Social Cost of Biomass Energy from Switchgrass in Western Massachusetts

David Timmons

Producing biomass energy requires much land, and effects of biomass production on ecosystem services could greatly affect total biomass energy cost. This study estimates switchgrass production cost in western Massachusetts at three levels: private production cost, private cost plus social cost of nitrogen fertilizer externalities, and those costs plus the social opportunity cost of foregone forest ecosystem services. Values for nitrogen externalities and forest ecosystem services estimated with benefit transfer suggest that social cost is much greater than private switchgrass production cost. The benefit-transfer estimates are only first approximations, but conclusions are robust to a large range of values.

Key Words: ALMANAC, biomass energy, ecosystem services, fertilizer externalities, switchgrass

Biomass energy is one of several renewable energy alternatives that will ultimately be needed to replace nonrenewable fossil fuels and to reduce anthropogenic carbon emissions. Biomass energy is indirect solar energy: sunlight powers photosynthesis and plant growth with solar energy being stored as hydrocarbons in plant matter. Compared to other renewable energy sources, biomass energy can be relatively inexpensive (Pimentel et al. 2002), but by its nature, it also requires large amounts of land. Pimentel et al. (2002) estimated that producing electricity from forest biomass (in a thermal plant) required 71 times more land area than producing the same amount of electricity from solar photovoltaic panels. Total biomass energy availability may ultimately be constrained by land availability. In Massachusetts, for example, primary energy consumption in 2009 was 1,505 petajoules (Energy Information Administration 2011) while a recent estimate of energy that could be sustainably produced from forest biomass was 13 petajoules (based on Kelty, D'Amato and Barten 2008) or less than 1 percent of current energy use. Crops like switchgrass yield more biomass per hectare than forests, in Massachusetts perhaps three to four times more per unit area (calculated from Innovative Natural Resource Solutions (2007) and Duffy (2008)). Yet even with a switchgrass yield of 9.7 metric tons per hectare (this study) at 15.6 gigajoules

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per metric ton (McLaughlin et al. 1996), using switchgrass to supply all of Massachusetts' primary energy needs would require 99,369 square kilometers, which is 4.9 times the land area of the state.

Biomass energy is thus very much about land use. Producing biomass energy will greatly affect the total value of goods and ecosystem services provided by the landscape. While many studies have looked at the availability of biomass energy in the United States (e.g., Perlack et al. 2005) and some have estimated biomass supply price as a function of quantity produced (e.g., Haq 2002), this study contributes to a smaller literature that aims to assess the full social cost of biomass energy production. Social cost includes both biomass producers' costs and external costs for society. Williams et al. (2009) and Hill (2007) described and quantified many externalities related to biomass crop production, including eutrophication (e.g., algal blooms) from nitrogen and phosphorous releases, greenhouse gas emissions and other forms of air pollution, and soil erosion. Though neither study attempted to monetize these external costs, Hill (2007) suggested that biomass from native prairies might have higher total value than biomass from crop land given the ecosystem service values provided by prairies. In a global study, Antoine, Gurgel, and Reilly (2008) included monetized forest recreation values in a model of land use change for biomass crops. The authors found that including forest recreation values improved estimates of welfare costs associated with land use changes, but they did not attempt to reflect those welfare costs in the total cost of biomass energy. Kusiima and Powers (2010) monetized externalities of biomass energy produced from corn, corn stover, switchgrass, and forest residue and found that total external cost ranged from \$0.57 per liter for corn ethanol to \$0.07 per liter for cellulosic ethanol produced from forest residue. Their study demonstrated some of the challenges inherent in estimating the full social cost of the complex biomass-energy production process, and the authors reported distributions of results as well as point estimates.

This study estimates the cost of producing switchgrass in Massachusetts at three levels: (i) private cost of production for switchgrass farmers, (ii) private cost plus monetized social cost of nitrogen fertilizer externalities, and (iii) those costs plus the social opportunity cost of foregone forest ecosystem services since in Massachusetts land used to grow switchgrass biomass could instead grow forest biomass. The following sections present the study's methods and results for each of the three cost estimates.

Massachusetts' climate can support many biomass crops, including switchgrass (the subject of this study), other cellulosic grassy crops (e.g., reed canarygrass), cellulosic tree crops (e.g., willow and poplar), and traditional non-cellulosic crops such as corn and soybeans. This study does not quantify biomass energy benefit, but we assume that the social benefit of biomass energy may exceed its total cost, now or in the future. Estimates of full social cost can help to determine whether biomass energy generates net social welfare.

We base our estimates of nitrogen externalities and foregone ecosystem services on a benefit-transfer methodology that relies on values from other studies in the literature. While accuracy can be difficult to achieve in both ecosystem service valuation and benefit transfer, this study demonstrates that approximations of such values can be adequate for policy purposes. Greater precision would not likely change major conclusions.

Results of this study indicate that the social cost of switchgrass production is much greater than the private production cost and that switchgrass can generate significantly more biomass than forest per unit of area. The policy question then becomes whether the social marginal benefit of additional energy available from switchgrass exceeds its social marginal cost.

Private Switchgrass Supply Function

We base estimates of private switchgrass supply functions on calculations of producers' costs for each parcel of land in western Massachusetts that might grow switchgrass. Sorting these cost estimates from lowest to highest provides an estimate of a marginal cost function assuming that markets are competitive.

Switchgrass is native to North America and can be grown on marginal land with fewer inputs than crops like corn. Yet switchgrass also responds positively to nitrogen fertilizer applications, and previous studies have shown that such applications are economically optimal for producers (Brummer et al. 2001, Nelson, Ascough and Langemeier 2006, Lemus et al. 2008). Since many production costs are constant regardless of yield (e.g., land, planting), larger yields reduce the cost per unit. This study estimates switchgrass yields for three levels of nitrogen fertilizer application: 0, 67, and 135 kilograms per hectare of nitrogen (0, 60, and 120 pounds per acre). The University of Massachusetts is field-testing these fertilizer levels, providing data to calibrate a crop-growth simulation model.

