Carbon Emissions, Renewable Electricity, and Profits: Comparing Policies to Promote Anaerobic Digesters on Dairies

Nigel Key and Stacy Sneeringer

Anaerobic digesters can provide renewable energy and reduce greenhouse gas emissions from manure management. Government policies that encourage digester adoption by livestock operations include construction cost-share grants, renewable electricity subsidies, and carbon pricing (offset) programs. However, the effectiveness and efficiency of these policies is not well understood. For the U.S. dairy sector, we compare predicted digester adoption rates, carbon emission reductions, renewable electricity generation and sales, and net returns and social benefits of several policies. We find that a carbon pricing policy provides the greatest net social benefit for a range of assumptions about the benefits of carbon reductions and renewable energy.

Key Words: anaerobic digester, carbon offsets, climate change, dairy, methane, renewable electricity, subsidy

Anaerobic digesters that collect and burn methane from manure can provide numerous benefits to livestock producers and the environment. Digesters can supply a renewable source of electricity, reduce odors from manure and the potential for surface water contamination, and recycle manure solids for animal bedding material. By burning methane, digesters also can reduce greenhouse gas (GHG) emissions from manure. Despite these benefits, anaerobic digesters have not been widely adopted in the United States. In 2011, there were only 167 operating digester systems, 137 on dairies and 23 on hog operations (AgSTAR 2011).

Rising fuel prices and an intensifying desire among citizens and government bodies to employ renewable energy and reduce carbon emissions have renewed interest in policies that encourage livestock producers to capture methane emissions from manure to produce energy. A 2010 joint statement by the Environmental Protection Agency (EPA) and the U.S. Department of Agriculture (USDA) announced their intention to provide \$3.9 million to encourage farm adoption of biogas recovery systems that generate energy from manure-based methane (EPA 2010a). EPA estimates that 2,645 dairy farms could feasibly generate 6.8 million megawatt hours (MWh) of electricity annually from biogas (EPA 2010b). In addition, cap-and-trade climate legislation recently debated in Congress, such as H.R. 2454, the American Clean Energy and Security Act of 2009 (www.govtrack.us/congress/bills/111/hr2454), could encourage adoption of farm-based methane digesters because such legislation would allow producers to sell offsets in the carbon market proposed in the act. Under such a system, livestock producers who reduce methane emissions via digester adoption could sell carbon offsets to other GHG emitters such as electric utilities, which face emission caps.

Despite an apparent political desire to encourage biogas capture at livestock facilities, little of the empirical research done so far on how to implement those goals has accounted for costs and benefits or examined the likely success and impacts of policies proposed. An EPA study that identified the 2,645 dairies that would find it "profitable" to adopt digesters explicitly noted that it did not include a cost analysis (EPA 2010b, p. 3) and instead defined profitability on the basis of the size of the

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operations and the manure management methods they employed. Other studies have examined the economic attractiveness of digesters for particular types of farms (Lazarus and Rudstrom 2007, Leuer, Hyde, and Richard 2008, Stokes, Rajagopalan, and Stefanou 2008, Bishop and Shumway 2009). Though such studies are useful, they may not be sufficiently generalizable to predict nationwide adoption levels. Only Gloy (2011) and Key and Sneeringer (2011a, 2011b) have considered national rates of adoption of digester systems, but those studies focused primarily on climate change policies related to carbon pricing and did not explore alternative policies to promote digester use.

Policies currently in place that aim to make digesters more profitable for operators include: (i) construction-cost subsidies (e.g., loan guarantees, accelerated depreciation, cost-share programs); (ii) supports that raise demand for, and hence the price of, digester-generated renewable electricity; and (iii) carbon pricing mechanisms (offsets). We estimate and compare the effects of these three policy approaches in terms of operators' net returns, adoption of digester systems, generation of electricity, GHG emissions, and program costs. We also estimate the net social benefit of each policy under various assumptions about the value of renewable energy and GHG-emission reductions. The analysis highlights these policies' potential long-run distributional and unintended consequences in terms of increased carbon emissions.

Our estimates are based on a model of digester profitability that incorporates farm size, manure management method, electricity generation and use, capital required to establish a digester, variable costs, and electricity and carbon offset prices. We estimate how alternative policies affect producers' profits and, consequently, their decisions about adopting biogas recovery systems. The model is parameterized using information from multiple case studies and vendor price quotes so that it reflects actual farm-level costs and producers' experiences with energy production using digester technology. We use state-level data from government sources to account for regional variations in electricity prices and methane emissions. We apply the model to data from the 2005 dairy version of the nationally representative Agricultural and Resource Management Survey (ARMS) (ERS 2005) by USDA.

The results from our model illustrate the relative cost efficiency and outcomes of each policy. Relative efficiency depends on the desired objective: we find that a carbon pricing policy such as an offset program is the most cost-effective mechanism for reducing GHG emissions from dairy manure management, that a construction cost-share program is the most efficient for stimulating renewable electricity generation, and that a renewable electricity subsidy is most cost-effective for inducing the greatest amount of electricity for sale from digesters. The findings indicate that a carbon pricing program provides the greatest net social benefit under a variety of assumptions about the external benefits of carbon reductions and renewable energy; a construction cost-sharing program provides the second greatest social benefit. We also find that electricity subsidies and construction cost-share programs induce some individual farms to switch from aerobic to anaerobic digestion and therefore increase manure methane emissions; however, all policies result in a net decrease in carbon emissions.

Methane Creation and Capture

Manure mixed with water is often stored in lagoons, ponds, tanks, or pits, which creates anaerobic (without oxygen) conditions. The anaerobic decomposition of livestock or poultry manure produces a biogas that contains about 60 percent methane.[1] When manure is handled as a solid or deposited on fields, it tends to decompose aerobically (with oxygen) and produce much less methane. The quantity of methane released also depends on climate (temperature and rainfall) and the conditions under which manure is managed (oxygen level, water content, pH, and nutrient availability).

Methane is a potent GHG; one ton of methane has 25 times the global warming potential of 1 ton of carbon dioxide (CO_2). In 2008, methane emissions from manure management were responsible for about 10.5 percent of GHG emissions from the U.S. agricultural sector.[2] Agriculture as a whole was responsible for 6.1 percent of total U.S. GHG

[1] The remainder of the emissions consists primarily of carbon dioxide (about 40 percent) plus small amounts of toxic gases that include hydrogen sulfide, ammonia, and sulfur-derived mercaptans.

[2] Livestock also emit methane from enteric fermentation produced during digestion. In 2008, more than three times as much methane was released from enteric fermentation as from manure management (EPA 2010b, Table 2-8).

emissions (EPA 2010c, p. 2–12).[3] Producers of dairy cattle and swine, who often use anaerobic manure management systems, were responsible for 43.1 percent and 43.6 percent, respectively, of total methane emissions from manure management (EPA 2010c, Table 6-3). Beef cattle, sheep, poultry, and horses were collectively the source of only 13.3 percent of total manure methane, mainly because manure from these animals is usually handled in aerobic conditions.

Biogas recovery systems, known as anaerobic digesters, methane digesters, biodigesters, and methane recovery systems, capture methane from anaerobic manure storage facilities.[4] Such systems collect manure, optimize it for the production of methane by adjusting temperature and water content, and then capture the biogas it produces. The captured methane can be burned for heat or, with the addition of a generator, can be used to produce electricity. Burning one ton of methane reduces its global warming potential from the equivalent of 25 tons of carbon dioxide to 1 ton.

Anaerobic digesters are generally added to two types of manure storage: lagoon-based systems in which a cover is placed over an earthen storage pond and pit-based systems in which manure is processed through a heated tank to encourage methane production. Operations that use aerobic manure storage systems (such as a stacking slab) and those that do not store manure generally cannot produce enough concentrated methane from manure to make a digester feasible.

Digesters fall generally into one of three categories: complete-mix, plug-flow, and covered-lagoon systems. A complete-mix digester is a large concrete or steel container, usually circular in shape. A plug-flow digester is often in the shape of a trough and is built below ground level with an air-tight expandable cover. Manure is collected daily and added to one end of the trough, and that "plug" slowly pushes manure already deposited there down the trough. A covered-lagoon digester is an earthen lagoon with a tightly fitted impermeable cover that rests on the surface of the lagoon. The industrial fabric cover collects biogas as it is produced from the organic waste.

