback years (with grants)
Covered lagoons					
Blakes Landing	$334,680	$155,261	46%	18.3	9.8
Castelanelli	$882,136	$547,396	62%	26.9	10.2
Cottonwood	$2,498,038	$840,000	34%	10.3	6.8
Hilarides	$1,239,923	$500,000	40%	8.5	5.1
Plug Flow Digesters					
Eden-Vale	$802,811	$300,000	37%	70.3	44.0
Koetsier	$1,361,087	$190,925	14%	56.2	48.3
Meadowbrook	$720,605	$462,449	64%	14.8	5.3
Van Ommering	$836,838	$394,642	47%	34.0	18.0
Modified plug flow					
IEUA	$3,551,448	$948,175	27%	n/a	n/a
Complete mix					
Fiscalini	$4,020,200	$2,291,500	57%	23.0	12.0

Source: Marsh, LaMendola, Schiffler, and Sousa (2009).

Table 11 lists the dairy digesters operating in California as of May 2013 (U.S. EPA 2013). Four of the nine cases studied by Cheremisinoff, George, and Cohen (2009) no longer operate (Blakes Landing is now listed as Strauss).[19] The four ceased operations had negative IRRs (Table 9). A closer analysis of the data provided in Cheremisinoff, George, and Cohen (2009) (not shown in the table) indicates that with the exception of the IEUA projects, these projects shared one factor—low generator utilization rates: 24% for Koetsier; 43%, Van Ommering; and 29%, Eden-Vale. The average rate for the other five systems was 71%. Marsh, LaMendola, Schiffler, and Sousa (2009) attribute low utilization rates were to lack of power purchase agreements. Wholesale prices of electricity were also low for the no-longer-operating systems.

[19] Among these projects was the IEUA's Chino Basin project, the first centralized system in California (Marsh, LaMendola, Schiffler, and Sousa 2009; ESA 2011).

Table 11. Operational Anaerobic Dairy Digesters in California as of May 2013

Farm/Project Name	Year operational	Co-Digestion	Biogas end use(s)	Installed capacity (kW)	GHG emissions reductions (MT CO_2e/yr)
Castelanelli Bros.	2004		Electricity	300	12583
Cottonwood	2004	Creamery waste water	Cogeneration	700	26544
Hilarides	2004		Electricity; CNG	750	1918
Meadowbrook	2004		Electricity	180	279
Straus Family	2004	Creamery waste water	Cogeneration	75	1639
Bullfrog	2008		Electricity	300	17519
CAL-Denier	2008		Electricity	65	3983
Fiscalini	2008	Whey, sudan grasss	Cogeneration	710	5343
Tollenaar Holsteins	2008		Electricity	215	4778
Bob Giacomini	2009	Creamery waste water	Cogeneration	80	1593

Cottonwood Dairy (Gallo Farms) uses 100% of the power generated for on-farm and cheese plant use. Despite high capital costs (second highest among the 10 reviewed projects), Cottonwood achieved reasonable economic performance (an IRR of 8.6%). It was reported to be the only dairy that sold carbon credits on the Chicago Climate Exchange, though that process was cumbersome and involved up-front transactions costs (approximately $1,000 to sign up, $1,000 per year, and $3,000–$4,000 for a verifier) (Marsh, LaMendola, Schiffler, and Sousa 2009).

California Digester Studies: Hypothetical Cases

After a brief review of Black and Veatch (2013), two reports–ESA (2011) and CDC (2013) are discussed. Both focus on large operations and consider centralized systems and natural gas production.

In evaluating recent Senate Bill (SB) 1122, Black and Veatch (2013) estimated costs of dairy digestion. SB 1122 establishes a standard tariff for bio-based electricity generation by requiring utility companies to supply a mandated amount of electricity generated from designated bio-feedstock. Due to the capacity eligibility limit under SB 1122, this study focuses on small-scale projects and investigates the likely availability of resources and projected cost of electricity for projects eligible for the tariff under SB 1122. Black and Veatch (2013), identifying interconnection problems as a major hurdle for the participation in this feed in tariff program, state that many types of bio-based feedstock are located in rural areas, which may have limited transmission availability. They indicate that problematic areas in California for interconnection include the state's far north, including Humboldt, Mendocino, Glenn, Plumas, and Sierra counties and some Central Valley locations. They report that two dairy digesters in California have applied for and received feed-in tariff power purchase agreements.