We construct a technically feasible switchgrass supply function for five western counties of Massachusetts using estimated yields and production costs. Following are major steps in estimating this supply function (additional details are available in Timmons (2012)):

- Define the area's potential land resource using Natural Resources Conservation Service (NRCS) soil maps, which indicate soils suitable for agricultural use, and GIS land use/cover maps that identify undeveloped areas.
- Estimate switchgrass yields on potential production land using the Agricultural Land Management with Numerical Assessment Criteria (ALMANAC) model (Kiniry et al. 1992), which has previous success in estimating switchgrass yields (Kiniry et al. 2005, Kiniry et al. 2008a, Kiniry et al. 2008b). ALMANAC comes from the Erosion Productivity Impact Calculator (EPIC) family of simulation models developed by the U.S. Department of Agriculture's Grassland Soil and Water Research Laboratory in Temple, Texas (Williams et al. 2008). Thirty-year ALMANAC simulations provide estimates of yields, which vary annually due to weather. The ALMANAC weather model is based on data from National Oceanic and Atmospheric Administration (NOAA) stations. The model estimates annual yields for each nitrogen treatment and soil type and for each 100-meter elevation band in which a soil type occurs. The ALMANAC model also provides annual estimates of nitrogen loss to surface waters and groundwater.
- Develop a switchgrass enterprise budget based on estimates from Duffy (2008) for Iowa with appropriate adjustments for Massachusetts. For example, we use published machine rates from Pennsylvania (USDA/Pennsylvania Department of Agriculture 2009) in place of the Iowa rates since Pennsylvania's conditions more closely resemble those of Massachusetts. A survey of landowners in western Massachusetts

(Timmons 2011) provides estimates for land rental rates, or returns to landowners.

- Adjust other spatially differentiated costs. For example, the GIS model assigns a greater cost per hectare to smaller fields due to increased travel and set-up time and less efficient machine operation.
- Use the results to generate supply functions for the region. Combining the yield and budget information results in a unit production cost estimate for each map grid cell. Sorting grid-cell costs from lowest to highest describes a marginal production-cost function: a technically feasible supply function that shows how much switchgrass could be produced at various prices if all land resources were available for this purpose.

Table 1 shows yield estimate statistics for the three nitrogen treatments studied. With no nitrogen (i.e., natural conditions), soils in the study area yield a mean of 3.82 metric tons (Mg) per hectare (ha) with a standard deviation of 0.81 Mg/ha and a range of 1.66 to 6.59 Mg/ha. Nitrogen (N) fertilizer both increases yields and reduces the coefficient of variation. Overall, yields range from a low of 1.66 Mg/ha (minimum with zero nitrogen) to a high of 11.17 Mg/ha (maximum with 135 kg/ha N). Yields vary by soil type, elevation, and fertilizer treatment. It is hardly meaningful to discuss a "typical" Massachusetts switchgrass yield without including more particulars. Added nitrogen plays a major role in switchgrass productivity.

The information in Figure 1 is based on estimated yields and shows hectares of land required to produce different switchgrass quantities in western Massachusetts. Note that nitrogen fertilizer greatly decreases the amount of land needed to grow a given quantity of switchgrass. Figure 2 shows private marginal cost functions for switchgrass (black lines) for each of the nitrogen fertilizer treatments modeled (social marginal costs shown in Figure 2 are discussed later). The supply functions show marginal production cost increasing in quantity supplied, mainly because of declining yields as poorer soils are used, though spatial attributes (e.g., field size and location) also affect the cost of production.

Since fixed production cost per hectare is relatively constant, yield variation due to fertilizer application is a primary component of final cost per ton variation. Nitrogen additions greatly increase switchgrass yields, and greater yields spread the fixed costs of producing switchgrass (especially land cost) over more tons of product. Nitrogen fertilizer thus greatly reduces private production cost per ton, as shown by the progressively lower private marginal cost functions in Figure 2. The higher yields from nitrogen fertilizer

Table 1. ALMANAC Yield Estimates for Western Massachusetts

Nitrogen Fertilizer	Mean Yield	Standard Deviation	Coefficient of Variation	Minimum Yield	Maximum Yield
Kilograms per Hectare	Metric Tons per per Hectare	Metric Tons per per Hectare		Metric Tons per per Hectare	Metric Tons per per Hectare
0	3.82	0.81	0.21	1.66	6.59
67	6.23	0.65	0.10	3.51	9.47
135	9.66	0.90	0.09	4.46	11.17

reduce private marginal costs at all production quantities, at least up to the 135 kg/ha N application modeled. Though a producer can grow switchgrass without nitrogen fertilizer, there is a financial incentive to use a significant amount of it. But a producer's optimum is not necessarily socially optimal given that the nitrogen pollution cost is external to a producer.

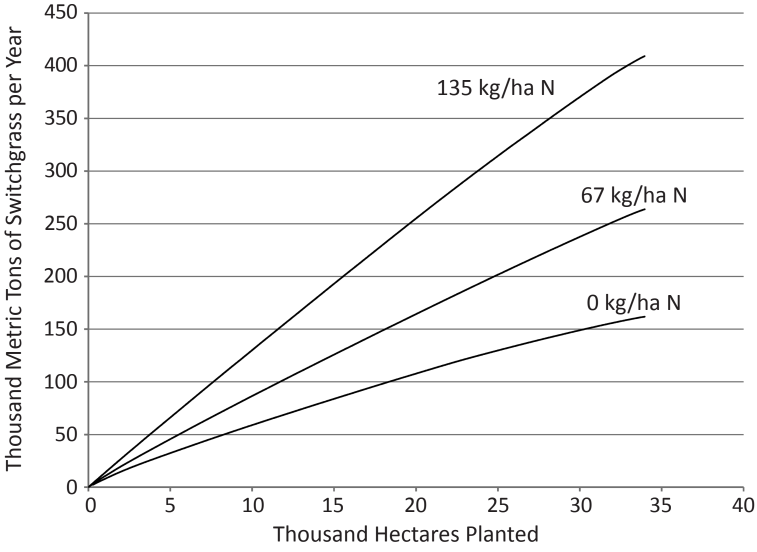

Figure 1. Land Required to Produce Switchgrass Quantities at Different Fertilizer Levels