Methane digesters have not been widely adopted because the cost of constructing and maintaining the systems typically exceeds the benefits they provide to farm operators. The cost of construction of a digester system is significant. Capital is required to design and install the manure collection and pretreatment system, the lagoon cover or tank, generator, and the utility connection. Our review of 23 case studies and vendor quotes (Key and Sneeringer 2011b) suggests that the cost for a system ranges from $204,000 to $3.9 million and an average of $990 per cow.[5] Covered-lagoon digesters are generally less expensive to construct than complete-mix and plug-flow systems but lagoon digesters generally are not feasible in cooler climates.

Determinants of Digester Adoption

The decision to adopt digester technology depends on a number of factors, including the manure management method already in place, the size of the producer's operation, the start-up and ongoing costs of the technology, the price of electricity, on-farm electricity expenditures, the amount of electricity that can be generated, and the physical capacity to sell electricity not used on the farm.

The existing manure management method and the operation's size are the two primary factors that affect adoption. A concentrated supply of methane is necessary for the digester to function effectively. Consequently, only operations that already store manure in anaerobic conditions, thus generating significant quantities of methane, are viable candidates for biogas recovery systems without conversion to a different management method. Data from the ARMS indicate that about 50 percent of U.S. dairies have anaerobic manure systems, 16 percent use aerobic systems (open slabs or covered sheds), and 34 percent report no manure storage system.

Anaerobic manure management systems are least common among small-scale operations. For example, only 46 percent of dairy operations that have less than 250 cows use anaerobic systems while 73–88 percent of larger operations do (Table 1). Larger operations also are more likely to

[3] This total does not include emissions from inputs to agricultural production that are attributed to other sectors, including fertilizer production, transportation, and electricity generation.

[4] Anaerobic digestion occurs when bacteria break down (or "digest") biodegradable material such as manure without the presence of oxygen.

[5] These values are in 2009 dollars. For details regarding the case studies and vendor quotes, see Key and Sneeringer (2011b).

Table 1. Characteristics of U.S. Dairy Operations

Category	Number of Farms in Category	Total Gross Value of Production (percent)	Operations with Lagoons or Pit Manure Systems (percent)	Operations with Lagoons (could also have pits) (percent)	Total Methane Emissions (percent)	Electricity Use per Head (kWh per year)	Electricity Price ($ per kWh)
All farms	52,237	100	42	11	100	1,048	0.069
Number of Head							
Greater than 2,500	248	13.0	55.6	48.0	19.7	494	0.078
1,000–2,499	917	18.3	63.5	38.9	20.9	723	0.081
500–999	1,615	14.1	71.3	41.5	18.4	743	0.079
250–499	3,040	13.5	72.8	40.0	16.0	775	0.068
Less than 250	46,417	41.1	38.0	6.9	25.0	1,085	0.068
Region							
West	6,095	33.3	56.5	38.1	43.1	893	0.058
Midwest	28,438	36.4	40.2	5.8	26.0	1,102	0.064
South	4,034	9.2	53.0	27.1	15.6	791	0.065
Northeast	13,670	21.1	34.3	3.8	15.3	1,080	0.085

Notes: Data are from the 2005 ARMS cost-of-production survey for dairies (ERS 2005). All dollar values are in real 2009 terms. kWh = kilowatt hours.

have lagoon systems, which have a greater rate of initial methane emission than pit systems.

Farm size is an important determinant of digester profitability because it is associated with the manure management method chosen and because of economies of scale in constructing and maintaining methane digesters (Key and Sneeringer 2011b). Additionally, larger operations can spread the fixed transaction cost associated with selling electricity or certifying and marketing offsets over a larger revenue base.

While an operation's size and existing manure storage system largely determine the technical feasibility of a digester, the farm's electricity use and generation capacity and the price of electricity are key factors that determine how much the operator will benefit financially from generating and selling surplus electricity.

The ARMS data indicate that an average dairy (one maintaining 154 cows) uses 128,918 kilowatt hours (kWh) of electricity per year, or 837 kWh per head. However, energy use varies with farm size with larger operations using substantially less electricity per head than smaller ones (Table 1). There is also significant variation in electricity use across regions: on average, dairies in the Midwest used 1,102 kWh of electricity per head compared to the South, which used only 791 kWh per head, a reflection of differences in the size of the average operation and the climate in each region.

Another determinant of digester profitability is whether the producer can sell generated electricity. Without the ability to sell surplus electricity to the grid, the only benefit from on-farm electricity generation is a reduction in the operation's energy costs. Another key determinant of digester profitability is the price received for surplus electricity that is sold. As we discuss in the next section, the price depends to some extent on policies established at the local, state, and federal level.

Climate also affects profitability. Anaerobic systems located in warmer climates generate more methane (methane levels increase with ambient temperature) and thus more electricity than digesters in cooler climates.

Policies That Affect Digester Profitability and Adoption

Construction Cost Subsidies

Policies now in place subsidize the cost of construction of methane digesters in a number of ways, including grants (e.g., USDA Rural Energy for America Program grants); tax credits such as the Renewable Electricity Production Tax Credit; accelerated depreciation through the Accelerated Cost Recovery System, which allows qualifying renewable energy systems to be depreciated using an accelerated schedule for tax purposes; and property and sales tax exemptions (usually at the state level).

Typically, cost-share programs are capped. For example, grants from the Rural Energy for America Program Grants / Renewable Energy Systems / Energy Efficiency Improvement Program (REAP/RES/EEI) cannot exceed 25 percent of the total cost of an eligible project and are limited to $500,000 for renewable energy systems.[6] Operators can sometimes obtain grants and subsidies from multiple sources, making it difficult to determine a broadly representative upper bound for cost-share programs.

Electricity Price Subsidies

Methane is a biogas that qualifies as a renewable energy source, so digester owners can obtain Renewable Energy Certificates (RECs) for the electricity they generate and supply to the grid. RECs represent claims to the environmental qualities of renewable electricity and are allocated to renewable energy providers by a certifying agency. Providers then feed the renewable energy into the electrical grid and sell the accompanying RECs in a market to individuals and organizations wishing to use renewable energy. Customers can purchase RECs regardless of whether they have access to renewable power through the local utility. These buyers effectively purchase renewable energy because the sellers—the producers of renewable energy—receive revenue from REC sales (DOE 2011). Hence, sales of RECs provide a production subsidy to generators of renewable electricity. The average unweighted residential price premium for RECs in 2010 was $18.90 per MWh, or about $0.02 per kWh (National Renewable Energy Laboratory 2010).

There are two main markets for RECs: compliance and voluntary. Compliance markets have been established in the 30 states that have a Renewable Portfolio Standard (RPS), sometimes called a Renewable Electricity Standard. The RPS is a mechanism that specifies how much of a utility's electricity must come from renewable energy sources. The utility can satisfy the RPS by purchasing RECs from renewable energy producers.

In 2009, the U.S. Senate Committee on Energy and Natural Resources passed federal legislation that would have required electric utilities nationwide to provide 15 percent of their electricity sales through renewable sources of energy by 2021. However, this legislation, the American Clean Energy Leadership Act, was not passed by the Senate. States have enacted a wide variety of renewable energy targets and technology standards. Take the two largest dairy states as examples. California's RPS requires that 20 percent of electricity be supplied by renewable sources in 2010 and increases the mandate to 33 percent in 2020 (California Energy Commission 2011). Wisconsin's goal is for 10 percent of its electricity to come from renewable sources by 2015 (Database of State Incentives for Renewable Energy 2010).

In voluntary REC markets, customers (usually corporations and households) voluntarily purchase renewable power. Generators of renewable energy located in states that lack a standard can sell their RECs in these voluntary markets, which typically provide lower prices than compliance markets.

