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# Economic Value of Ecosystem Services from Agriculture

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If ecosystem services describe the benefits that people get from nature, then those services must have value. What are they worth? Can we use values to choose desirable farming systems?

The idea of "value" in the sense of worth can be understood in two very different ways (Heal 2000). *Intrinsic value* refers to inherent worth. *Economic value* refers to relative scarcity. The diamond–water paradox elucidates the difference between the two (Heal 2000). Clean water, which is essential for human life, has great intrinsic value, yet its price is often very low. Diamonds have negligible intrinsic value yet they fetch very high prices. Prices express economic values based on supply and demand. *The amount of a good or service that producers will supply depends on the cost of producing it and the price offered for its purchase. The amount that consumers demand depends on how well they like it and its price of sale*. Economic methods for estimating the values of nonmarketed ecosystem services seek to capture these underlying market relationships.

Although the food, fiber, and bioenergy products from agroecosystems tend to be the only agricultural ecosystem services whose economic values are directly measured by market prices, research from the Kellogg Biological Station Long-Term Ecological Research (KBS LTER) project is beginning to provide estimates of the economic value of the nonmarketed ecosystem services from agriculture. In this chapter, we first introduce the principles for economic valuation of nonmarketed services, then offer a typology of valuation methods, and then review four KBS LTER–related studies we have conducted that estimate the value of nonmarketed ecosystem services.

## Principles for Economic Valuation

A challenge to making sound measurements of nonmarket economic values is to capture the kinds of relationships that exist in markets. Markets are settings where people make choices about buying and selling. Market prices have three key traits: first, they are determined "at the margin." Put differently, prices are linked to quantities, so what a consumer is willing to pay depends on how much that person has already consumed. The price that a consumer and producer agree on is based on how badly the consumer wishes to buy a little more and what it would cost the producer to make that little more (above what each already has bought and produced, respectively). Second, there are limits to what choices are feasible. Consumer purchases are limited by budgets, and producer sales are limited by the productive resources and technology at hand. Third, both producers and consumers have substitutes and complements available to them. They tend to choose the most feasible alternative (not necessarily an extrapolation of current practice). So a farmer whose melon vines bear few fruit due to poor pollination may opt to rent honeybee hives rather than invest in habitat restoration for native pollinators. Some celebrated attempts at placing economic values on ecosystem services have extrapolated values to levels that violate these principles (Costanza et al. 1997), resulting in estimates that have been criticized for not being economically credible (Pearce 1998, Bockstael et al. 2000).

Economic valuation of ecosystem services uses methods that attempt to capture the effects of relevant markets. Those markets may be real or imagined. The relevant market for an ecosystem service varies with the scale over which people experience that ecosystem service. The nutrient cycling service of a soil microbial community may be fully captured at the farm field scale, whereas the climate regulation service rendered by the same microbial community (e.g., uptake of atmospheric methane) is realized only at the scale of global climate. For ecosystem services from agriculture, this scale effect means that farmers may care about certain services that directly benefit the farm, while viewing others as external to their management decisions (see focus group results in Swinton et al. 2015, Chapter 13 in this volume).

Depending on how consumers and producers experience an ecosystem service, there are many different methods to estimate its value (Freeman 2003, Shiferaw et al. 2005). The methods used for agricultural ecosystem services focus on values that people obtain from *use* of the services. Nonuse values—from existence of an ecosystem, the opportunity to pass it on intact to the next generation, or the possibility of discovering unknown benefits from it—are assumed to matter little in agricultural ecosystems. Research on economic valuation of agricultural ecosystem services in KBS LTER–related cropping systems can be divided into two strands. Revealed preference methods capture values revealed by existing markets. Production input, profitability trade-off, and stated preference methods estimate the value of changes to the status quo, such as changing current farmer cropping systems to include ecologically recommended practices.

## Revealed Preference Estimates to Value Landscape-Level Ecosystem Services

The economic value of landscape-level ecosystem services such as wildlife habitat and recreation can be inferred from existing markets for land and recreational services. Revealed preference methods use information on expenditures and market prices to deduce the implied willingness to pay for environmental benefits. For example, the value of recreational fishing and hunting services can be inferred from what people spend to travel to fishing and hunting sites and from the characteristics of the sites they visit. Another revealed preference approach, hedonic valuation, uses price data and product characteristics to infer the component value of those characteristics by statistical regression methods. Just as real estate analysts use hedonic valuation to estimate the value that a second bathroom adds to a house, environmental economists can use the same method to estimate the value that adjacent forest adds to a farm field. Both travel cost and hedonic land price analyses have been used by KBS LTER economists (e.g., Knoche and Lupi 2007, Ma and Swinton 2011).

Among its many roles, agricultural land provides valuable habitat for wildlife. Hunting is one major ecosystem service experienced by 12.5 million adult Americans in 2006 (U.S. Department of the Interior and U.S. Department of Commerce 2006). Of those, ~750,000 hunted in Michigan. The value of Michigan's agricultural land as wildlife habitat is captured by a travel cost analysis of deer hunting in the state. Knoche and Lupi (2007) used data on the cost of hunting trips to calculate how much hunters are willing to pay for various attributes of the hunting experience and found that hunters were effectively paying $39 per acre for access to 10% of the private agricultural land in the southern Lower Peninsula of Michigan. This represents 7% of the per-acre market value of farm products in the area in 2004, a significant value (part of which is already captured by farmers who offer hunting leases for their lands). By providing a varied landscape with abundant food, agriculture enhances the habitat for deer. Knoche and Lupi (2007) estimated that in a nonagricultural landscape that supported only half as many deer, the annual value to hunters would decline by $15 million.

Hedonic valuation of ecosystem services through land prices can capture the values of a range of services, as compared to the single value from the travel cost of hunting trips. The price of a land parcel represents a bundle of attributes embodied in that parcel. Hedonic analysis applies statistical regression of land prices to different attributes of the parcel to infer the values of specific property traits (Palmquist 1989, Palmquist and Danielson 1989). In an attempt to measure the value of land-based ecosystem services in the KBS vicinity, Ma and Swinton (2011) estimated a hedonic model of land prices in four counties surrounding KBS—Allegan, Barry, Kalamazoo, and Eaton (Fig. 3.1). This work included variables describing traits of both the natural and built environments and subdivided these into traits that affect both the production value of the land and its consumption value (e.g., residential and recreational attributes). To capture the effect of surrounding ecosystems, the study analyzed spatial data on the proportions of land cover in a 1.5-km radius around property parcels.