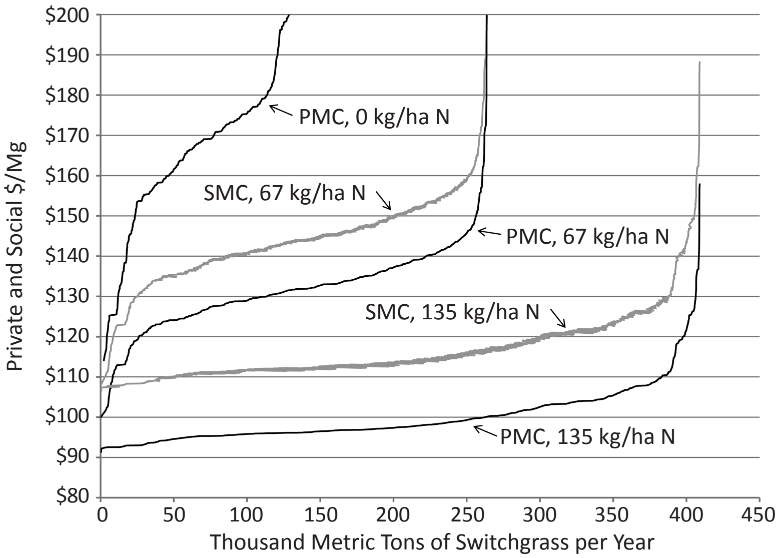

Figure 2. Switchgrass Supply Functions by Nitrogen Fertilizer Application Level

PMC = private marginal cost. SMC = social marginal cost, including nitrogen fertilizer externalities.

Social Cost of Nitrogen Externalities

Agricultural production can create a number of negative externalities. This section describes estimates of nitrogen pollution costs expected for switchgrass production in western Massachusetts based on a benefit-transfer approach. We multiply estimated externalities in dollars per unit of nitrogen emitted by estimated emissions for western Massachusetts to determine total externality cost.

In this study we consider only nitrogen fertilizer externalities. While producers may also apply phosphorus fertilizer, phosphorous is generally less critical for switchgrass than nitrogen (Haque et al. 2012), and the University of Massachusetts' experimental switchgrass plots are grown without added phosphorus. ALMANAC does not model potassium use or loss. Other than fertilizer use, perennial switchgrass production has few externalities compared to most food crops. Soil erosion rates are low, pesticides are not normally used, and herbicides are typically used only at planting. Other common agricultural externalities—such as those related to irrigation and management of animal waste—do not apply to switchgrass.

Using a benefit-transfer method (Boyle and Bergstrom 1992), externality values estimated at different study sites are transferred to the western Massachusetts policy site. There are several ways to conduct benefit transfer. The most basic is a point transfer where a researcher directly uses a study site value at a policy site—the method used here. A number of papers have described potential errors in benefit transfer (e.g., Brouwer 2000). Boyle and Bergstrom (1992) suggested three "idealistic" criteria for benefit transfer validity: (i) an identical good valued at study and policy sites, (ii) identical population characteristics at both sites, and (iii) the same assignment of property rights at both sites (to determine whether willingness-to-pay or willingness-to-accept is the appropriate value measure). In general, then, more similarity between study and policy sites will promote more accurate benefit transfer. Loomis and Rosenberger (2006, p. 344) observed that "even a simple benefit transfer . . . may provide an indication of whether the benefits and costs are in the same order of magnitude." This study in fact demonstrates that some knowledge of externality and ecosystem service costs is critical for evaluating the social cost of biomass energy and that rough estimates provided by point transfers are likely adequate for this task.

Nitrogen pollution has many known negative consequences. The nitrogen cycle is complex; after emission, nitrogen from fertilizer may change chemical form several times before it is neutralized (Galloway et al. 2003), and each nitrogen compound has a different impact on the environment. Table 2 presents a typology of reactive nitrogen impacts from Compton et al. (2011) along with benefit-transfer values for the specific effects used in this study (present values of losses for one nitrogen application). Although there have been many studies of nitrogen fertilizer externalities, few employed models that calculated the final externality cost of each nitrogen unit added (expressed in \$/kg N). The research by Compton et al. (2011) provides a good summary of such studies, but more research is warranted in this area.

Atmospheric nitrogen emissions from agricultural fertilizer include nitrous oxide (N_2O), ammonia gas (NH_3), and nitrogen oxide and dioxide (NO and NO_2), all of which are air pollutants. NO_2 is not modeled in this study. Nitrogen oxide has direct adverse human health effects. Both nitrogen oxide and ammonia