Note that the seller's price for surplus electricity depends on whether the utility is subject to "net metering." Under net metering laws, a farm's electricity bill is based on a net monthly meter reading. When surplus electricity is being produced on the farm, the meter spins backward, effectively saving the extra electricity until it is needed. This system eliminates concerns regarding when the surplus electricity was produced and differential pricing over the course of a day. These net metering laws, which have been adopted in 40 states, vary widely from state to state (Database

[6] For more information on REAP/RES/EEI, see http://www.rurdev.usda.gov/rbs/busp/9006grant.htm.

of State Incentives for Renewable Energy 2010). For this study, we assume for simplicity that net metering is available for all operations.

Pricing Reductions of GHG Emissions

One way to mitigate GHG emissions from manure management is to pay farmers for reducing the amount of gases the operations emit. Farmers can be compensated directly with government payments or through carbon offset sales.

As with RECs, carbon offsets can be exchanged in compliance and voluntary markets. Compliance markets usually operate in conjunction with a cap-and-trade regime that places a legal limit on the quantity of GHG that can be emitted by regulated firms. To meet the emission targets, regulated firms can reduce their own emissions or purchase permits from other capped firms. Alternatively, firms can pay unregulated emitters such as livestock operations to reduce emissions by purchasing offsets.

Compliance markets have been established internationally, nationally, and regionally. Regimes that govern international compliance markets include the Kyoto Protocol and the European Union's Emissions Trading Scheme. In the United States, ten eastern states recently implemented the Regional Greenhouse Gas Initiative (RGGI), the first mandatory market-based effort to reduce GHG emissions. Under the RGGI, the capped sector (power generators) can purchase emission offsets from other sectors such as agriculture that reduce or sequester GHG emissions, including projects that reduce methane emissions from manure. In 2009, the U.S. House of Representatives approved climate change legislation (H.R. 2454, the American Clean Energy and Security Act of 2009) that, had it been signed into law, would have established a national cap-and-trade system and provided a further opportunity for farmers to sell offsets from reducing methane emissions.

Voluntary offset markets function outside of compliance markets and allow companies and individuals to voluntarily purchase carbon offsets. In some privately administered cap-and-trade systems, such as the Chicago Climate Exchange, reductions of methane emissions from livestock operations qualify as offset projects.

The revenue that a livestock operation can earn from offsets for a digester system depends on the type of manure storage and handling facility in use. Offset programs usually require documentation of a livestock operation's baseline emissions and certification that offsets are generated by reducing emissions to less than the baseline (the "additionality" requirement). Because the methane produced by an aerobic management system (or no management system) is much less concentrated than the methane from an anaerobic one, the aerobic operation's baseline emission level would be low. Digesters often increase the amount of methane produced from the manure, which would make it nearly impossible for an operation with an aerobic baseline to reduce emissions further using a digester. Consequently, only operations that had been using an anaerobic manure storage facility before creation of the offset market would be likely to qualify to generate offsets. This requirement largely limits the pool of potential offset market participants to swine and dairy operations that already have lagoon or pit systems. Operations with slab or shed manure systems or with no storage facilities would not generate sufficient methane to satisfy the "additionality" requirements for offset certification.

In the major international compliance markets, carbon offset prices have ranged from \$15 to \$30 per ton of carbon-dioxide-equivalent (CO_2e) emissions in the last decade.[7] There is a great deal of uncertainty about eventual carbon prices under a national cap-and-trade system. EPA estimated that H.R. 2454 would have resulted, in the near term, in a price of \$13 per ton of CO_2e emissions (EPA 2009).

Climate change legislation that would raise the price of carbon would be expected to increase the price of electricity. Regions that generate electricity using carbon-intensive methods would likely see larger price increases.

Empirical Framework

Gloy (2011) and Key and Sneeringer (2011a, 2011b) developed models of digester profitability to estimate the potential supply of carbon offsets from livestock manure management. Key and Sneeringer also explored the distributional implications of such a policy. Our modeling

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[7] Offsets are measured in tons of carbon-dioxide-equivalent emissions. Reductions in other GHG emissions such as methane are converted to an equivalent quantity of carbon dioxide based on the relative global warming potential of the gas.

approach is similar to those studies but incorporates several extensions.

Our basic approach is to estimate the net present value (NPV) of a digester project and then predict, using nationally representative data, adoption and other outcomes under various policy scenarios. NPV is the sum of future cash flows (e.g., revenue from electricity or carbon offsets minus capital and variable costs) over the life of the project discounted to present value. We assume that an operator adopts a digester if its NPV is positive.

The outcomes we are interested in comparing across policies include (i) the rate of digester adoption among producers, (ii) the quantity of electricity generated, (iii) the quantity of electricity sold, (iv) the percentage of producers using aerobic manure management systems who switch to anaerobic systems, (v) reductions in carbon emissions, and (vi) the net revenue generated by digesters.

We also estimate the net social benefit of each policy, which is defined as the private benefit plus the public benefit minus the policy cost. Private benefits are profits that accrue to digester adopters. Public benefits are the benefits to society that result from renewable electricity generation and from fewer carbon emissions.[8]

Investment Model

Operators' investment decisions depend on their present manure management methods.[9] An operator with an anaerobic system who is considering investing in a methane digester chooses the investment option that gives the highest expected NPV, which is given by:

a) $\sum_{t=0}^{T}\left[\left(RE_{isft}+RO_{isft}-CD_{ift}-CO_t\right)/\left(1+d\right)^t\right]$;

build a digester with an electricity-producing generator and sell offsets.

b) $\sum_{t=o}^{T}\left[\left(RE_{isft}-CD_{ift}\right)/\left(1+d\right)^t\right]$;

build a digester with an electricity-producing generator but do not sell offsets.

c) $\sum_{t=0}^{T}\left[\left(RO_{isft}-\gamma CD_{ift}-CO_t\right)/(1+d)^t\right]$;

build a digester without a generator, flare the collected methane, and sell offsets.

d) 0; do not build a digester.

RE_{isft} is the value of electricity generated by a digester and generator for operation i located in state s and using manure management facility type f in time t. RO_{isft} is the value of offset sales. CD_{ift} is the cost of the digester plus the electricity generator, and CO_t is the transaction cost to sell offsets. γCD_{ift} is the cost of a digester with no electricity generator where $0 < \gamma < 1$, T is the lifespan of the digester, and d is the discount rate. We assume there is no salvage value to the digester.

An operator who has an aerobic manure management system can switch to one of two anaerobic systems and adopt a digester but would not be eligible to sell offsets because of the additionality rules. Allow $f = 1$ to denote lagoon manure management, $f = 2$ to denote pit-based manure management, and W_{ift} to denote the cost of switching systems. The expected discounted stream of net revenue for each option is:

a) $\sum_{t=o}^{T}\left[\left(RE_{is1t}-CD_{i1t}-W_{i1t}\right)/(1+d)^t\right]$;

switch to a lagoon system and build a digester with an electricity-producing generator.

b) $\sum_{t=0}^{T}\left[\left(RE_{is2t}-CD_{i2t}-W_{i2t}\right)/(1+d)^t\right]$;

switch to a pit system and build a digester with an electricity-producing generator.

c) 0; do not switch manure management methods and do not build a digester.

The value of electricity generated by the digester, RE_{isft}, depends on time, on whether the quantity generated on-farm, E_{if}^{G}, is less than or greater than the quantity used on-farm, E_i^U, and on buying (retail) and selling (wholesale) prices of electricity, P_s^{ER} and P_s^{EW} respectively. In the

[8] Since we allow for the possibility of markets for renewable electricity credits and for carbon offsets, we use the term public rather than nonmarket to refer to the benefits associated with reductions in the externalities that result from renewable electricity and reduced carbon emissions.