The study inferred ecosystem service values from the influence that particular landscape features had on agricultural land prices in southwestern Michigan (Ma and Swinton, 2011). Three lessons stood out. First, recreational and production-supporting services tend to make the largest contributions to land values. On-site lakes and woodlands as well as nearby rivers, wetlands, and conservation lands enhance values, while on-site streams (which can flood) detract from value. Second, certain ecosystem services are likely to be only partially capitalized into land prices, either because landowners are unaware of their value or because the benefits are dispersed to areas external to the parcel. Examples include beneficial soil microbial communities and habitat for pollinators and natural enemies of agricultural pests. Third, it appears that land prices do not reflect benefits that are largely realized outside the parcel, such as greenhouse gas mitigation or habitat for large wildlife.

The contribution of the surrounding landscape to agricultural land prices is particularly meaningful in light of expanding research into the provision of ecosystem services at the landscape scale that involves multiple property owners. To date, much of the research on how landscape structure affects ecosystem services to agriculture has focused on arthropod-mediated ecosystem services, such as natural biocontrol of insect pests (Costamagna and Landis 2006, Gardiner et al. 2009, Landis and Gage 2015, Chapter 8 in this volume) and pollination (Kremen and Ricketts 2000, Kremen et al. 2002, Steffan-Dewenter et al. 2002, Ricketts et al. 2004, Ricketts et al. 2008). However, recreational and production-related ecosystem services at the landscape level are also significant, based on Ma and Swinton's (2011) research using rural property prices. For example, when the proportion of wetlands rose 1% in the 1.5-km radius of surrounding land, land parcel prices rose by 3%, suggesting that land markets place value on certain ecosystem services of wetlands (perhaps

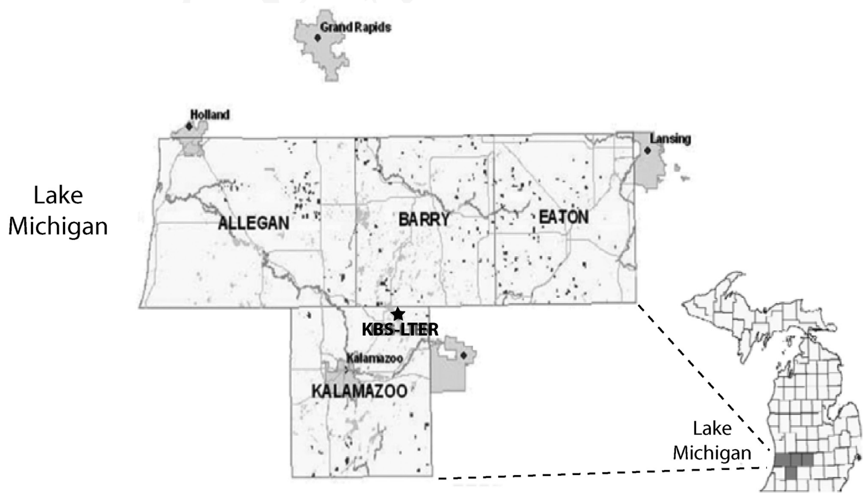


Figure 3.1. Property parcel locations (small black polygons) used in the hedonic study of ecosystem service values embodied in southwestern Michigan land prices. Counties and major cities are also shown. Redrawn from Ma (2010).

flood buffering capability, wildlife habitat, open space). A similar finding was that a 1% rise in surrounding conservation lands (e.g., natural preserves, parks, public forests) raised parcel prices by 2%. Value for purposes of recreation and crop irrigation likely explains the finding that parcel prices rose by 6% for each kilometer closer to a river (Ma and Swinton 2011). That some of these sources of landscape-level value result from land management choices—set-aside of conservation land and preservation of wetlands—points to the potential for landowners to improve land values by coordinating management across parcels within a given landscape, a direction deserving future research.

## Valuing Ecosystem Services from Improved Cropping Practices

While land prices embody the local value of certain ecosystem services that emerge from landscape composition, other methods are needed to measure environmental values of specific farm management practices.

Developing estimates of economic values due to specific management practices is a two-stage process. Applying the framework of Collins et al. (2011), the first stage is to measure the changes in ecosystem service flows resulting from a change in crop management practices. The change must be measured from some baseline, such as the conventional corn-soybean-wheat management system of the KBS LTER Main Cropping Systems Experiment (MCSE) (Table 3.1) (Robertson and Hamilton 2014, Chapter 1, this volume). The second stage converts those changed service flows into economic values. As economic values, these are based on real or hypothetical markets that measure how much the people who gain would be willing to pay to obtain the changed service flows, or how much the people who lose would accept in order to be equally well off as they were before the change (Polasky and Segerson 2009). So, for example, if farmers reduce fertilizer use that prevents a lake from becoming eutrophic, economic value would be measured on the demand side by how much the lake users are willing to pay for maintaining its uses and on the supply side by how much farmers would be willing to accept as compensation for any income lost due to reduced fertilizer use. Where markets for ecosystem services or their near substitutes exist, prices may reflect an economic equilibrium where the value to those who gained from a specific change in ecosystem service is in balance with the compensation to those who lost by making the necessary management changes. Where markets do not exist, aspects of markets can be simulated to infer economic values.

The MCSE results point to several ecosystem services that alternative management of cropping systems can provide: nutrient cycling (biological in lieu of synthetic chemical fertilizer supplements), crop pest regulation (via natural biocontrol), climate regulation (via reduced greenhouse gas emissions), and water-quality regulation (via reduced nutrient leaching to groundwater and runoff to surface waters). Details can be found in other chapters in this volume (Paul et al. 2015, Chapter 5 in this volume; Landis and Gage 2015, Chapter 8 in this volume; Hamilton 2015, Chapter 11 in this volume; and Gelfand and Robertson 2015, Chapter 12 in this volume).