Table 2. Nitrogen Externality Typology and Benefit-Transfer Estimates

Effect	$/kg Nitrogen	Study Site	Reference
Ground-level Air Pollution			
Reduced visibility	$0.31	Chesapeake Bay	Birch (2011)
Nitrogen oxide human health cost	$14.93	Chesapeake Bay	Birch (2011)
Ammonia gas human health cost	$15.18	Chesapeake Bay	Birch (2011)
Ozone effect on crops	$1.51	Chesapeake Bay	Birch (2011)
Ozone effect on forests	$0.89	Chesapeake Bay	Birch (2011)
Nitrous Oxide Ozone Depletion			
UV damage	$1.33	United States	Compton (2011)
N_2O Greenhouse Gas Effect			
Climate change	$13.98	World	Tol (2008)
Terrestrial Acidification			
Building damage	$0.09	Chesapeake Bay	Birch (2011)
Forest damage	NE		
Freshwater Acidification			
Recreational fishing cost	NE		
Lake and stream aesthetics	NE		
Freshwater Eutrophication			
Waterfront property values	NE		
Fresh water recreational use	NE		
Endangered species	NE		
Harmful algae blooms	NE		
Nitrate Groundwater Pollution			
Well water nitrate treatment	$0.16	United States	Compton (2011)
Nitrate health cost	NE		
Coastal Eutrophication			
Recreational estuary use	$6.38	Chesapeake Bay	Birch (2011)
Commercial fishing impact	NE		
Total damages for which estimates are available	$54.76		

NE = no estimate.
Source: Compton et al. (2011).

gas are associated with increases in atmospheric particulate matter and ozone pollution (Galloway et al. 2003), which degrade air quality and have additional adverse health effects (Curtis et al. 2006). Nitrogen oxide also contributes to acid rain, damaging both natural and built environments. Table 2 shows the benefit transfers used for these damage values, all from a Birch et al. (2011) model of the Chesapeake Bay region. While this area is not "identical" (Boyle and Bergstrom 1992) to the western Massachusetts study area, as a U.S. East Coast location, it is similar. Sensitivity analysis for the sum of benefit-transfer values indicates that any value-transfer errors would not likely change major conclusions about nitrogen fertilizer use.

Nitrous oxide depletes stratospheric ozone (Galloway et al. 2003) and is a potent greenhouse gas with 298 times more global warming potential than carbon dioxide (Solomon et al. 2007). It persists in the atmosphere and affects climate for about 100 years (Galloway et al. 2003). Many studies have estimated costs of climate change with widely varying results. This study uses a value of $110/Mg carbon (C), the median value (in 2011 dollars) from a Tol (2008) metastudy. The Tol (2008) study included 211 peer- and nonpeer-reviewed studies of climate change cost. The $110 figure is considerably higher than another figure reported in the same study, $34/Mg C (2011 dollars) based on only peer-reviewed estimates using a 3 percent discount rate and no equity weight. The $110 figure is also much lower than the $314/Mg C figure calculated by the Stern Review (2006). Again, sensitivity analysis indicates that using neither the lower Tol (2008) nor higher Stern (2006) estimate would change conclusions. We adjust the $110/Mg C figure for the global warming potential of nitrous oxide and the nitrogen weight in nitrous oxide to arrive at the climate change cost of $13.98/kg N (Table 2).

In addition to atmospheric nitrogen emissions, nitrogen is lost as soluble nitrate (NO_3) in water flows (Galloway et al. 2003). Ingestion of nitrate in groundwater has been associated with several health conditions: methemoglobinemia ("blue baby" syndrome) in infants and elevated risks of colon and thyroid cancer (Powlson et al. 2008, Kilfoy et al. 2011). While the strength of these effects has been debated, there appears to be a consensus that nitrate has some ill health effect in humans. For example, a Dutch study estimated that nitrate in drinking water was associated with a 3 percent increase in incidences of colon cancer (Grinsven, Rabl and de Kok 2010) and Compton et al. (2011) reported a cost range of $0.14 to $3.38/kg N from that paper. In this study, we exclude that health value since it is not clear that the Dutch study site and our western Massachusetts policy site are sufficiently similar for benefit transfer given that both nitrogen-use patterns and population distributions on the landscape likely differ. Since more appropriate studies of the health impacts of nitrate pollution are not apparent, a limitation of this research (and of many studies of environmental externalities) is that estimates are incomplete. Totals exclude real values that have not yet been rigorously estimated in a place similar to the policy site, biasing estimates downward.

Nitrate emissions also impact downstream water quality and are especially a concern as a cause of eutrophication in coastal areas (Conley et al. 2009). This study includes a recreational-use value of $6.38/kg N from Birch et al. (2011), which updated and revised estimates based on Bockstael, McConnell, and Strand (1989) and Morgan and Owens (2001), both of which studied recreational swimming, boating, and fishing in Chesapeake Bay. Another missing value is the impact of nitrates on commercial fishing, though nitrate emissions from western Massachusetts and associated eutrophication clearly impact commercial fishing in Long Island Sound, which receives drainage from the study area (Driscoll et al. 2003). Diaz and Rosenberg (2011) described eutrophication effects on fisheries and asserted that costs associated with those effects have not been quantified, although Compton et al. (2011) presented an estimate of $56/kg N from an unpublished source for the Gulf of Mexico fishery. This is the largest externality value identified in the Compton study and including it in our model would raise the externality estimate for the 167 kg/ha N treatment by a factor of about 3.9. We exclude it because the fishery in the Gulf of Mexico is not necessarily similar to the one in Long Island

Sound and the quality of the study for the Compton et al. (2011) estimate is unknown.

Given the number and complexity of nitrogen fertilizer effects on the environment, monetizing ecosystem damage from nitrogen use is inherently challenging. Damage varies with nitrogen application level and timing, crop nitrogen uptake, soil characteristics, site hydrology, and proximity of human populations to pollution. As previously shown, supplying missing estimates could change results significantly. For example, Dodds et al. (2009) estimated nitrogen-related eutrophication costs for U.S. freshwaters totaling $2.2 billion annually, including losses to recreation and property values. Yet when Compton et al. (2011) divided these losses by total nitrogen flux to U.S. freshwater ecosystems, the resulting cost estimate was less than $0.01/kg N. This low estimate might be taken as more indicative of the difficulty of estimating true costs than of actual freshwater eutrophication damages. Also, the cost of the marginal nitrogen input is almost certainly higher than the cost of the average nitrogen input. More research in valuing nitrogen externality costs is needed.