[9] These decisions are illustrated in a decision tree in the Appendix (Figure A1), which is available online.

event that the selling price exceeds the retail price, the producer will sell all electricity generated and buy back any needed electricity at the retail price regardless of whether the operation generates more electricity than is used on-farm (this is equivalent to selling RECs for the full amount generated). If the buying price exceeds the selling price, the producer uses digester-generated electricity for all on-farm needs and sells any leftover electricity generated at the wholesale price:

$$(1)\ RE_{isft} = \begin{cases} 0 & if\ t = 0 \\ P_s^{EW} \cdot E_{isft}^G & if\ 1 \le t \le T \text{ and } P_s^{ER} < P_s^{EW} \\ P_s^{ER} \cdot E_{isft}^G & if\ 1 \le t \le T \text{ and } P_s^{ER} \ge P_s^{EW} \text{ and } E_{isf}^G \le E_i^U \\ P_s^{ER} \cdot E_i^U + P_s^{EW} \cdot \left(E_{if}^G - E_i^U\right) & \\ \quad if\ 1 \le t \le T,\ P_s^{ER} \ge P_s^{EW}, \text{ and } E_{isf}^G > E_i^U. \end{cases}$$

We assume that net metering laws are in effect and that the operation therefore can replace used electricity with generated electricity at the retail rate.

The selling price of farm-generated electricity will likely increase with the carbon price. For simplicity, the selling price of electricity is assumed to be proportional to the retail price: $P_s^{EW} = \theta^W + \eta$ where θ^W is the constant difference between retail and wholesale prices and η is the renewable electricity subsidy parameter that is varied in our policy simulations.

Electricity generation depends on the manure storage method and the quantity of manure produced. Since the quantity of manure produced is a linear function of the number of head of livestock, the quantity of electricity generated can be expressed simply as $E_{isf}^G = e_{sf} \cdot N_i$. For covered lagoon systems, the electricity generated per head, e_{sf}, depends on climate (which is characterized at the state level in the model). For plug-flow and complete-mix digesters, we assume that generation does not depend on climate because those digesters are generally heated.

The cost of the biogas system consists of the capital investment, K_{if}, at the beginning of the project ($t = 0$) plus maintenance and operating costs, V_{if}, for years 1 through T. The capital investment includes costs for design and construction of pumps, pits, heating systems, buildings, solid separators, effluent holders, generators, and power lines. Operations located in areas designated as "non-attainment" for ozone under the Clean Air Act face an additional scrubber cost, A, associated with the electricity generator in the initial year of operation. We allow the scrubber cost to be A in non-attainment areas and zero elsewhere. The cost of installing a digester without a generator is modeled as a portion, γ, of the overall digester-plus-generator cost. For operations that adopt a digester and a generator, let $\gamma = 1$. In the model, a share of the capital investment, $1 - \lambda$, is borne by a government cost-share program such that operators face only the capital cost multiplied by λ. Operating costs are therefore:

$$(2)\qquad CD_{ift} = \begin{cases} \lambda\gamma(K_{if} + A) & if\ t = 0 \\ \gamma V_{if} & if\ 1 \le t \le T \end{cases}$$

Since scrubber costs are associated only with electricity generation, those who adopt a digester and flare off the gas would not incur them (i.e., if $\gamma < 1$, then $A = 0$). Since operators can obtain subsidies from multiple sources, we do not model a grant cap in this analysis.

Capital investment costs, K_{if}, increase with the scale of the operation at a decreasing rate that depends on manure-management-specific parameters of a_f and b_f: $K_{if} = a_f \cdot (N_i)^{b_f}$. Annual variable costs, V_{if}, include the cost of maintenance and repairs. Following past studies, we assume that the variable cost is proportional to the quantity of electricity generated (which depends on the farm's size and manure handling facility): $V_{if} = v \cdot E_{if}^G = v \cdot e_{sf} \cdot N_i$.

For digester adopters, the revenue from selling carbon offsets is dependent on time, the price of carbon, and how much the methane emissions can be reduced below the baseline:

$$(3)\qquad RO_{isft} = \begin{cases} 0 & if\ t = 0 \\ \dfrac{24}{25} \cdot P^M \cdot M_{isft} & if\ 1 \le t \le T \end{cases}$$

where P^M is the price of the carbon offsets in dollars per ton of CO_2e emissions and M_{isft} is the quantity of methane produced. Revenue from offsets is zero when $t = 0$ since the digester has not yet been built. One ton of methane emitted to the environment has the global warming potential of 25 tons of CO_2 (Forster et al. 2007); however, burning that ton of methane reduces its global warming potential to 1 ton of CO_2. Hence, each ton of methane burned is equivalent to eliminating 24 tons of CO_2.

The quantity of methane produced before installing a digester varies according to the manure management method used. For operations with anaerobic systems, the quantity of methane produced in CO_2e, M_{isf}, is $N_i \cdot m_{sf} \cdot 25 \cdot 365 \cdot 0.001$ where N_i is the number of head of livestock and m_{sf} is the methane emission factor: kilograms of methane per head per day. The total amount of methane produced in kilograms ($N_i \cdot m_{sf}$) is multiplied by 25 tons of CO_2e per ton of methane, 365 days per year, and 0.001 tons per kilogram to express M_{isf} in tons of CO_2e. For operations with aerobic systems, we assume that the amount of methane produced by manure management is zero.

Changes in the amount of methane produced as a result of installing a digester also vary according to the original manure management method. The amount of methane reduction (M^R_{isft}) is

$$(4)\quad M^R_{isft} = \begin{cases} 0 & \textit{if } t = 0 \\ \dfrac{24}{25} \cdot M_{isft} & \textit{if } 1 \le t \le T \text{ and anaerobic} \\ -N_i \cdot m_{sf} \cdot 365 \cdot 0.001 & \textit{if } 1 \le t \le T \text{ and aerobic.} \end{cases}$$

Again, the amount of reduction when $t = 0$ is zero because the digester is not yet completed.

The transaction cost associated with selling carbon offsets includes the initial, one-time, fixed start-up cost for entering the offset market (Z^E) plus the ongoing annual cost of monitoring and verification (Z^V):

$$(5)\qquad CO_t = \begin{cases} Z^E & \textit{if } t = 0 \\ Z^V & \textit{if } 1 \le t \le T \end{cases}.$$

Finally, the cost of switching from aerobic to anaerobic manure management depends on the type of digester system, f, and the number of animals, N_i; those costs are only incurred in $t = 0$:

$$(6)\qquad W_{isft} = \begin{cases} \lambda(\varpi_f + \pi_f N_i) & \textit{if } t = 0 \\ 0 & \textit{if } 1 \le t \le T \end{cases}.$$

Policy Costs and Benefits

The cost of a policy that prices carbon emission reductions equals the discounted total value of the offsets provided by digester adopters:

$$(7)\qquad \sum_i D_i \left\{ \sum_{t=0}^{T} \left[\left(RO_{isft} \right) / (1+d)^t \right] \right\}$$

where D_i is an indicator variable equal to 1 if operator i adopts a digester. In an emission-trading scheme, the policy cost is borne by the capped industries and by consumers of the products produced by the capped industries. Alternatively, if carbon is priced using a government subsidy paid to farmers for reducing emissions, then the cost is borne by taxpayers.

The policy cost of subsidizing digester construction is incurred only in the first year and includes the capital investment, the cost of scrubbers, and switching costs. Allowing Q_i to be an indicator variable equal to one if operator i switches from an aerobic to an anaerobic system, the policy cost is

$$(8)\qquad \sum_i D_i (1-\lambda) \left[\gamma K_{if} + A + Q_i\, (\varpi_f + \pi_f \mathrm{N}_i) \right].$$

The policy cost of subsidizing the electricity selling (retail) price equals the subsidy rate, η, multiplied by the amount of electricity sold. We assume that the subsidy rate is constant over the life of the digester. Like the value of electricity generated by the digester, the electricity-subsidy cost depends on time and on how the quantity of electricity generated compares to the amount used on the farm and on the retail and selling (subsidized) prices for electricity. For each operator i in time t, the cost of providing the electricity subsidy, GE_{isft}, is

$$(9)\quad GE_{isft} = \begin{cases} 0 & \textit{if } t = 0 \\ \eta\, E^G_{isf} & \textit{if } 1 \le t \le T \text{ and } P^{ER}_s < P^{EW}_s \\ 0 & \textit{if } 1 \le t \le T \text{ and } P^{ER}_s \ge P^{EW}_s \text{ and } E^G_{isf} \le E^U_i \\ \eta \left(E^G_{isf} - E^U_{isf} \right) & \textit{if } 1 \le t \le T,\ P^{ER}_s \ge P^{EW}_s, \text{ and } E^G_{isf} > E^U_i. \end{cases}$$

The total cost of the electricity-subsidy policy is

$$\sum_i D_i \left\{ \sum_{t=0}^{T} \left[\left(GE_{isft} \right) / (1+d)^t \right] \right\}. \tag{10}$$

If the electricity price premium is derived from a mandated RPS, the policy cost is borne by the utility company and electricity consumers.