Table 3.1. Description of the KBS LTER Main Cropping System Experiment (MCSE) examined in economic valuation.[a]

| Cropping System | Dominant Growth Form | Management |
|---|---|---|
| *Annual Cropping Systems* | | |
| Conventional (T1) | Herbaceous annual | Prevailing norm for tilled corn–soybean–winter wheat (c–s–w) rotation; standard chemical inputs, chisel-plowed, no cover crops, no manure or compost |
| No-till (T2) | Herbaceous annual | Prevailing norm for no-till c–s–w rotation; standard chemical inputs, permanent no-till, no cover crops, no manure or compost |
| Reduced Input (T3) | Herbaceous annual | Biologically based c–s–w rotation managed to reduce synthetic chemical inputs; chisel-plowed, winter cover crop of red clover or annual rye, no manure or compost |
| Biologically Based (T4) | Herbaceous annual | Biologically based c–s–w rotation managed without synthetic chemical inputs; chisel-plowed, mechanical weed control, winter cover crop of red clover or annual rye, no manure or compost; certified organic |
| *Perennial Cropping Systems* | | |
| Alfalfa (T6) | Herbaceous perennial | 5- to 6-year rotation with winter wheat as a 1-year break crop |
| Poplar (T5) | Woody perennial | Hybrid poplar trees on a ca. 10-year harvest cycle, either replanted or coppiced after harvest |

[a]Codes that have been used throughout the project's history are given in parentheses. Systems T1–T7 are replicated within the LTER main site. For further details, see Robertson and Hamilton (2015, Chapter 1 in this volume).

### *Inferring Value of Pest Regulation by Natural Enemies Using the Production Input Method*

When an ecosystem service can substitute for an existing marketed input or when the service contributes to measurable marketed output, the economic value of changes in the level of the service can readily be inferred using information from the related input and/or crop (output) markets (Freeman 2003, Drechsel et al. 2005). An example of a widespread application of this method is the calculation of the fertilizer replacement value to measure the value of biological nutrient cycling in cereal–legume systems (Bundy et al. 1993).

Recognizing an opportunity to apply experimental results, KBS LTER researchers used the factor input method to estimate the value of a loss in natural pest biocontrol due to changed crop cover. Field research had revealed that more corn area in the landscape reduces natural biocontrol of the soybean aphid (Gardiner et al. 2009). When U.S. Midwest corn acreage jumped 19% in 2007 in response to soaring prices, Landis et al. (2008) realized that this change reduced habitat for natural enemies of the soybean aphid. They calculated the lost value of natural biocontrol services based on predicted soybean yield loss and associated increased insecticide costs. They estimated impacts both on farmers who follow integrated pest management (IPM) to guide insecticide sprays and on the roughly 1% of farmers who rely entirely on natural biocontrol. Depending on whether the soybean price was the 1996–2007 median or the higher post-2007 level, the reduced value of biocontrol services to soybeans due to the 19% increase in corn acreage was estimated to be \$18–25 $ha^{-1}$ for IPM farmers and \$110–199 $ha^{-1}$ for natural biocontrol farmers (Landis et al. 2008).

Another KBS LTER approach to measuring the economic value of the soybean aphid natural enemy complex used a bioeconomic model that predicted densities of the pest and its predators (as opposed to habitat features in the landscape). Zhang and Swinton (2009) developed a dynamic optimal control model for soybean aphid IPM that incorporates the economic value of natural enemy survival when making profit-maximizing decisions on insecticide applications to the aphid. They found that natural enemies are particularly valuable for suppressing low populations of soybean aphid, preventing them from multiplying to the point of causing significant crop damage. A single, typical natural enemy (comparable to the multicolored Asian lady beetle, *Harmonia axyridis*) per soybean plant is worth \$32.60 $ha^{-1}$ when 5 to 30 aphids are present per plant during its early flowering stage (Fig. 3.2). However, above 30 aphids, the value of a single natural enemy would fall to just \$4.20 $ha^{-1}$, because at higher aphid populations, a single natural enemy could not control the infestation so insecticides would be needed to maximize profit (in spite of the collateral or nontarget damage to natural enemy populations) (Zhang and Swinton 2012). Based on evidence that soybean aphid density averaged 21 per plant in

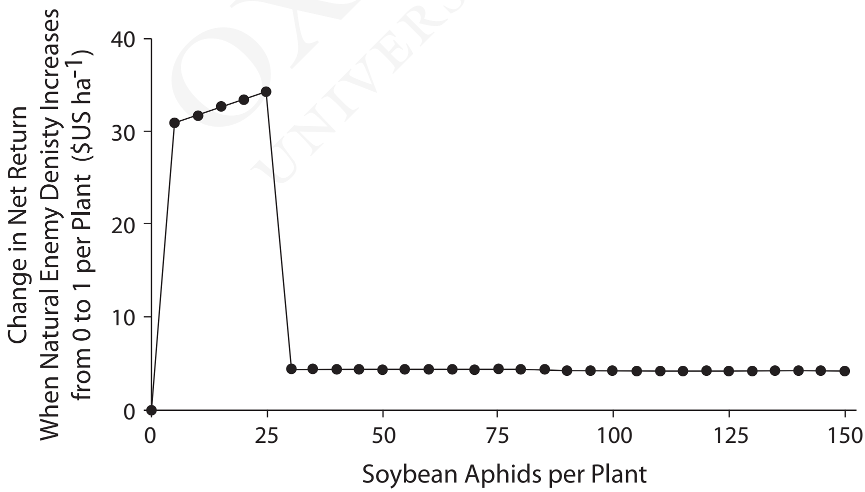


Figure 3.2. Value of one natural enemy per soybean plant (as compared to none) as a function of the abundance of soybean aphids. The aphids are assumed to be in reproductive stage R1, the natural enemy is assumed to have a daily predation rate of 35 aphids, and the initial soybean yield potential is assumed to be 2.69 Mg $ha^{-1}$. Redrawn from Zhang and Swinton (2012) with permission from Elsevier.

2005, the authors estimated that the value of going from no natural enemies to one per soybean plant is worth \$32.60 $ha^{-1}$, amounting to some \$84 million during 2005 for the U.S. states of Iowa, Illinois, Michigan, Minnesota, and Wisconsin. This estimate is based solely on natural enemy contributions to soybean profitability, so it ignores other sources of value to society, such as the health and environmental benefits of reduced insecticide use. Because the model uses the factor input valuation method, it is sensitive to assumptions about the aphid predation rate per natural enemy, as well as the price and pest-free yield of soybean.