The ALMANAC model estimates the two largest nitrogen fluxes (by weight) for the western Massachusetts project area: (i) nitrate lost in surface and subsurface water flows and (ii) nitrate that percolates to groundwater. Since the ALMANAC model does not estimate atmospheric nitrogen losses, this study uses fixed percentages from the literature for three sources of atmospheric nitrogen fertilizer loss: nitrogen oxide (Veldkamp and Keller 1997), ammonia gas (Mikkelsen 2009), and nitrous oxide (Eggleston et al. 2006).

As shown in the second column of Table 3, the largest proportion of nitrogen under an application of 67 kg/ha is lost to surface and subsurface waters that eventually drain to rivers and oceans. Under the 135 kg/ha treatment, the largest proportion of nitrogen is lost through nitrate percolation to groundwater. The nitrogen losses shown in Table 3 comprise 14.3 percent of the nitrogen applied under the 67 kg/ha treatment (9.57 kg/ha) and 22.1 percent of the nitrogen applied under the 135 kg/ha treatment (29.80 kg/ha). Since ALMANAC does not model atmospheric nitrogen emissions, the percentage losses remain constant under different fertilizer application scenarios. With larger applications of nitrogen, a higher proportion of soluble nitrogen is lost, so losses increase sharply as the application level rises. Nitrogen loss also varies widely by soil type since some kinds of soil are more prone to nitrogen loss.

The ALMANAC nitrate-loss estimates in our model are similar to values found in the literature. For example, in the 135kg/ha N treatment, ALMANAC estimates that 17 percent of applied nitrogen is lost as nitrate. A study by Babcock et al. (2007) that used the Soil and Water Assessment Tool (SWAT) estimated that 13 percent of the nitrogen applied was lost as nitrate in an Iowa watershed that was converted entirely to switchgrass production. Van Breemen et al. (2002) used a formula for nitrate loss from grassland that resulted in an estimate of 14 percent nitrogen loss as nitrate. Other studies have found much higher levels of nitrate emission for row crops like corn and soybeans (Powers 2007).

Table 3 also shows externality costs from Table 2 multiplied by estimates of nitrogen releases for western Massachusetts. We estimate the cost of nitrogen releases to be $75.35/ha per year under the 67 kg/ha treatment and $162.79/ha per year for the 135 kg/ha treatment (the zero nitrogen treatment is assumed to have a zero nitrogen externality cost). While it would be more appropriate to use increasing marginal costs for additional emissions, full marginal cost

Table 3. Nitrogen Loss and Cost Estimates

Nitrogen Fertilizer Application: 67 Kilograms per Hectare						
Nitrogen Loss Pathway	**Loss Rate: Percent of Application**	**Kilogram per Hectare Release**	**Percent of Total Loss**	**Cost per Kilogram Released[e]**	**Cost per Hectare**	**Percent of Total Cost**
Ammonia gas (NH_3) – atmospheric	2.4[a]	1.61	17	\$15.18	\$24.41	32
Nitrogen oxide (NO) – atmospheric	0.5[b]	0.34	3	\$17.73	\$5.94	8
Nitrous oxide (N_20) – atmospheric	2.3[c]	1.51	16	\$15.31	\$23.09	31
Nitrate (NO_3) – surface and subsurface water	5.0[d]	3.37	35	\$6.38	\$21.47	28
Nitrate (NO_3) – groundwater	4.1[d]	2.75	29	\$0.16	\$0.44	1
Total nitrogen loss	14.3	9.57	100		\$75.35	100

Nitrogen Fertilizer Application: 135 Kilograms per Hectare						
Nitrogen Loss Pathway	**Loss Rate: Percent of Application**	**Kilogram per Hectare Release**	**Percent of Total Loss**	**Cost per Kilogram Released[e]**	**Cost per Hectare**	**Percent of Total Cost**
Ammonia gas (NH_3) – atmospheric	2.4[a]	3.24	11	\$15.18	\$49.18	30
Nitrogen oxide (NO) – atmospheric	0.5[b]	0.68	2	\$17.73	\$11.97	7
Nitrous oxide (N_20) – atmospheric	2.3[c]	3.04	10	\$15.31	\$46.52	29
Nitrate (NO_3) – surface and subsurface water	6.1[d]	8.27	28	\$6.38	\$52.79	32
Nitrate (NO_3) – groundwater	10.8[d]	14.58	49	\$0.16	\$2.33	1
Total nitrogen loss	22.1	29.80	100		\$162.79	100

[a] Mikkelsen (2009).
[b] Veldkamp and Keller (1997).
[c] Eggleston et al. (2006).
[d] ALMANAC model for western Massachusetts.
[e] Table 2, this study.

functions are not available so marginal externality costs are assumed to be constant.

Figure 2 shows the social marginal cost functions (gray lines) generated from private production costs (as previously described) plus the nitrogen externality costs from Table 3. Since nitrogen additions are so effective at increasing switchgrass yields and reducing the private production cost per ton, adding the social nitrogen externality cost does not change the optimality of significant fertilizer use: full social cost is lowest at the highest nitrogen fertilizer level modeled. While nitrogen externalities per hectare rise with nitrogen application, yields also rise. In Figure 2, for example, at a quantity of 100,000 metric tons per year, the externality cost is greater for the 135 kg/ha N application than for the 67 kg/ha application, but the social marginal cost of switchgrass is still much lower with more nitrogen use. In this case, nitrogen use reduces private cost more than it increases externality cost. Thus fully internalizing the externality cost estimated here (e.g., with a Pigouvian nitrogen tax) would not change the optimality of high nitrogen use for farmers. Given a choice of applying 0, 67, or 135 kg/ha N, a profit-maximizing farmer would choose 135 kg/ha even if she had to pay a tax equal to the externality cost.