The total social benefit of a policy is the private benefit plus the public benefit. The private benefit is the discounted stream of net revenue previously defined. The public benefit is derived from use of renewable electricity and reductions in carbon emissions. Let P^{EX} be the externality cost per kWh associated with conventionally generated electricity. Since the renewable, digester-generated electricity replaces conventionally generated electricity, the total social benefit from digester-produced electricity is

$$\sum_i D_i \left\{ \sum_{t=0}^{T} \left[\left(P^{EX} E_{if}^{G} \right) / (1+d)^t \right] \right\}. \tag{11}$$

The social benefit from reduction of carbon emissions is

$$\sum_i D_i \left\{ \sum_{t=0}^{T} \left[\left(P^{CX} M_{isft}^{R} \right) / (1+d)^t \right] \right\}. \tag{12}$$

where P^{CX} is the externality cost per ton of CO_2e emissions.

Risk and Uncertainty

The NPV (net present value) approach in this model is deterministic in the sense that real prices are assumed to be known by the operator and constant throughout the economic life of the digester. In fact, many of the benefits and costs associated with a digester are uncertain and variable. For example, the price of electricity—both retail and wholesale—is likely to fluctuate with global economic conditions and policy changes that are difficult to predict (Leuer, Hyde, and Richard 2008, Stokes, Rajagopalan, and Stefanou 2008). Similarly, carbon offset prices have varied dramatically over time and estimating them in the future is difficult.[10] Chicago Climate Exchange's (CCX's) carbon price has ranged from \$1 to \$7 per ton since 2004 but traded at its floor price since 2009 (CCX 2009). The average price for carbon allowances in the RGGI (2008–2011) has ranged from \$1 to \$3 per ton since its inception in 2008. EPA (2009) estimated that H.R. 2454 would have generated a price of \$13 per ton in the near-term. There is also uncertainty about the variable costs of operating a digester and the digester's methane and electrical output, which could fluctuate from year to year depending on system reliability, weather, and mechanical failures.

Our NPV framework also assumes that investors are risk-neutral, a reasonable assumption when analyzing decisions by governments and large corporations but less realistic in the case of small-scale investors. For most farmers, an investment in a digester represents a substantial income risk that cannot be mitigated through insurance markets or a diversified portfolio. Consequently, farmers are likely to place less value on risky digester returns compared to equivalent risk-free returns and a NPV analysis would overestimate net benefits and digester adoption.

Data and Parameters

We estimate parameters for electricity generation and digester costs using information from multiple case studies drawn from compiled and individual project descriptions and from a data set of vendor quotes for prospective digester projects. These case studies and vendor quotes are described in Key and Sneeringer (2011b). Information on electricity prices by state and methane emission factors comes from government sources. We list the model parameters, further describe the data, and provide citations in the Appendix.

The model predicts digester adoption for the farms surveyed in the 2005 Dairy Production Practices and Costs and Returns Report, a portion of the ARMS, the nationally representative survey collected jointly by USDA's National Agricultural Statistics Service (NASS) and Economic Research Service (ERS). The ARMS contains information on each farm's head of livestock, type of manure management system, location, and cost of electricity consumed.

Public benefits from digester-generated electricity can be broken down into benefits related to climate and all other benefits. We estimate that the public benefits that are not

[10] Europe's experience illustrates the difficulty in predicting prices—in 2006, Europe's Emissions Trading System suffered a precipitous drop in the carbon price when policymakers initially misjudged the quantity of allowances to allocate. The market subsequently stabilized.

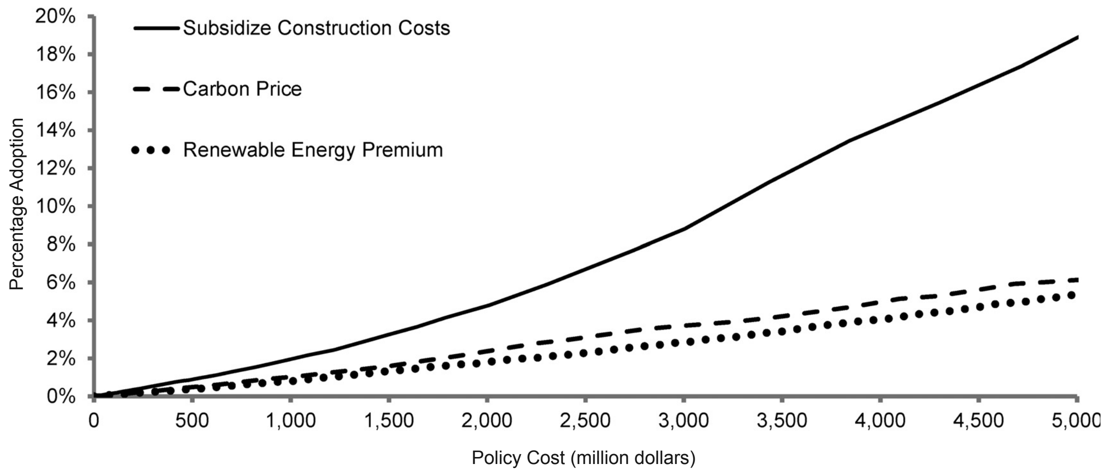

Figure 1. Adoption Rates by Policy Cost

Note: Policy cost is the discounted cost for a 15-year project.

related to climate are worth \$0.016 per kWh, a figure that is based on estimates of the health and environmental benefits of renewable digester-generated electricity replacing electricity generated using coal and natural gas.[11] We estimate that the climate-related benefits are worth \$0.017 per kWh based on the value of associated reductions in carbon emissions.[12] Together, then, digester-generated electricity provides a public benefit worth \$0.033 per kWh.

For the baseline scenario, we use \$28 per ton of CO_2e as the value to society from reducing GHG emissions. This is the mean value reported by Tol (2008) in his meta-analysis of 211 studies that estimated the social cost of carbon emissions.[13] We use a sensitivity analysis to illustrate how different values for the benefits of renewable energy and reductions in carbon emissions alter the net social benefit derived from three policies: construction cost-sharing, renewable electricity subsidies, and carbon offsets.

Results

We first compare the cost-effectiveness of the three digester-support policies over a range of policy costs in terms of achieving the particular objectives of the policies. If the overriding goal is the most widespread adoption of digesters possible given a particular cost, the construction-cost subsidy is the most effective policy (see Figure 1); the carbon price subsidy and the renewable energy premium both result in about half as many farms adopting digesters. Under all three policies, the average size of farms that adopt declines as the digester adoption rate increases.[14] Because the construction-cost subsidy induces the most farms to adopt, that policy results in a substantially smaller average farm size for adopters.

Figure 2 compares the amount of electricity produced and sold (not used on the farm) over the 15-year life of the digester for each policy. The quantity of electricity generated per dollar declines as the policy cost increases because operators of increasingly smaller farms choose to adopt as the

[11] The non-climate-related external costs per kWh for coal and natural gas reported by the National Research Council (2010) in 2007 dollars are \$0.0320 and \$0.0016 per kWh respectively. We use the percentage of U.S. electricity generated by these two sources in 2007—48 percent for coal and 22 percent for natural gas—to compute a weighted average of the external cost of electricity per kWh after assigning a zero value for the external cost related to other sources of electricity. This yields an average external cost for benefits not related to climate of \$0.016 per kWh.

[12] The value of carbon emission reductions depends on the carbon content of the electricity replaced and the social cost of carbon. According to the Department of Energy (DOE), the United States in 2000 emitted an estimated 1.341pounds of CO_2 per kWh (DOE 2000), resulting in a climate-related external cost of electricity of \$0.017 per kWh when the social cost of carbon is \$28 per ton of CO_2e.