### *Estimating Economic Supply and Demand for Ecosystem Services When Markets Are Missing*

The economic valuation of climate-regulating and water-quality–regulating ecosystem services is more complicated because these services lack the direct market links of agricultural pest regulation services. Markets designed for climate- and water-quality–regulating services have been piloted since the late 1990s. However, localized water-quality nutrient trading has not been established successfully in the United States for a variety of reasons (Hoag and Hughes-Popp 1997), many of which also apply to carbon trading markets.

Developing economic values for these regulating services calls for understanding the cost to producers of supplying them and the benefits to consumers of enjoying them. Valuation of both costs and benefits can be measured by studying trade-offs (Polasky and Segerson 2009). On the supply side, how much would farmers need to be compensated to adopt practices that reduce net greenhouse gas emissions and/or reduce waterborne nutrient losses from topsoil? On the demand side, how much would consumers be willing to pay to be equally well off with improved climate and water quality vs. degraded environmental quality?

Farmers earn their livelihoods from farming, so income matters a great deal. But farmers generally make management decisions based on the welfare of their households using more complex criteria than simply profit maximization. When eliciting from farmers how they weigh trade-offs between increased production costs and increased ecosystem services, these other objectives are automatically factored in. So-called stated preference methods of economic valuation that are described below can be used to capture this complex decision-making process.

But when analyzing ecological experimental data—such as that from the MCSE—making the simplifying assumption that farmers aim to maximize profit enables a trade-off analysis of private profitability compared to the public benefits from ecosystem services. A common feature of many ecosystem services is that they generate benefits beyond the boundary of the farm. These benefits may not be factored into the farmer's decisions, especially if generating them entails added costs. Given that agroecosystems generate a multiplicity of products and services, trade-off analysis enables comparing these outputs, typically using profitability as a numéraire for comparison (Wossink and Swinton 2007).

### *Trade-off Analysis of Profitability vs. Ecosystem Service Provision*

Trade-off analysis can illustrate the relationship between profitability and ecosystem services such as greenhouse gas fluxes or nitrate leaching for the MCSE systems (Antle and Capalbo 2002). When graphed in two or three dimensions, the method provides a visual illustration of trade-off vs. win-win outcomes for farmers and the public. It also permits an indirect way to calculate the cost to the farmer of increasing output of nonmarketed ecosystem services. Farmers face two kinds of monetary costs: direct costs and opportunity costs. Direct costs are subtracted from revenues to calculate profitability. Opportunity costs are measured indirectly, as the difference in earnings between the most profitable system and an alternative. Trade-off analysis can measure the opportunity cost of reduced profitability in exchange for increased supply of ecosystem services.

Trade-off analysis can also be used to evaluate the efficiency of providing targeted outcomes. By comparing profitability and ecosystem service outcomes for the full set of MCSE systems, it identifies some systems that do not excel in any outcome. Such systems are termed “inefficient” because other ones (either alone or in combination) could provide the same or higher levels of all desired outcomes.

A caveat for trade-off analysis is that it only captures the supply side of economic value, focusing on the marginal cost to the farmer of providing more of an ecosystem service. In our research on the MCSE, we use budgets with static prices, so the analysis implicitly assumes that any shift to an alternative cropping system would be sufficiently limited in scale that it would not generate market price feedbacks.

The KBS LTER trade-off analyses begin with partial enterprise budgets for the MCSE systems. Annualized partial enterprise budgets were calculated by Jolejole et al. (2009) using standard enterprise budgeting techniques (Boehlje and Eidman 1984), with a focus on only those costs that vary across systems, as per the CIMMYT (1988) methodology for analysis of agronomic data. For systems involving perennial crops, net present values were calculated over the crop lifetime and converted to an annualized value using a standard financial annuity formula (Weston and Copeland 1986). The resulting profitability measure is the gross margin, which represents revenue above costs that vary. Gross margins capture the differences among MCSE cropping systems, although they do not account for other kinds of costs that are unchanging across MCSE systems but tend to vary substantially from farm to farm (e.g., land rental, compensation of family labor). Mean values for global warming impact (GWI) were compiled for all MCSE systems by Robertson et al. (2000) and Syswerda et al. (2011) and for nitrate leaching by Syswerda et al. (2012).

Global warming impact results show that among the six MCSE cropping systems evaluated during 1993–2007, Poplar had the lowest overall impact (–105 g $CO_2$e m$^{-2}$ yr$^{-1}$ or –1.05 Mg ha$^{-1}$ yr$^{-1}$, where the negative sign connotes net $CO_2$ uptake from the atmosphere), but it was also one of the least profitable cropping systems (\$73 acre$^{-1}$ or \$179 ha$^{-1}$), though more profitable than Alfalfa. In profitability, the No-till system dominated at standard prices (\$139 acre$^{-1}$ or \$345 ha$^{-1}$), while the Biologically Based certified organic system dominated at organic prices (\$185 acre$^{-1}$or \$458 ha$^{-1}$) (Fig. 3.3).

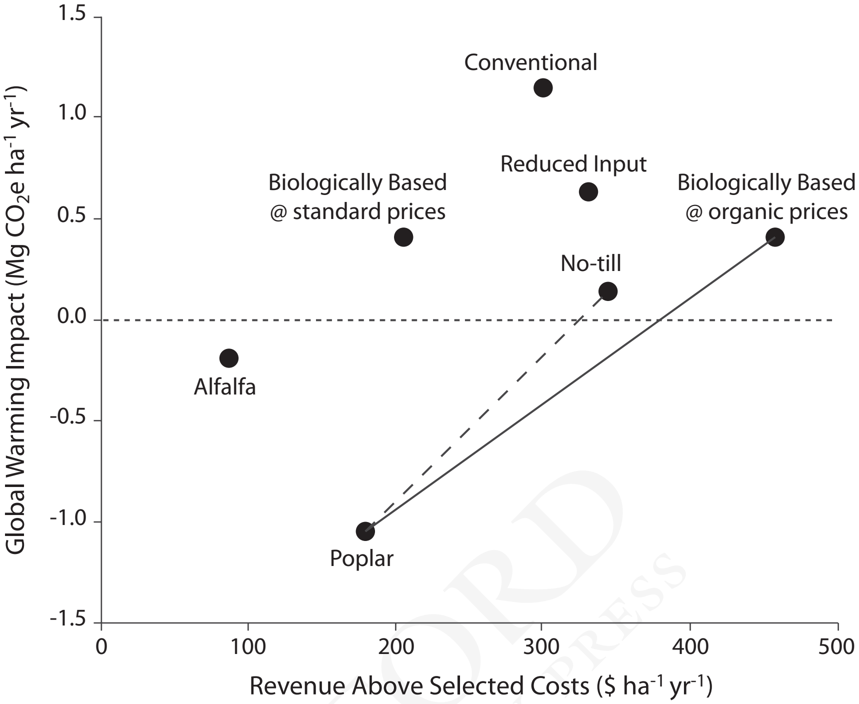


Figure 3.3. Trade-offs between global warming impact (in $CO_2$ equivalents, $CO_2$e) and revenue above selected costs for different Main Cropping System Experiment systems, 1993–2007 (except for Alfalfa, 1989–2004 and Poplar, 1989–1998). See text for explanation of lines. Adapted and updated from Jolejole (2009) and Robertson et al (2000).