ESA (2011) assesses the economic feasibility of dairy digesters in Calfornia's Central Valley and focus on co-digesting possibilities. Their economic model is based on the Generalized Revenue Requirements

Model made available by the California Biomass Collaborative.[20] ESA (2011) considers four scenarios: (1) manure only and electrical generation, (2) co-digestion and electrical generation, (3) manure only with bio-methane generation, and (4) manure only with a centralized bio-methane system. The first two scenarios are based on 1,000 cows, and the latter two, on 10,000 cows. In the co-digestion scenario, electricity production doubles with food waste, and under the centralized scenario, biogas is collected from eight nearby farms. The centralized system is more capital intensive than a single farm system of the same capacity. Bio-methane production adds costs for biogas upgrade, testing, and utility connection.

CDC (2013) investigates the potential to develop a centralized dairy digester project in the Central Valley with a natural gas production option.[21] The hypothetical project is located in Kern County within a few miles of a SoCal gas pipeline. Three scenarios are considered: (1) electricity generation, (2) renewable natural gas production, and (3) vehicle fuel (compressed gas) production. The farm cluster operates with 50,000 cows from 11 dairy farms. Although the study assumes a centralized system, Scenario 1 treats each farm as an individual electricity generator eligible for feed-in tariff benefits that are designed for small producers.[22] The study assumes no grant, a \$10/MTCO_2e carbon credit for all scenarios, and other subsidies such as renewable energy credits or low-carbon fuel standard (LCFS) credits when applicable.

ESA (2011) and CDC (2013) are similar in their scenario settings and assumptions. Table 12 summarizes their scenarios and results, presenting results by project type. System performance is evaluated (1) unit cost of energy production, (2) shortfall from the current energy price, and (3) additional revenue required to achieve the 18% of IRR—the rate that both studies consider adequate to cover uncertainty and risk.[23]

Overall, CDC (2013) uses a higher capital cost estimate but a lower O&M cost estimate, so that the unit production cost is almost equal to that of ESA (2011). Both estimates of unit electricity production costs are higher than the market price of electricity—by a margin of \$0.09–\$0.21 per kWh. For natural gas and CNG generation, the CDC calculations indicate that more than half of the O&M cost is the annual SoCal gas tariff service costs. Despite the studies' different assumptions, unit costs are similar. The closest cost estimates are found for natural gas production with a centralized facility using a raw-biogas pipeline from on-farm digesters: Scenario 4 for ESA (ESA-S4) and Scenario 2 for CDC (CDC-S2). The unit cost per 1,000 cubic feet of natural gas is \$20.52 for ESA-S4 and \$20 for CDC-S2.

Under all scenarios considered in both studies, including the scenario of co-digestion (ESA-S2), digester projects incur substantial losses. Judging from the shortfalls from the current price, production costs are double or even triple the market price of electricity per unit. These shortfalls are even worse for natural gas production. Production costs for gas are five times the market price of natural gas. Interconnection costs of bio-methane injection represent a major cost for natural gas production. To meet an 18% of IRR, the current revenue has to be increased by 22% to 110%. According to calculation made by CDC (2013), government grants equivalent to these revenue increases are 60% of capital costs for electricity generation, 87% for natural gas injection, and 68% for vehicle fuel production.

The main implication of both studies is that a centralized system of bio-energy production based on dairy digesters, even when scale economies are fully incorporated, is not economically viable without very large subsidies.

[20] The model can be accessed from http://biomass.ucdavis.edu/tools/.
[21] Dusault (2008) examines the possibility of a centralized digester system in Sacramento County. The considered farm would have a total of less than 2,000 cows.
[22] Feed-in tariff programs are designed for small-scale electricity generators. The 1 MW at maximum is expanded to 3 MW under SB 1122.
[23] CDC (2013) published only the revenue indicator; the other two indicators were derived using the results provided in CDC (2013).

Table 12. Economic Feasibility of Centralized Dairy Digester System in California's Central Valley: Comparison of Two California Studies: ESA (2011) and CDC (2013)