[13] Tol (2008) found that the social cost of carbon emissions varied over a wide range and had a standard deviation of \$96 per ton of CO_2e.

[14] The relationship between the policy cost per farm and the average size of the farm of digester adopters is shown in Appendix Figure A1.

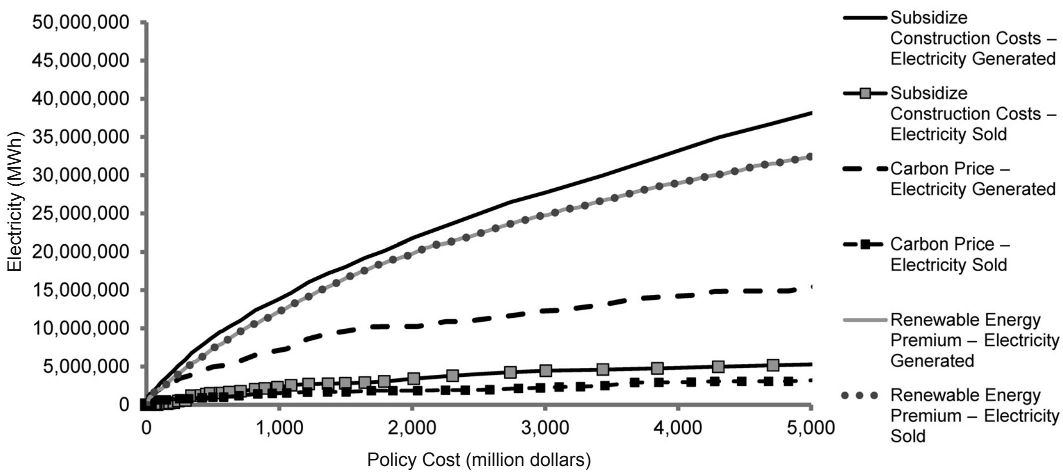

Figure 2. Electricity Generated and Sold by Policy Cost

Note: Policy cost is the discounted cost for a 15-year project. Electricity amounts are totals over 15 years.

level of subsidy increases and these smaller farms generate less electricity. The construction subsidy spurs the most electricity generation at every level of policy cost, followed closely by the renewable energy premium. Under the renewable energy premium, there is no digester adoption until the premium causes the selling (wholesale) price of electricity to exceed the retail (purchase) price.[15] Consequently, the renewable energy premium causes adopters to sell all of their generated electricity and results in the most electricity sold. Carbon pricing and construction subsidies result in substantially less electricity being sold at a given policy cost.

The construction cost-share policy is more efficient than the electricity subsidy in terms of electricity generation because the cost-share policy directly subsidizes the cost of generating electricity. In contrast, the electricity price subsidy directly raises the incentive to sell (rather than generate) electricity. As reflected in the adoption rates, a significant number of producers would find it profitable to provide their own electricity with a digester. Many of those same producers would not find it profitable to sell electricity until the selling price exceeded the retail price, which could not be achieved without a substantial price subsidy and, consequently, a large program outlay.

In terms of reducing carbon emissions, the carbon pricing/offset policy provides the greatest reduction at every policy cost and the renewable energy premium provides the least (Figure 3). The relative efficiency of the carbon pricing policy for this purpose makes sense because it directly increases the incentive to reduce GHG emissions. For all three policies, the marginal reduction in emissions decreases with additional policy expenditures as an increasing number of smaller farms adopt digesters.

Construction subsidies and renewable electricity premiums encourage some farmers to switch from aerobic to anaerobic manure management systems so they can produce methane for electricity generation (Figure 4). The carbon pricing program does not provide an incentive to switch manure management methods because of additionality rules—i.e., the rules prevent additional methane from being included in baseline emissions and therefore do not allow the burning of this methane to qualify for offsets. When an operation switches from an aerobic to an anaerobic system, its GHG emissions from manure management increase. However, much of the additional methane produced by the anaerobic system is combusted in the process of generating electricity, so the net increase in emissions is limited. Despite the

[15] Under the renewable energy premium policy, no dairy operator adopts digesters when the premium is below \$0.035 per kWh (shown in Appendix Figure A3). In the model, the difference between the retail and the wholesale price of electricity is assumed to be \$0.031 per kWh.

switching of some farms from aerobic to anaerobic manure systems, all three policies lead to net reductions in carbon emissions at every policy cost (Figure 3).

Figure 5 displays the level of public benefit (through use of renewable energy and reductions in carbon emissions that are valued at "social" prices) and private benefit (digester net revenue that accrues to the producer) of the three policies over a range of policy costs. As the cost of each policy increases, the amount of renewable electricity generated and carbon emissions reduced per dollar falls. The private benefit, on the other hand, increases approximately in proportion to the

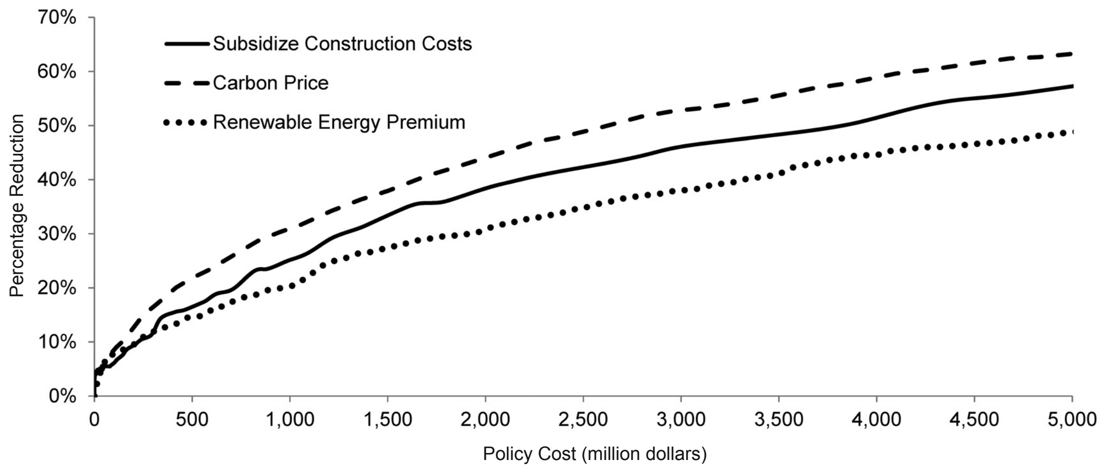

Figure 3. Percentage Reduction in Total Carbon Emissions from Manure Management by Policy Cost

Note: Policy cost is the discounted cost for a 15-year project.

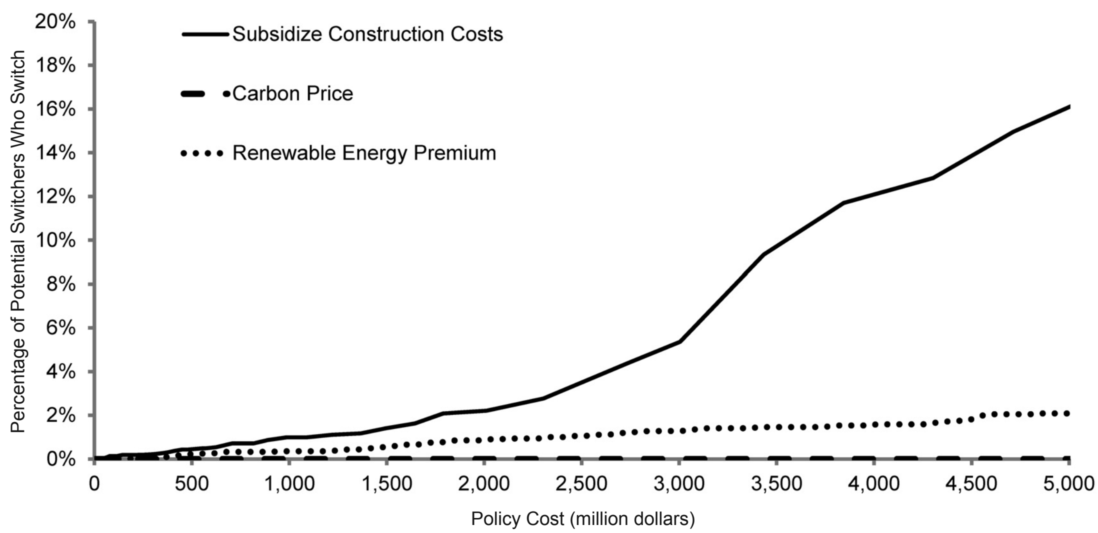

Figure 4. Percentage of Operators Using Aerobic Manure Management Systems Who Switch to Anaerobic Systems by Policy Cost

Note: Policy cost is the discounted cost for a 15-year project.