Two important findings emerge. First, four of the MCSE systems are relatively inefficient (Conventional, Reduced Input, Poplar, and Biologically Based at standard nonorganic prices) in that other systems alone or in combination could provide better outcomes for both profitability and GWI. As illustrated in Table 3.2, at nonorganic prices, the No-till system offered greater profitability and lower GWI than the other three annual cropping systems (Conventional, Reduced Input, and Biologically Based). So switching from any of those systems to No-till would improve one or both outcomes. Likewise, Poplar dominates Alfalfa in both dimensions, so both profitability and GWI outcomes could be improved by switching land from Alfalfa to Poplar. The dashed line in Figure 3.3 illustrates the efficient frontier connecting the points that are efficient in the sense that at nonorganic prices, no other system excels in terms of both profitability and GWI. At organic prices, No-till is not dominated by Biologically Based alone (No-till has lower GWI), nor is it dominated by Poplar alone (No-till has greater profitability), but it is dominated by a combination of the two. So at organic prices, shifting land from No-till to a combination of the efficient systems (roughly three-quarters Biologically Based and one-quarter Poplar) could improve both profitability and GWI relative to No-till alone.

Table 3.2. Efficiency gains and trade-offs from cropping system changes.[a]

| Change in Cropping System | Change in Global Warming Impact (Mg $CO_2$e ha$^{-1}$ yr$^{-1}$) | Change in Profitability (\$ ha$^{-1}$ yr$^{-1}$) |
|---|---|---|
| Efficiency gains | | |
| Conventional to No-till | –1.00 | 45 |
| Conventional to Reduced Input | –0.51 | 31 |
| Reduced Input to No-till | –0.49 | 14 |
| Alfalfa to Poplar | –0.85 | 93 |
| Trade-offs along efficient frontier | | |
| No-till to Poplar | –1.19 | –166 |
| Biologically Based to Poplar | –1.46 | –279 |

[a]Based on MCSE systems assuming nonorganic grain prices. Changes in GWI and profitability represent mean outcomes. Negative GWI indicates greenhouse gas mitigation, a positive outcome.
*Source:* Profitability data for 1993–2007 adapted and updated from Jolejole (2009). Global warming impact (GWI) data for 1991–1999 from Robertson et al. (2000).

The second important finding is the very high implied marginal cost per metric ton of reducing GWI by moving along each of the efficient frontiers. The marginal cost is the amount of crop net income given up per metric ton of GWI gain by moving from one efficient system to another along the efficient frontier. Arithmetically, it is the change in revenue above selected costs divided by the corresponding change in GWI. The dashed line between Poplar and No-till shows that the implied cost of reducing GWI by shifting land from the No-till corn–soybean–wheat rotation to Poplar is \$140 Mg$^{-1}$ $CO_2$e ha$^{-1}$ yr$^{-1}$. As illustrated by the solid line from Poplar to Biologically Based, the higher value of certified organic production raises the unit cost of reducing global warming by switching between these systems to \$191 Mg$^{-1}$ $CO_2$e ha$^{-1}$ yr$^{-1}$. These implied costs exceed the traded prices of carbon credits on international exchanges in the early 2000s by an order of magnitude. The implication is that other methods can abate $CO_2$e emissions at far lower cost (e.g., improving efficiency of coal-fired power plants or reducing N fertilizer use). Indeed, if substantial cropland were shifted out of grain crops into poplar, market prices of grain crops would rise and those of poplar would fall, making the implied marginal cost of shifting even greater than shown here.

### *Supply of Crop Land to Boost Ecosystem Service Provision: Application of Stated Preferences to Capture Farm Heterogeneity*

Commercial farm conditions vary in terms of land quality, equipment availability, managerial ability, and farmer attitudes. Ecological experiments like the MCSE intentionally hold all these factors constant, limiting the scope of outcomes that can be explored in a trade-off analysis of experimental results. Farmers, however, vary in their resources, priorities, and perceptions of the costs and benefits of farming activities, and this variability generates a number of trade-off outcomes that can be compared. Jolejole (2009) and Ma et al. (2012) captured all three of these aspects of heterogeneity directly in their analyses of the 2008 Crop Management and Environmental Stewardship Survey of Michigan corn and soybean farmers.

The survey was motivated, in part, by a desire to understand why most Michigan field crop farmers choose different cropping systems than the corn–soybean–wheat rotations of the MCSE. In Michigan, corn and soybean are widely grown, often in a two-crop rotation, but wheat is only sometimes included in the rotation. In Michigan during 2006–2010, mean planted areas for corn and soybean were 970,000 and 790,000 ha, compared to 250,000 ha for wheat (NASS 2011). No-till crop farming has expanded greatly during the lifetime of the KBS LTER. By 2006, 48% of soybean land was farmed without tillage in Michigan, which reflected the national trend of 45%. Rates for corn and wheat were less than half this level (Horowitz et al. 2010), and only a fraction of the no-till area was in permanent no-till, as in the MCSE. Cover crops, which the MCSE uses to furnish nitrogen, augment soil organic matter, and prevent soil erosion, were planted on less than 20% of U.S. commercial family farms in 2001, with rates slightly lower on grain farms (Lambert et al. 2006). The same study found that fewer than 30% of farmers conducted soil tests before planting corn and soybean crops.