Biomass crop production is likely to generate significant releases of nitrogen, a conclusion that is robust to a large range of externality values. At 100,000 metric tons per year, the externality cost for the high-nitrogen application would have to increase 813 percent before it would have the same social cost per ton as the 67 kg/ha treatment. But with the larger fertilizer treatment, a greater proportion of nitrogen is lost to groundwater, so including a groundwater nitrate health cost estimate would disproportionately increase the externality estimate for the high-fertilizer-use scenario. Implementation of best management practices (BMPs) might also reduce nitrogen losses at little cost to producers, though these are not modeled in this study.

Hypothetical marginal benefit functions can demonstrate possible uses of the social marginal cost functions. For example, assume that the combined private and social marginal benefit is $120 per metric ton and constant (depicted in Figure 2 as a horizontal line at $120 per metric ton). Then the privately optimal switchgrass quantity (where marginal benefit equals private marginal cost) is 397,000 metric tons per year. When the social cost of nitrogen externalities is added, the quantity where marginal benefit equals social marginal cost drops to 306,000, a reduction of 23 percent from the private optimum (not considering foregone forest ecosystem services, as discussed later). But because of the shapes of the supply curves, this result is highly sensitive to the marginal benefit assumed for switchgrass. As Figure 2 shows, at a constant marginal benefit of $150/Mg, there is virtually no difference between private and social optima (since supply is highly inelastic in this price range). At a marginal benefit of $100 per metric ton, the privately optimal quantity is 264,000 metric tons while the socially optimal quantity is zero (since supply is elastic in this case and, with the nitrogen externality costs included, switchgrass cannot be produced for less than $100 per metric ton). Thus while welfare estimation based on switchgrass supply curves is possible, such estimates are highly sensitive to estimates of marginal benefit used. Also note that, for a given marginal benefit, a change in the nitrogen externality cost can affect welfare estimates much more than farmers' decisions about nitrogen use.

Social Cost of Foregone Forest Ecosystem Services

In Massachusetts, as in much of the eastern United States, land used to grow switchgrass could instead grow forests for biomass. Land use in Massachusetts has changed continuously since colonial times; land that was deep forest at settlement was slowly cleared for agriculture with many different crops raised over the following centuries (Russell and Lapping 1982). As better farm land became available in the West, many eastern farms were abandoned, and a large portion of Massachusetts farm land reverted to forest (Foster, Motzkin, and Slater 1998). Much of the land in this region can support farming or forests, both of which can produce biomass for energy. Thus if forest land provides greater ecosystem service value per hectare than switchgrass, foregone forest ecosystem services must be considered as an opportunity cost of growing switchgrass. We outline how such an opportunity cost can be estimated and provide evidence that suggests the magnitude of that cost. As with estimates of nitrogen externalities, estimates of foregone forest ecosystem services involve wide confidence intervals, but even minimal knowledge of these ecosystem service values can inform policy decisions.

There is ample evidence that forests generate greater ecosystem service value than switchgrass fields. To make this determination, one must estimate and compare ecosystem service value for each type of land cover. Table 4 presents a typology of twelve ecosystem services used by Liu et al. (2010), a truncation of the full 22-service typology by de Groot, Wilson, and Boumans (2002). Some categories from the de Groot typology were combined (e.g., gas regulation and climate regulation), and categories that described the provision of market goods were omitted (in this study, the market good is biomass for energy).

Liu et al. (2010) estimated the total nonmarket value of ecosystem services for New Jersey. There has been widespread criticism of such studies in the economic literature (Pearce 1998, Bockstael et al. 2000), particularly about the practice of aggregating total value for a large area like New Jersey (Liu et al. 2010) or the world (Costanza et al. 1997). But such ecosystem service valuation is perhaps better suited to assessing the marginal changes in land cover considered here.

Table 4 shows values for forest cover and for the mean value of crop and grass cover, representing possible switchgrass ecosystem service value. Liu et al. (2010) used a benefit-transfer method to estimate values, calculating a simple mean of point values from all identifiable studies for each service type. Table 4 shows the mean value for each type of ecosystem service along with the number of studies used in the estimate (since some estimates reflect numerous studies, we do not cite the studies individually; a complete list is available in Costanza et al. (2007)). Values shown on the left side of Table 4 are based on peer-reviewed studies only. In the peer-reviewed columns, missing estimates greatly affect totals, especially for grass and crop land. Thus, as an additional indicator, estimates shown on the right side of Table 4 include results from studies that did not undergo peer review.

In ecosystem service categories with estimates for both types of land cover, Table 4 shows that forest generates greater value in all cases but one. Forest thus has greater total ecosystem service value based on both groups of studies. Yet the question of the applicability of these results to Massachusetts remains (i.e., about the quality of the benefit transfer). The New Jersey project included only studies of ecosystem services found in New Jersey and only studies that were conducted in temperate regions of North America and Europe (Liu et al. 2010). These criteria apply equally to Massachusetts, though none of the studies cited for forest or grass and crop land was actually conducted in Massachusetts (or in New Jersey). More importantly, no studies cited considered the same ecosystem service for both land covers; the land cover comparisons were always between studies done at different times and places and frequently involved different methods. A rigorous approach to assessing the value of switchgrass versus forest as land cover would use identical methods to measure their respective ecosystem service values. The extent of forest cover and other types of cover would also be important in determining marginal values. In the absence of such data, results shown in Table 4 indicate only possible differences in ecosystem service value.

There are several reasons to believe that forest offers greater ecosystem service value than switchgrass plantings. Forests provide a range of ecosystem services that agricultural biomass production either cannot provide or provides to a lesser extent:

- Water retention: Grasslands have more runoff and less evapotranspiration than forests, which can impact downstream water resources and flood control (Zhang et al. 2007, Turner and Daily 2008). For example, a study of the impacts of land cover on hydroelectric production on the Yangtze River in China (Guo, Xiao, and Li 2000) found that grassland had only 35 percent of the water conservation capacity of mixed forests. Forests' water retention services also provide more consistent groundwater recharge and municipal water system supply.
- Habitat: Second-growth native forests as found in Massachusetts have more plant diversity than a monoculture biomass crop like switchgrass. Such plant diversity creates a basis for wildlife diversity (Fromm 2000), a resource for which the public consistently shows positive (if variable) willingness-to-pay (Nunes and van den Bergh 2001).