Study and scenario	Description of scenario	Total energy production	Capital cost ($mil)	% of O&M	Energy cost	Shortfall from current energy price	Revenue needed for IRR=18%[a]	Comments
ELECTRICITY GENERATION								
ESA (2011) ESA-S1	1,000 cows, manure only, electricity generation	744 MWh/yr	1.5	4	$0.28/kWh	$0.21/kWh	106%	Revenue includes carbon credits at $2.25/MTCO2e (14% of revenue).
ESA (2011) ESA-S2	1,000 cows, co-digestion, electricity generation	1,488 MWh/yr	1.7	8	$0.17/kWh	$0.09/kWh	49%	Revenue includes carbon credits (10% of revenue), tipping fees (22% of revenue), and digestate sale.
CDC (2013) CDC-S1	50,000 cows, electricity generation	50,839 MWh/yr	69	2	Min $0.18/kWh[b]	$0.10/kWh, estimated[c]	31%	Revenue includes carbon credits at $10/MTCO$_2$e.
BIOGAS GENERATION (Natural gas and CNG)								
ESA (2011) ESA-S3	10,000 cows, on-site biogas production, upgraded and pipeline injection	94.4 mil cu.ft (12,600 MWh)	9.7	7.5	$10.79/1,000 cu.ft	$6.62/1,000 cu.ft	48%	Revenue includes carbon credits, digestate sale, and renewable energy credits.[d] Costs on biogas upgrade, test system, and utility connection amount to 41% of capital cost.

ESA (2011) ESA-S4	10,000 cows, off-site biogas produced by on-farm digesters; gas pipelined to centralized facility for upgrade and pipeline injection	94.4 mil cu.ft (12,600 MWh)	16.2	7.5	$20.52/1,000 cu.ft.	$16.35/1,000 cu.ft.	110%	Revenue includes similar credits as in the case of off-site biogas production.[d] Each farm requires a costly digester. Biogas is transferred by pipeline from 8 off-site farm digesters.
CDC (2013) CDC-S2	50,000 cows, biomethane (natural gas) generation	519.4 mil cu.ft	38.8	11 (e)	Min $20/1,000 cu.ft.[b]	$16/1,000 cu.ft, estimated[c]	57%	Revenue includes carbon credits and renewable energy credits—available under the California Renewable Portfolio Standard.[d]
CDC (2013) CDC-S3	50,000 cows, CNG generation for vehicle fuel	519.4 mil cu.ft	38.8	11 (e)	Min $23/1,000 cu.ft.[b]	$19/1,000 cu.ft, estimated[c]	22%	Revenue includes additional credits available for vehicle fuel (Low Carbon Fuel Standard and Renewable Identification Numbers).[d]

Sources: ESA (2011) and CDC (2013).

[a] The appropriate IRR to initiate the investment is considered to be18.5% in ESA and 18% in CEC.

[b] CDC provides no unit costs. They are conjectured by dividing total annual operating cost by production. Given the 25% of equity financing assumption, these estimated figures are presented as minimum production costs.

[c] The shortfall from market price is calculated by using $0.08/kWh for electricity and $4/1,000cu.ft for natural gas.

[d] Renewable energy credits, available for renewable electricity, can be rewarded when bio-methane is used to generate electricity.

[e] 92% of O&M is for biogas gathering, conditioning, and upgrading services.

Final Remarks on the Economics of GHG Mitigation for Livestock in California

Most economic research on GHG emissions reductions from livestock production relate to dairy digesters. This report noted that little work on enteric fermentation has been conducted and that recent work attempts to use changes in feeding practices to show the very high per-unit costs of GHG emissions reductions.

The costs of covering anaerobic lagoons and flaring methane gas may be quite low per unit of methane emissions mitigated. Therefore, this option may be worthy of serious consideration and study. Its financial costs and complexity are much lower than those of dairy digester adoption, and its emissions mitigation may be significant. Other complications need to be explored, and full feasibility studies are needed.

In 2008, 21 anaerobic digester systems were operating in California, but by 2009 only 15 were operating; 6 had closed due to financial difficulties (ESA 2011). As of May 2013, 11 digesters—10 on dairy farms—were operating in California. Despite technology improvements, accumulated technical knowledge and experience, continued subsidies, increased incentives for renewable energy sources, and continued climate and scale advantages, anaerobic digesters failed to be widely adopted in California, suggesting that there are serious problems with the economic prospects of digester projects here.

Most studies agreed that under recent market and related policy conditions, digester systems are not likely to be viable, suggesting that even larger direct government subsidies or significant payments for carbon credits are needed to induce dairy farmers to adopt digesters. A number of studies have evaluated forms of carbon credits as a potential source for new incentives. The majority of these studies indicate that even under favorable conditions for capital investment, relatively high carbon offset prices would be required to provide an incentive sufficient for farmers to adopt digesters on a widespread basis. Carbon incentives have not yet provided sufficient added revenue.