The purpose of the survey was to understand why Michigan corn and soybean farmers farm as they do, and how they perceive conservation practices like growing wheat, planting cover crops, and reducing fertilizer rates. The survey used contingent valuation methods to elicit whether farmers would be willing to adopt some of these practices in exchange for payments.

The survey questionnaire asked respondents to answer questions regarding four proposed cropping systems. The systems proposed to farmers were loosely based on MCSE Reduced Input and Biologically Based systems, but the first two proposed systems omitted wheat because it is less commonly grown in the region than the other two crops. The proposed systems were:

A. a chisel-tilled corn–soybean rotation fertilized according to university recommendations based on soil testing, including a pre-sidedress nitrate test for corn;
B. same as system A with winter cover crops added;
C. same as system B with winter wheat added to the rotation after soybean; and
D. same as system C but with fertilizer and pesticides reduced by one-third by banding applications over crop rows.

In order to elicit their willingness to change practices in exchange for payments, respondents were asked the following question:

> "If a program run by the government or a nongovernmental organization would pay you $X per acre each year for 5 years for using cropping system (Y), how many acres of land would you enroll in this program?"

The question presented them with a predetermined payment level ($X), and the question was repeated with different payment levels for each of the four systems. If respondents answered that they would not participate, they were asked if they would be willing to consider participating in exchange for a higher payment. The questionnaire was sent out in 16 different versions that varied three experimental factors: (1) the payment levels offered, (2) whether the payment came from the government or a nongovernmental organization, and (3) whether the sequence of cropping systems went from least complex (A) to most complex (D) or vice versa. The sample of 3000 Michigan corn and soybean farms was stratified by farm size into four levels: under 100 acres, 101–500 acres, 501–1000 acres, and over 1000 acres. The sampling and mailing lists were managed by the National Agricultural Statistics Service Michigan field office. Usable responses were received from 1688 farms, representing a response rate of 56% (Jolejole 2009).

The econometric analysis of farmer willingness to change was divided into two steps: willingness to consider participation in the program (probit statistical model) and, for those willing to participate, the number of acres they would enroll (tobit regression) (Ma et al. 2012). The determinants of farmers' willingness to adopt these alternative systems differed sharply between the two levels of analysis.

Farmer conservation attitudes, prior experience, and equipment availability largely drove their willingness to consider participating in the hypothetical program to shift land into the proposed cropping systems in exchange for a payment (Ma et al. 2012). Respondents who agreed with the statement "nature provides services that improve my crop production" were 5% more likely to consider the program. Likewise, farmers with prior experience in federal agricultural programs that pay farmers for environmental stewardship practices were more likely to consider this program (though farmers involved in a state environmental assurance program were not). Not surprisingly, farmers who were already doing similar practices (such as no-till or planting wheat) were more inclined to consider proposed practices that were similar. This effect may be linked to the fact that farmers who owned the necessary equipment (e.g., band applicator for fertilizer or pesticides) were more prone to consider participating with the relevant practice than those who did not.

For those willing to consider participating in the program that would pay them for changed cropping practices, how much land they would enroll depended chiefly on benefit–cost and feasibility criteria (Ma et al. 2012). Most important was the size of the payment offered. For the obvious feasibility reason, farmers with more total cropland area would offer to enroll more land in the program. On the other hand, farmers using moldboard plows tended to enroll smaller acreages in the program.

The supply of land that farmers were willing to dedicate to specific cropping systems over a range of different subsidy payments represents their perceived underlying costs and benefits from adopting those practices. Some of these were the direct costs and opportunity costs discussed previously with the trade-off analysis of MCSE systems. But the land area offered for conservation practices also reflects the attitudes and preferences of individual farmers. The importance of attitudes was illustrated by increased program enrollment of farmers who expressed the belief that nature benefits the farm and who had previously participated in environmental programs. So the supply of land that respondents would devote to low-input cropping practices reflects not just farm heterogeneity due to differences in land and equipment, but also farmer heterogeneity due to differences in attitudes and management ability.

Two general patterns emerged from the supply response analysis. First, farmers were willing to supply much more land for System A, the simplest cropping system, than for the three systems that involved cover crops or more complicated management (Ma et al. 2012). In economic terms, this greater price elasticity of supply meant that for a given increase in payment, farmers would offer to devote more land to System A than to the other cropping systems. Second, farmers with over 500 acres (202 ha) were much more willing than operators of smaller farms to respond to higher payments by offering more acreage, especially for System D (Jolejole 2009). It was evident that these larger farms are the low-cost suppliers of environmental services. So payment-for-environmental-services policies that aim for cost-effective gains will likely achieve most of their impact from larger farms.

For measuring the economic value of *individual* ecosystem services from agriculture, the analysis was indeterminate for an important reason: management decisions affect multiple ecosystem services simultaneously. Put colloquially, ecosystem services come in bundles. A given agroecosystem generates ecosystem services in relatively fixed proportions (Antle and Capalbo 2002, Wossink and Swinton 2007). There exists no sound method for allocating costs among the different system outputs without an understanding of consumer demand for them.

### *Consumer Demand for Land-Based Ecosystem Services*

How do consumers value the kinds of ecosystem services that farmers can help to produce? The answer to that question can inform the demand side of economic valuation for these services.

Few consumers perceive ecosystem services as scientists do. Ecosystem services like climate regulation, water quality regulation, nutrient cycling, and pest population regulation are meaningful to ecologists, but opaque to the general public. As a first step before designing a consumer survey, Chen (2010) developed a graphical model of how agricultural practices generate intermediate environmental changes that lead to the ecosystem services experienced by the general public (Fig. 3.4). To highlight one set of relationships in the figure, crop fertilization and tillage and their effects on nutrient cycling may carry little meaning for most citizens. But when lakes become eutrophic as a result of excess nutrients, the meaning to recreational users is very clear.