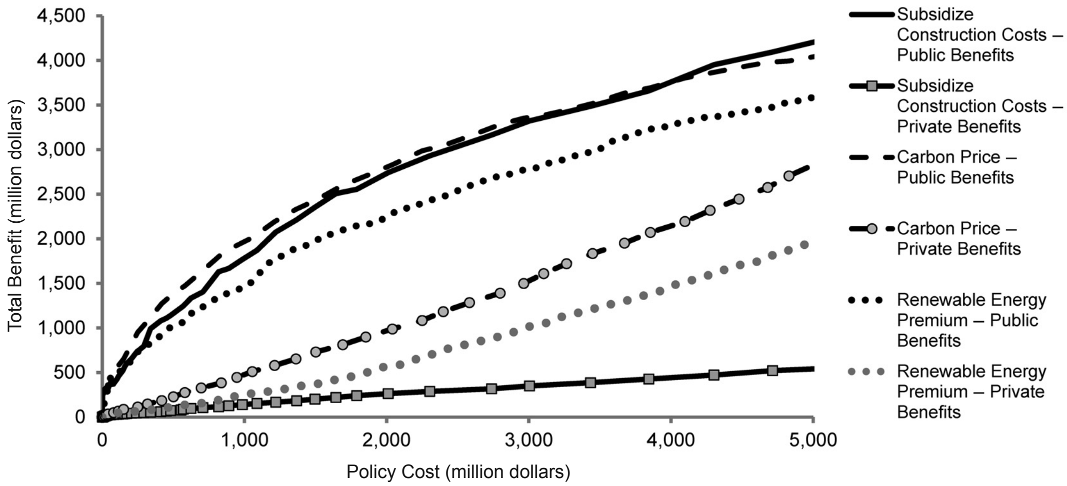

Figure 5. Public and Private Benefits by Policy Cost

Note: Policy cost is the discounted cost for a 15-year project.

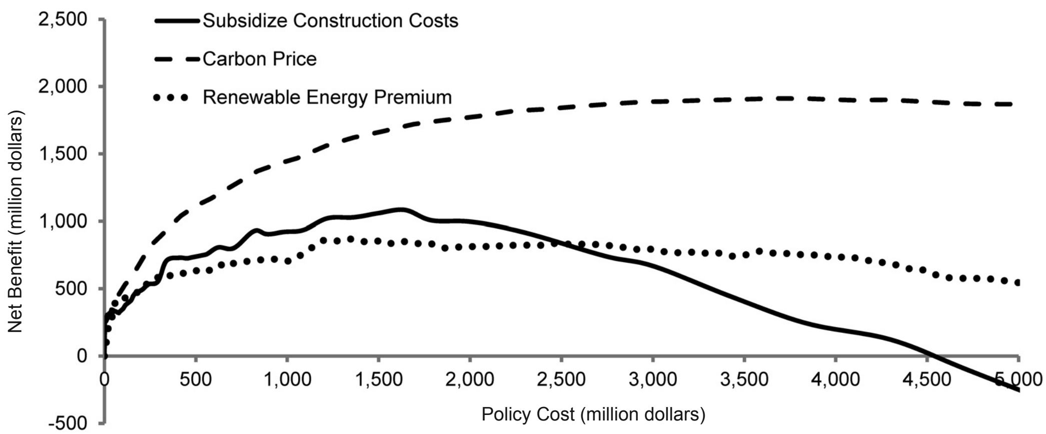

Figure 6. Net Social Benefits by Policy Cost

Note: Policy cost is the discounted cost for a 15-year project.

policy cost. Thus, the private benefit represents a growing share of the gross social benefit (public plus private) as the policy cost increases. The share increases most rapidly for the carbon pricing policy, which provides farmers with the greatest profit per policy dollar.

The decreasing marginal public benefit per policy dollar causes the net social benefit to become negative when the policy cost is sufficiently high (Figure 6). The construction cost-share program generates the maximum net social benefit ($1.1 billion) when expenditures on the subsidy are about $1.6 billion over 15 years, and the net social benefit of that policy becomes negative when expenditures on construction subsidies exceed $4.7 billion. For the renewable energy premium, the net social benefit peaks at $870 million when the policy expenditure is about $1.4 billion and becomes negative when the policy cost exceeds $7.8 billion (not shown in the figure). The net social benefit from the carbon pricing policy peaks at $1.9 billion when the policy expenditure

Table 2. Predicted Outcomes under Three Policies for a $500 Million 15-Year Policy Expenditure

	Policy		
	Subsidized Construction Cost	Renewable Electricity Selling Price Premium	Price on Carbon Emission Reduction
Policy instrument value	71.1%	$0.098 per kWh	$11 per ton CO_2e
Digester adoption rate (percent of operations)	0.9%	0.4%	0.5%
Average size of adopter operations (number of cows)	2,266	4,228	3,080
Average profit of adopters (dollars)	162,059	541,102	895,624
Electricity generated (million MWh)	9.0	7.5	5.0
Electricity sold (million MWh)	1.5	7.5	1.0
Carbon emission reduction (percent)	17%	14%	22%
Cost per unit of electricity produced (dollars per MWh)	5.37	6.54	9.71
Cost per carbon reduction (dollars per ton CO_2e)	14	17	11
Digester net revenue (million dollars)	75	110	228
Public benefit of renewable electricity (million dollars)	204	171	113
Public benefit of carbon reduction (million dollars)	976	846	1,273
Total benefit (million dollars)	1,255	1,127	1,614
Net benefit (million dollars)	755	616	1,114

Notes: All values are totals over the 15-year life cycle of the digester. Dollar values are discounted. The actual policy cost for the construction-cost subsidy and for carbon pricing is $500 million. However, because farms cannot adopt partial digesters, the actual policy cost for the renewable electricity premium is $512 million.

is $3.7 billion and becomes negative when the policy cost exceeds $21.4 billion.

Of the three policies, construction cost-sharing generates a negative social benefit in response to the smallest policy investment ($4.7 billion) in part because the subsidy induces a relatively large number of producers to adopt digesters. At a policy cost of $4.7 billion, the adoption rate is 15 percent under this policy compared to 4.9 percent for an electricity subsidy and 6.0 percent for a carbon pricing policy. As shown in Figure 5, in aggregate these smaller-scale producers generate comparable amounts of electricity and reductions in emissions but earn lower profits from the digesters, which reduces the net social benefit.[16]

Next, we compare the three policies across metrics while holding the policy cost for the life of the digester at $500 million (Table 2).[17] At that cost, 71.1 percent of construction costs would be borne by the government, the renewable electricity premium would be $0.098 per kWh, and the carbon price would be $11 per ton of CO_2e. At this price, all three policies would yield an adoption rate of at least 0.4 percent, a 14 percent reduction in carbon emissions, and 5.0 million MWh of electricity production. Hence, the policies would result in very low adoption rates. However, the few large-scale adopters could substantially reduce GHG emissions from manure management.

Given a $500 million policy cost, the carbon pricing policy induces the largest reduction in GHG emissions at 22 percent. The construction-cost subsidy ranks second in emission efficiency with a 17 percent reduction at a cost of $14 per ton of CO_2e. The cost-share policy results in the most electricity production: 9.0 million MWh at a cost of $5.37 per MWh.