Table 4. Forest and Crop and Grass Land Ecosystem Service Typology and Benefit-Ttransfer Estimates

	Peer-reviewed Studies		Peer- and Nonpeer-reviewed Studies	
Ecosystem service	Forest Land $/ha[a] (Studies)	Grass and Crop Land $/ha[a] (Studies)	Forest Land $/ha[a] (Studies)	Grass and Crop Land $/ha[a] (Studies)
Gas and climate regulation	$176 (31)	$9 (1)	$159 (39)	$9 (3)
Disturbance regulation	NE	NE	NE	NE
Water regulation	NE	NE	NE	$4 (1)
Water supply	$26 (1)	NE	$479 (2)	NE
Soil formation	NE	$10 (1)	$15 (1)	$7 (2)
Nutrient cycling	NE	NE	NE	NE
Waste treatment	NE	NE	$129 (1)	$66 (1)
Pollination	$476 (1)	$24 (2)	$476 (1)	$22 (4)
Biological control	NE	NE	$6 (1)	$24 (2)
Habitat and refugia	$2,713 (8)	NE	$2,713 (8)	$2,443 (2)
Aesthetic and recreation	$ 382 (14)	$18 (4)	$359 (15)	$19 (5)
Cultural and spiritual	NE	NE	$3 (1)	NE
Total value estimated	$3,774 (55)	$60 (8)	$4,338 (69)	$2,594 (20)

[a] 2011 dollars per hectare.

NE = no estimate. The number of studies for each value is listed to the right in parentheses.

Source: Liu et al. (2010) and Costanza et al. (2007), which included estimates and references for all original studies.

- Carbon storage: Standing forests and forest soils represent significant carbon sinks. While switchgrass may accumulate carbon in soil at a rate that is similar to forests (Lemus and Lal 2005), switchgrass lacks forests' above-ground carbon storage capacity.

Available evidence suggests that forest biomass is less expensive to produce than switchgrass biomass. For example, the New England market price for forest wood chips is about $55/Mg dry ($30/ton at 40% moisture) (New Hampshire Timberland Owners Association 2011). This also represents the marginal production cost if the forest wood chip market is competitive and in equilibrium. At a quantity of 100,000 metric tons per year, the marginal switchgrass production cost is estimated at a much higher $95.82/Mg dry. While some of this cost difference is due to differences in landowner rent from forest land versus crop land, New England forest biomass is still less costly to produce than switchgrass after accounting for landowner returns.

If switchgrass offers no production cost advantage over forest biomass, any foregone forest ecosystem service value is an additional cost of switchgrass production along with the nitrogen fertilizer externalities previously described. To harness the greater yield potential of switchgrass, one must pay a greater biomass production cost, accept damage from nitrogen fertilizer externalities, and forego the ecosystem services that forests provide. While these ecosystem service costs are difficult to estimate, the results reported in Table 4 (right side) suggest a value difference of $1,744 per hectare or $181 per metric ton of switchgrass at the mean yield of 9.66 Mg/ha. As shown in Table 5, this makes the total social cost of switchgrass much greater than the private production cost, perhaps three times greater (if all externality and ecosystem service costs were correctly estimated). The majority of the social cost is due to foregone forest ecosystem services.

This general conclusion about the social cost of switchgrass is robust to a range of ecosystem service values. For example, if the ecosystem service opportunity cost of switchgrass were only half of the value shown in Table 5,

Table 5. Switchgrass Total Cost with Nitrogen Externalities and Loss of Forest Ecosystem Services, 135 Kilograms per Hectare of Nitrogen

Description	Cost per Hectare	Cost per Metric Ton at 9.66 Mg/ha Yield	Percent of Total Switchgrass Social Cost
Switchgrass private production cost, q = 100,000 metric tons per year	$925.04	$95.76	33
Switchgrass nitrogen externality cost[a]	$162.79	$16.85	6
Switchgrass ecosystem service opportunity cost of not growing forest[b]	$1,744.47	$180.59	61
Total social cost of switchgrass	$2,832.30	$293.20	100

[a] Table 3 of this study, 135 kg/ha nitrogen application.

[b] Table 4 of this study, peer- and nonpeer-reviewed studies, difference between forest land and grass and crop land totals.

Note: All figures in 2011 dollars.

the social cost of switchgrass would still be more than twice the private cost. Foregone forest ecosystem services make up 45 percentof the social cost. The social cost of switchgrass cannot be less than the social cost of forest biomass unless the ecosystem service value of switchgrass is greater than the ecosystem service value of forest plus the switchgrass nitrogen externality cost.

Whether switchgrass biomass at a social cost of $293 per metric ton (Table 5) would be a reasonable expenditure for society depends on the cost of energy alternatives and specifically on the cost of other renewable energy sources. For example, if switchgrass biomass that costs $293 per metric ton were to be used in a biomass electricity-generating plant with typical plant operating and maintenance costs (Timmons et al. 2007), the final cost of electricity produced (prior to distribution) would be about $0.31 per kilowatt hour. While this cost is likely greater than the future cost of alternatives like offshore wind power or solar photovoltaic energy, it might be tenable in some circumstances, especially since biomass energy is more readily available on demand than other renewable sources are.

Biomass Quantities Available from Switchgrass and Forest

While the social cost of forest biomass appears to be significantly less than for switchgrass, cost is not the only concern. Another question is how yields for forest and switchgrass biomass compare since one of the motivations for switchgrass production is greater output per unit of land. Kelty, D'Amato, and Barten (2008) estimated a maximum sustainable forest biomass yield of about 809,000 dry metric tons per year in Massachusetts, which could supply less than 1 percent of the state's current primary energy use.