Based on the literature and pretests of the questionnaire, KBS LTER economists focused on two high-profile endpoints: the proportion of eutrophic lakes and percentage of progress toward international goals for abatement of climate change. The survey population were all residents of the state of Michigan. The 2009 Michigan Environmental Survey went to 6000 Michigan households stratified by population in each county to cover the full geographic extent of the state; the final response rate was 41% (Chen 2010). Respondents were first presented with information about climate change and eutrophication of lakes, along with the links between land management practices and changes in those outcomes. Householders were

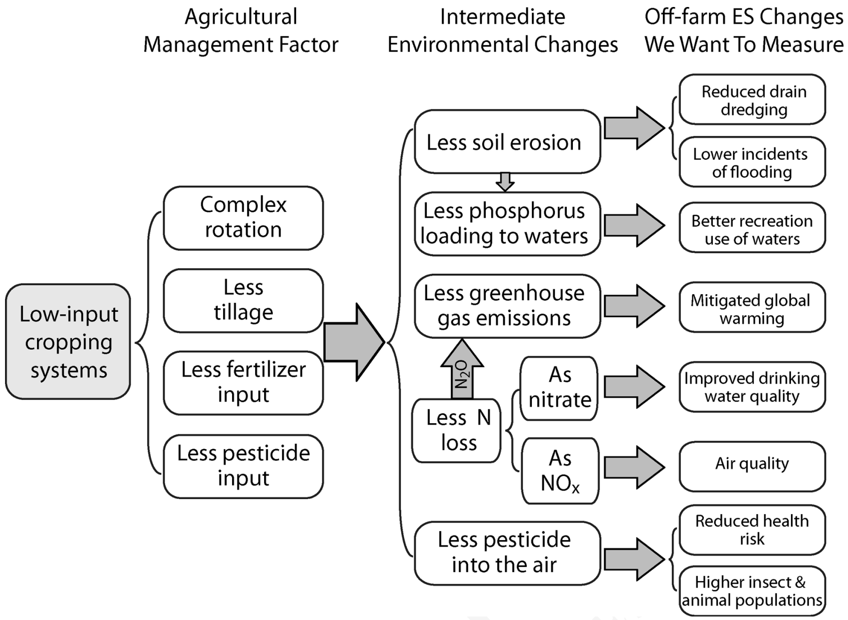


Figure 3.4. Conceptual model of linkages from agricultural management factors to off-farm ecosystem service (ES) changes. Modified from Chen (2010).

then presented with three land stewardship programs, with each proposing to make different changes in (1) the number of lakes with excess nutrient levels and (2) the percentage change in greenhouse gas emissions that scientists estimate is needed to slow global warming. For each of three programs, respondents were asked:

> "Would you vote for program (Y) if it increased income taxes and your share of the increased tax was $X per year?"

The questionnaire was mailed in 14 versions, varying the tax rate ($X), the levels of eutrophic lakes and greenhouse gas abatement, and whether the recipients of the program payments were described as farmers or land managers.

The survey found significant public willingness to finance policies that would pay land managers for changed practices to mitigate lake eutrophication, but less support for financing mitigation of global warming (Chen 2010). The overall mean marginal willingness to pay of Michigan residents was $175 per household per year to reduce the number of eutrophic lakes by 170 and to reduce greenhouse gas emissions by 0.52% of their 2000 levels. They did not care whether the funds for changed land management went to farmers or other land managers.

Support for cleaner lakes was clear-cut. Respondents were willing to pay $0.45 per eutrophic lake per household per year, or $1.7 million annually per eutrophic lake, based on the 3.8 million households in Michigan.

Most households were unwilling to pay for reduced greenhouse gas emissions. This finding was due, in part, to the fact that 60% of households were unconcerned about climate change. But the unwillingness to pay may also have resulted from the smallness of the potential emission reductions—just 0 to 1.2%—based on the crop systems proposed in the 2008 Crop Management and Environmental Stewardship Survey. Ordinarily, economists expect that people will pay more to buy more. But statistical tests showed that willingness to pay was unaffected by the level of proposed reduction in greenhouse gas emissions. Hence, overall mean willingness to pay for reduced emissions was zero. Among the 40% of households that were concerned about climate change, however, the mean household would pay $141 per year for a 1% reduction, compared to year 2000 greenhouse gas emission levels. Scaling up to the 1.52 million Michigan households that were concerned about climate change, this would amount to $214 million annually for a 1% reduction in greenhouse gas emissions, compared to the year 2000 level.

### *Linking Demand and Supply for Ecosystem Services from Agricultural Stewardship*

KBS LTER research into how people judge the economic value of ecosystem services from agriculture in the region around KBS has reached a critical stage for establishing value at the equilibrium between supply and demand. On the supply side, the results clearly document the willingness of corn and soybean farmers in Michigan to change their cropping practices so as to generate more ecosystem services if paid to do so. Farmers would expand both the complexity of management practices and the acreage under improved stewardship in response to rising payment levels—generating a supply of land under management for enhanced ecosystem services. On the demand side, state residents are willing to pay for reduced numbers of eutrophic lakes and—some residents at least—for reduced greenhouse gas emissions.

Bringing together the supply and demand sides presents two major challenges. First, the supply units need to be converted from all land area under a given practice to land area *under changed management that provides additional ecosystem services*. Resident taxpayers expect to buy *increases* in ecosystem services, not simply to express gratitude to environmentally minded farmers who were already providing those services. When the land area under stewardship practices offered by farmers is meticulously recalculated to ensure that it refers to practices that bring *additional* ecosystem services, the proportion of enrolled farms that newly adopt each practice may be small—our study revealed just 7% for chisel plow. But for other cropping practices, it may be quite large: 89% for pre-sidedress nitrate test, 96% for cover crop, 70% for adding wheat to a corn–soybean rotation, and 72% for reducing nitrogen fertilization by one-third (Ma 2011).

Second, a bridge is needed to connect the units of supply of ecosystem services with those of demand. Under the hypothetical payment-for-environmental-services program included here (which was modeled on existing U.S. farm programs such as the Environmental Quality Incentives Program, Conservation Reserve Enhancement Program, and Conservation Stewardship Program), farmers were offered payments to supply land under changed practices, not to supply changes in specific ecosystem services. Residents expressed a willingness to pay for enhanced levels of specific services that improve their experiences with lakes and climate. Because changed farming practices generate bundles of changed ecosystem service levels, the most practical way forward is to convert demand for changed ecosystem service levels into the area of land under changed management that would be required to generate jointly the desired levels of change.

Preliminary results using converted supply units, which incorporate stewardship as described above, indicate that what residents are willing to pay for enhanced ecosystem services would cover the costs to farmers of providing them. Moreover, the likely cost of such a program could be covered at the same level as the federal direct subsidy payments made to Michigan field crop farmers during the 2007–2012 period (Ma 2011).