[16] A cap on the construction-cost subsidy can extend the private benefit of a policy to a larger number of producers. The result is more adoption but not necessarily an increase in the net social benefit. This is illustrated in Appendix Figures A4 and A5, which show the social benefit of a policy that caps the construction-cost subsidy at $500,000.

[17] To put this policy expenditure into perspective, the REAP budget for 2011 was $70 million; over a 15-year horizon, that amounts to a discounted total of $726 million.

Table 3. Net Social Benefits under Alternative Public Benefit Assumptions: 15-Year Policy Cost of $500 Million[a]

	Public Value of Carbon Emission Reduction Net Social Benefit in Million Dollars		
Policy	\$0 per ton CO_2e[b]	\$28 per ton CO_2e[c]	\$56 per ton CO_2e[d]
Public Climate-unrelated Value of Renewable Energy = \$0.00/kWh			
Subsidize Construction Costs	–425	655	1,736
Electricity Selling Price Premium	–401	532	1,466
Price Carbon Emission Reductions	–272	1,059	2,390
Public Climate-unrelated Value of Renewable Energy = \$0.016/kWh			
Subsidize Construction Costs	–326	755	2,336
Electricity Selling Price Premium	–318	616	2,061
Price Carbon Emission Reductions	–217	1,114	2,946
Public Climate-unrelated Value of Renewable Energy = \$0.032/kWh			
Subsidize Construction Costs	–227	854	1,935
Electricity Selling Price Premium	–235	699	1,633
Price Carbon Emission Reductions	–162	1,169	2,500

[a] Dollar values are discounted. The actual policy costs for the construction cost subsidy and the carbon emission reduction pricing policies is \$500 million. However, due to the fact that farms cannot adopt partial digesters, the actual policy cost for the renewable electricity selling price premium is \$512 million.

[b] When carbon is \$0 per ton of CO_2e and the climate-unrelated value of renewable energy is \$0.00, \$0.016, or \$0.032 per kWh, the total (climate and non-climate) social value of renewable energy is \$0.00, \$0.016, or \$0.032 per kWh, respectively.

[c] When carbon is \$28 per ton of CO_2e and the climate-unrelated value of renewable energy is \$0.00, \$0.016, or \$0.032 per kWh, the total (climate and non-climate) social value of renewable energy is \$0.017, \$0.033, or \$0.049 per kWh, respectively.

[d] When carbon is \$56 per ton of CO_2e and the climate-unrelated value of renewable energy is \$0.00, \$0.016, or \$0.032, the total (climate and non-climate) social value of renewable energy is \$0.034, \$0.050, or \$0.066 per kWh, respectively.

Of the three policies, carbon pricing achieves the greatest net social benefit (\$1.1 billion), followed by construction-cost subsidies (\$0.8 billion) and electricity subsidies (\$0.6 billion). Carbon pricing also generates the greatest total profit for operators (\$228 million) and the highest average profit per adopter (\$895,624).

Carbon emission reductions comprise a much larger share of the total public benefit than does renewable electricity. At a policy cost of \$500 million, public (externality) benefits from carbon emissions reductions comprise 83–92 percent of all public benefits.[18] Of course, the ratio depends on the value placed on renewable electricity versus emission reductions.

Table 3 illustrates how the net social benefit of the policies varies when different assumptions are made about the value of renewable energy and GHG reductions to society. We set the non-climate-related value of renewable energy to three values: \$0.0 per kWh, \$0.016 per kWh as a baseline, and \$0.032 per kWh. We set values for a reduction in CO_2e emissions at \$0 per ton, \$28 per ton as a baseline, and \$56 per ton.[19] As before, this analysis assumes a \$500 million policy cost over 15 years.

[18] These percentages remain relatively constant when considering other policy costs (see Appendix Figure A6).

[19] The public benefits of renewable energy can be categorized as climate-related and non-climate-related. The climate-related component depends on the social cost of carbon and is adjusted accordingly for the simulations as shown in the notes in the table.

When no social value is given to carbon reductions, no project has a positive net present value at the selected renewable electricity values. However, if reductions of GHG emissions are valued at \$28 or \$56 per ton of CO_2e, all of the projects obtain a positive value.[20] Carbon pricing achieves the greatest net social benefit regardless of the renewable electricity value. For non-zero values of emission reduction, the construction-cost subsidy provides the second greatest level of net social benefit, yielding 62–79 percent of the benefit of carbon pricing. When GHG emissions reductions are valued, the electricity price subsidy provides the smallest social benefit of the three policies because it is the least efficient at reducing emissions (shown in Figure 3).

Conclusion

We developed a model of digester investment and applied it to a nationally representative survey of dairies to compare the benefits and costs of three policies designed to promote renewable electricity and reduce carbon emissions. We found that subsidizing the cost of constructing digesters is the most cost-efficient policy for simultaneously achieving a desired digester-adoption rate among producers and promoting renewable energy generation. The electricity subsidy is the most cost-efficient policy for achieving targeted levels of renewable electricity sales while carbon pricing is the most cost-effective mechanism for reducing emissions.

Under baseline assumptions, we found that the carbon price subsidy provides for the greatest net social benefit over a wide range of policy investment levels. The results demonstrate that the cost of promoting digesters can outweigh the benefits derived from them if policy expenditures are too high. For example, we found that the net social benefit for construction cost-sharing peaks when expenditures are about \$1.6 billion over 15 years but is negative when expenditures exceed about \$4.7 billion.

[20] Considering a policy cost of \$500 million, when the non-climate-related public benefit of renewable energy is set to zero, a public cost of carbon of between \$5 and \$13 (depending on the policy) is necessary for the net social benefit to be non-negative (see Appendix Figure A7). Likewise, if the public cost of carbon is set to zero, a public benefit of renewable energy unrelated to climate would need to be between \$0.068 and \$0.081 per kWh (depending on the policy) for the net social benefit to be non-negative (see Appendix Figure A8).

While a construction-cost subsidy leads to the highest rate of adoption regardless of the policy cost, the adopters have, on average, smaller operations than adopters in the other two programs. Overall, this greater number of smaller farms still reduces emissions by a comparable amount and provides about the same amount of renewable electricity as the other two policies. But the individual producers earn less on average, and the programwide total profit is less under the construction-cost program. Under carbon pricing and the renewable electricity premium, an individual operator's profit is greater but revenue flows to a smaller number of farmers and the farms are larger in scale. As policy expenditures increase, the average size of adopter operations decreases, meaning that a smaller social benefit (carbon emission reductions and renewable electricity generation) is generated from each additional policy dollar. The marginal social benefit decreases most rapidly for the construction cost-sharing program.

Our results indicate that relatively few operations would adopt a digester when any of the policies invested \$500 million over 15 years. Carbon emissions, however, would decline by 14–22 percent and 5–9 million MWh of renewable electricity would be generated over 15 years. The electricity subsidy and construction cost-share programs would induce some farmers to switch from aerobic to anaerobic manure systems that produce more methane. However, in aggregate, the policies would result in a substantial net decrease in carbon emissions.

Our results illustrate the robustness of the social benefit rankings to the social externality values assumed for renewable electricity and carbon emission reductions. We find that, over a wide range of values, carbon pricing produces the greatest net social benefit of the three policies.

As previously discussed, our analysis does not account for the risk and uncertainty associated with investment in a digester; that is, it does not account for the value that investors place on delaying an investment to resolve uncertainty nor for investors' preference for guaranteed returns over uncertain returns. The policies considered in this analysis would likely alter the degree of risk and uncertainty associated with investing in a digester. A construction cost-share policy that reduces the initial investment cost would decrease the option value and the risk associated with a digester project by reducing the potential for financial loss if prices or revenues declined in later periods. Uncertainty surrounding future renewable electricity subsidies or carbon prices, on the other hand, could reduce the effectiveness of these policy instruments for promoting digester adoption. Public or private mechanisms such as futures markets or long-term price contracts could be established to help reduce the risk associated with electricity and carbon prices. While accounting for option value and risk aversion was beyond the scope of this study, the central role of risk and uncertainty in digester adoption decisions makes it an important area for future research.

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