This review of studies on dairy digesters indicates the importance of site-specific factors in determining the viability of digester projects. For example, California studies of 10 digester operations indicate the importance of local electricity markets. Electricity prices per kWh varied from \$0.05 to \$0.15 at the retail level and from \$0.03 to \$0.11 at the wholesale level in the Central Valley alone. The importance of site-specific and farm-specific features in the viability of digester projects contributes to the lack of usefulness of a financial profile of a digester project for a "typical" farm. The resulting uncertainty appears to be a significant issue for potential adopters. The difficulties associated with developing a profile of the "typical" adopter also mean that aggregate studies based on simulations may mischaracterize the costs of using dairy digesters to reduce GHG emissions, particularly if the studies cannot appropriately create distributions of the key components that cause costs and returns to vary.

Using the available studies to infer future costs and benefits of dairy digester projects is complicated by the wide variation in study assumptions and farm conditions. The results obtained from studies of the farms that continue to operate digesters also suffer from selection bias. Digester systems examined in *ex post* case studies were self-selected or pre-screened by design to have had at least a projection of economical viability, even if in practice and contrary to several of the studies that examined them, many soon failed. Such systems may be unrepresentative in the ways that their experiences cannot be transferred to other farms. In addition, the potential for selection bias means that parameters estimated from studies may not provide reliable information about the potential performance of a typical hypothetical farm.

One clear finding, however, is that the establishment and operation of digesters exhibits considerable scale economies. This finding is important because herd size continues to increase, and most cows are already in relatively large herds. Of course, even on large farms, digesters have not generally been a positive contributor to farm profits.

Development of an anaerobic digester system represents a major investment and differs from other investments that dairy farms regularly make. A digester investment involves uncertainty outside the scope of dairy market variability with which farmers are familiar. Issues outside farmers' control that affect the payoff include unfamiliar technical specifications and operational details, variable energy prices, unexpected shifts in energy policy, rapidly evolving environmental regulations and policy, and uncertain carbon prices.

For an investment for which profitability depends primarily on subsidies and regulations, lack of policy clarity is particularly discouraging. To be economically feasible, digesters in California must be designed to reflect regulatory constraints, operate efficiently through the use of recovered heat and co-digestion, capture all potential revenue streams, and secure power purchase agreements at favorable prices. Achieving these conditions is extremely challenging, which is why there are so few successful dairy digesters in the state.

CONCLUSIONS

Table 13 summarizes the costs and mitigation potential of GHG mitigation options in California's agricultural sector based on the publications and reports reviewed in this report.

Use of findings from the studies reviewed in this report is subject to three important caveats. First, climate change and the GHG emissions concerns are global but we focus solely on California. Reducing emissions within California can have only negligible direct effects on total global GHG emissions. Moreover, if agricultural production declines in California as a result of mitigation efforts, replacement production in other places could imply higher global emissions than before the efforts commenced.

Second, a vast literature on agricultural GHG emissions focuses on long-term (often 100-year) carbon sequestration related to dryland cropping systems and shifts of land from forestry. These foci are less directly relevant to issues of California agriculture. Little economic research has examined GHG mitigation for tree and vine crops, and no economic studies on these crops were specifically relevant to the California situation.

Third, no economic studies that focus on California have developed full life-cycle and crop livestock linkage models that could be used to trace system-wide emissions implications of a change in one set of practices or changes across commodities. Further extension is even possible by considering processing, transport, and specific non-farm impacts.

These caveats mean that this report's findings, which reflect the state of the literature, are necessarily tentative.

Table 13. Cost of Mitigation Options and Policy Instruments and Corresponding Mitigation Potential

Mitigation Option or Policy Instrument (tax or credit)	Agricultural Sector	Marginal Cost of Mitigation or Incentive Price ($/MTCO2e)	GHG Mitigation Potential (MTCO2e)	% of Total Ag GHG Emissions (2009)[a]	% of GHG Emissions from Appropriate Agricultural Sector (2009)[a]	Source
C-emission tax or credit	Crops - California (Central Valley)	$5	1,400,000	4.4%	15.5%	Garnache et al. (2013)
C-emission tax or credit	Crops - California (Central Valley)	$10	1,900,000	5.9%	21.1%	Garnache et al. (2013)
C-emission tax or credit	Crops - California (Central Valley)	$20	2,600,000	8.1%	28.8%	Garnache et al. (2013)
C-emission tax or credit	Crops - California (Central Valley)	$30	3,100,000	9.7%	34.4%	Garnache et al. (2013)
Anaerobic digestion - dairy	Manure management - US National	$ -	770,000	0.2%	2.4%	Gloy (2011)
Anaerobic digestion - dairy	Manure management - US National	$5	2,590,000	0.6%	8.2%	Gloy (2011)
Anaerobic digestion - dairy	Manure management - US National	$10	7,910,000	1.7%	25.0%	Gloy (2011)
Anaerobic digestion - dairy	Manure management - US National	$15	11,590,000	2.5%	36.6%	Gloy (2011)
Anaerobic digestion - dairy	Manure management - US National	$20	13,340,000	2.9%	42.1%	Gloy (2011)
Anaerobic digestion - dairy	Manure management - US National	$25	15,570,000	3.4%	49.1%	Gloy (2011)
Anaerobic digestion - dairy	Manure management - US National	$30	16,800,000	3.7%	53.0%	Gloy (2011)
Anaerobic digestion - dairy	Manure management - US National	$35	18,030,000	3.9%	56.9%	Gloy (2011)
Anaerobic digestion - dairy	Manure management - US National	$40	18,650,000	4.1%	58.8%	Gloy (2011)
Anaerobic digestion - dairy	Manure management - US National	$45	19,000,000	4.1%	59.9%	Gloy (2011)