In Massachusetts, most forest is native second growth that grew as farms were abandoned rather than intentional plantings (Foster, Motzkin, and Slater 1998). Plantation-style forestry is possible, as is production of tree crops such as willow, which may receive nitrogen fertilizer applications at levels similar to switchgrass (Adegbidi et al. 2001). While there is a continuum of production-intensity options, this study considers only switchgrass and second-growth native forests as found in most of New England.

To better assess when there might be an incentive to produce crop-based biomass instead of forest biomass, we conduct an experiment using the ALMANAC model previously described that compares biomass yields from native forest and switchgrass. Are switchgrass crop yields actually greater, and if so, how much?

ALMANAC estimates switchgrass yields on forest land in Berkshire, Hampden, Hampshire, and Worcester counties as though the forest had been removed and replaced by switchgrass. Table 6 compares estimated forest growth and possible switchgrass growth by county. ALMANAC provides switchgrass yield estimates for approximately 75 percent of forest land (since some forest soils have parameter values outside of the range ALMANAC can use for estimates, indicating likely unsuitability for switchgrass). Forest-growth data obtained from the U.S. Forest Service's Forest Inventory and Analysis (FIA) Program provide a basis for comparison. We estimate forest growth in dry tons by county using FIA data for dry biomass stock in tons and stock volume in cubic feet. For comparison, Table 6 shows forest growth totals adjusted to the same area as the switchgrass yields obtained from ALMANAC. With no nitrogen fertilizer, switchgrass dry tonnage exceeds forest output by about 47 percent. This may

Table 6. Forest and Switchgrass Yield Estimates in Western Massachusetts

	Berkshire County	Hampden-Hampshire Counties	Worcester County	Four-County Area	Four-County Switchgrass vs. Forest
Forest biomass growth estimated from U.S. Forest Service FIA data, dry metric tons	452,954	534,672	597,374	1,585,000	
Percent of forest land with switchgrass yield estimate	67	78	79	75	
Forest biomass growth at switchgrass land percentage, dry metric tons	304,897	418,188	473,278	1,196,363	
ALMANAC switchgrass yield estimate, 0 kg/ha of nitrogen, dry metric tons	372,002	601,058	786,922	1,759,982	+47%
ALMANAC switchgrass yield estimate, 67 kg/ha of nitrogen, dry metric tons	681,075	975,086	1,274,930	2,931,091	+145%
ALMANAC switchgrass yield estimate, 135 kg/ha of nitrogen, dry metric tons	910,449	1,489,653	1,949,988	4,350,090	+264%

be a high estimate of yield difference since the omitted forest areas (with no switchgrass yield estimates) are likely to have smaller than average yields of switchgrass (given that the soils are likely unsuitable for switchgrass). But with additions of nitrogen fertilizer, switchgrass biomass yields clearly exceed those of native forests—by a factor of 2.6 for switchgrass using 135 kg/ha N. To a large extent, switchgrass is a vehicle for utilizing growth-enhancing nitrogen.

Conclusions and Policy Implications

For biomass energy, land is a key resource that would be needed in large quantities. Production of biomass energy generates significant externalities from this land, costs and benefits that accrue to society as a whole rather than to producers. For biomass crops, negative externalities are generated from nitrogen fertilizer use with nitrate polluting groundwater aquifers and contributing to eutrophication. Nitrogen fertilizers also add to greenhouse gas emissions and other air pollution problems. And in areas where crops replace native forests, forest ecosystem services are lost.

The total cost of biomass energy crops hinges to a large extent on associated ecosystem service values. Of the estimated $293/Mg switchgrass social cost, private production cost represents only 33 percent (Table 5). While nitrogen fertilizer externalities contribute 6 percent of total cost, the largest cost portion (61 percent) comes from the foregone value of forest ecosystem services (if ecosystem service values presented in Table 4 hold for Massachusetts). Although social cost of production is likely smaller for forest biomass than for switchgrass in most circumstances, the magnitude of the difference depends largely on the marginal value of ecosystem services.

Context has a large impact on ecosystem service values. For example, the marginal value of forest land may be low in a landscape that is already predominantly forested. In Massachusetts, then, there may be smaller differences between crop and forest biomass values, at least for small increases in crop production. Conversely, the marginal value of a land cover like forest may be much higher in the less-forested landscapes of the central United States, where biomass crops are more actively being considered. In such areas, appropriately placed forests could both produce biomass energy and provide ecosystem services with high marginal value.

In Massachusetts, switchgrass could provide an estimated 2.6 times more biomass per hectare than native forest growth (Table 6). A significant policy question is under what circumstances the renewable energy provided by switchgrass might have marginal benefits of at least $293 per metric ton, a question that requires comparison of switchgrass to other renewable energy alternatives.

The prominence of externalities in biomass economics suggests that private markets will fail to optimize outcomes and that appropriate public policy is imperative. While kilowatt hours of energy from different sources may be identical, energy production methods may have very different social costs. For biomass energy, public policy could provide incentives to landowners to produce biomass energy in socially optimal ways, such as in forests or native prairies that offer significant ecosystem services (Hill 2007). And if some land covers generate positive externalities for society, mechanisms that compensate landowners for providing those services may be needed (Daily and Matson 2008).

This study demonstrates both the use and limits of ecosystem service valuation for agricultural policymaking. Ecosystem service values are needed for appropriate decisions about land uses, and those decisions will become more critical as populations grow and needs increase for land-derived resources like biomass. While it is difficult even to completely understand ecological processes and more difficult still to ascribe marginal ecosystem service values appropriate to each time and place, one can make useful conclusions with incomplete knowledge. This study shows that approximate values for ecosystem services can be used to describe broad outlines of optimal policy decisions.

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