Anaerobic digestion - dairy	Manure management - US National	$50	19,220,000	4.2%	60.6%	Gloy (2011)
Anaerobic digestion - dairy	Manure management - US National	$100	21,210,000	4.6%	66.9%	Gloy (2011)
Anaerobic digestion - dairy	Manure management - US National	$150	22,120,000	4.8%	69.8%	Gloy (2011)
Anaerobic digestion - dairy	Manure management - US National	$200	22,160,000	4.8%	69.9%	Gloy (2011)
Wheat - N fertilizer reduction (10%)	Crops - California	$2	Not significant	Not significant	Not significant	ICF International (2013)
Corn - reduced till to no till	Crops - California	$16	39,000	0.1%	0.4%	ICF International (2013)
Corn - conventional till to no till	Crops - California	$20	39,000	0.1%	0.4%	ICF International (2013)
Wheat - reduced till to no till	Crops - California	$63	40,860	0.1%	0.5%	ICF International (2013)
Wheat - conventional till to no till	Crops - California	$106	36,320	0.1%	0.4%	ICF International (2013)
Cotton - reduced till to no till	Crops - California	$542	48,060	0.1%	0.5%	ICF International (2013)
Cotton - conventional till to no till	Crops - California	$1,178	42,720	0.1%	0.5%	ICF International (2013)
Covering anaerobic lagoons (>2500 cows)[b]	Manure management - California	$6	2,500,000	7.8%	24.2%	ICF International (2013)
Covering anaerobic lagoons (>1000 cows)[b]	Manure management - California	$6.5	4,800,000	15.0%	46.4%	ICF International (2013)
Covering anaerobic lagoons (>500 cows)[b]	Manure management - California	$8	6,100,000	19.0%	59.0%	ICF International (2013)
Covering anaerobic lagoons (>200 cows)[b]	Manure management - California	$9	6,700,000	20.9%	64.8%	ICF International (2013)
N fertilizer reduction - 4c	Crops - Yolo county,	$9-46	2,791	1.9%[c]	1.9%	Mérel et al. (2013)

N-tax	California					
N fertilizer reduction - 8c N-tax	Crops - Yolo county, California	$17-91	4,903	3.3%[c]	3.3%	Mérel et al. (2013)
N fertilizer reduction - 12c N-tax	Crops - Yolo county, California	$26-137	6,706	4.5%[c]	4.5%	Mérel et al. (2013)
N fertilizer reduction - 16c N-tax	Crops - Yolo county, California	$34-182	8,481	5.7%[c]	5.7%	Mérel et al. (2013)
Enteric emission reduction-dairy	Enteric fermentation - California	$244	198,000	0.6%	2.1%	Moraes et al. (2013)
Enteric emission reduction-dairy	Enteric fermentation - California	$544	1,320,000	4.1%	14.2%	Moraes et al. (2013)
Enteric emission reduction-dairy	Enteric fermentation - California	$2,270	1,584,000	4.9%	17.1%	Moraes et al. (2013)

Sources: U.S. data—EPA (2011); California—CARB (2011).
Note: Note the applicable agricultural sector—United States, California, or Yolo County, California—for which cost and mitigation potential were given. A range for the nitrogen (N) fertilizer reduction option is presented, because a MTCO2e-based tax equivalent to the N-tax (per lb) depends on the emissions factors (both direct and indirect) used in the conversion calculation.
[a] This column applies to California, Yolo County, or the United States, depending on the study.
[b] Results apply to dairies with the number of cows greater than the stated limit.
[c] Yolo County has very few livestock—hence the assumption of zero emissions from the livestock.

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