## Cautions and Emerging Opportunities from Economic Valuation of Agroecosystem Services

Ecosystem services to and from agriculture are valuable—both intrinsically and economically. KBS LTER–related research has estimated economic values using methods that range from simple to complex. The simplest methods require the most limiting assumptions. For example, trade-off analysis based on budgeting of experimental results assumes that farmer objectives are few and known (such as profitability and specific environmental outcomes), that prices of agricultural inputs and products are constant and known at levels from the period of study, and that the experimental biophysical setting and management practices are highly representative and known at observed levels. The most complex methods have fewer limiting assumptions, because they explore more fully the heterogeneity and dynamics of human behavioral interactions with ecosystems (e.g., as depicted in the conceptual model in Figure 1.4 in Robertson and Hamilton 2015, Chapter 1 in this volume).

Table 3.3 lists some of the ecosystem services whose economic values have been estimated in KBS LTER research. Each service has a range of estimates and a set of assumptions that arise from the valuation method (Champ et al. 2003, Freeman 2003). All are limited, too, by the time and place of the underlying data, because economic systems—like ecological ones—are subject to complex feedbacks. Hence, extrapolation of values to other settings calls for an understanding of system dynamics, methodological assumptions, and data limitations (Spash and Vatn 2006, Wilson and Hoehn 2006).

Economic valuation of ecosystem services can highlight the potential appeal of changes in agricultural management that deliver enhanced ecosystem services—specifically those supporting and regulating ecosystem services that lack markets. Two broad avenues exist for facilitating this: technological innovation and policy design.

Technological innovation can offer alternative ways to provide such nonmarketed ecosystem services as reduced greenhouse gas emissions and improved water quality. Ecological and economic knowledge from KBS LTER research has direct technological application and possibilities are emerging for manipulating agroecosystem components for newly understood benefits. One example is to inoculate soil

Table 3.3. Values of ecosystem services in agricultural systems: assumptions and indicative estimates.

| Ecosystem Service | Proxy Variable(s) | Valuation Method | Key Assumptions | Units | Value(s) | Source |
|---|---|---|---|---|---|---|
| Pest regulation (soy aphid) | Corn area in 1.5-km radius | Production function and input cost | Crop yield response known to proxy variable. Prices 2005–2008 | $ ha$^{-1}$ yr$^{-1}$ | $20–39 (IPM[a]) $70–264 (no insecticides) | (Landis et al. 2008) |
| Pest regulation (soy aphid) | Natural enemies and aphids per soy plant | Production function and input cost | Crop yield response known to proxy variable | $ ha$^{-1}$ yr$^{-1}$ for 1st NE[b] plant$^{-1}$ | $4–33 | (Zhang 2012) |
| Recreation (swim, fish) | Rivers near agricultural land parcels | Hedonic analysis of land prices | Regression model fully specified; Data: Agric. land in southwest Michigan 2003–2007 | % change in price km$^{-1}$ distance | 3–9% of agric. land price | (Ma and Swinton 2011) |
| Recreation (hunting) | Hunting access to 10% of agric. land in southern Michigan | Travel cost | Full accounting of travel costs to hunt; Data: Hunter mail survey, Michigan 2003 | $ trip$^{-1}$ yr$^{-1}$ | $1.90–2.20 | (Knoche and Lupi 2007) |
| Flood regulation, native habitat | Wetland % area in 1.5-km radius of agric. land parcels | Hedonic analysis of land prices | Regression model fully specified; Data: Agric. land in southwest Michigan 2003–2007 | % price change per % change in wetland area | 2–4% of agric. land price | (Ma and Swinton 2011) |
| Water-quality regulation | Eutrophic lake number | Contingent valuation | Respondents fully understand scenario; Regression model fully specified; Data: Survey Michigan residents, 2009 | $ hhd$^{-1}$ lake$^{-1}$ yr$^{-1}$ kept noneutrophic | 0.45[c] | (Chen 2010) |
| Water-quality regulation; Climate regulation; Nutrient cycling | Changed crop mgmt. practices (one of four crop systems) with additionality | Contingent valuation | Respondents fully understand scenario; Regression model fully specified; Data: Survey Michigan corn/soy farmers, 2008, and Mich. residents, 2009 | $ ha$^{-1}$ yr$^{-1}$ | $30–67 | (Ma 2011) |

[a]Integrated Pest Management (IPM; includes insecticides when pest density exceeds economic threshold).
[b]Natural Enemy (NE; roughly equivalent to multicolored Asian lady beetle, *Harmonia axyridis*).
[c]This marginal value at the median was jointly determined with $141 hhd$^{-1}$ yr$^{-1}$ among the 40% of respondent households that were concerned about global warming for a 1% reduction in greenhouse gas emissions from the level in 2000.
*Note*: hhd = household.

with methane-consuming microbial communities to stimulate greater removal of atmospheric methane, a potent greenhouse gas (Levine et al. 2011). Another possibility is to manage noncrop areas in agricultural landscapes to support the natural enemies of agricultural pests, thereby reducing the need for chemical pest control (Landis et al. 2000). The viability of this strategy hinges on the opportunity cost of not growing crops (Zhang et al. 2010), which in turn depends on land productivity and crop prices. A third possibility is improvement of perennial grain crops such as wheat, so that grain production can be maintained while cycling nutrients and sequestering carbon in deep perennial root systems. This could enhance soil organic matter and reduce greenhouse gas emissions (DeHaan et al. 2005), though great strides remain for perennial grains to become competitive with current systems (Weir 2012). And a fourth is to manage nitrogen fertilizer more precisely to reduce emissions of the greenhouse gas nitrous oxide, using better estimators of fertilizer need, precision or on-the-go fertilizer application, or other emerging technologies (Liu et al. 2006, Millar et al. 2010).

Policy design is the second key to enhanced ecosystem services from agroecosystems. The contingent valuation survey research summarized above suggests that the public (at least in Michigan) is willing to pay what it would cost for farmers to adopt practices that improve water quality and climate. This evidence that the public is open to such a policy justifies research into the design of programs. Viable programs must tackle thorny problems, such as (1) how to monitor invisible management changes like lower fertilizer rates, and (2) how to balance the cost-effective provision of *additional* services desired by taxpayers with fairness in rewarding farmers (some who were practicing good stewardship without payment). Developing viable programs requires a sound understanding of why farmers adopt changed practices (Swinton et al. 2015, Chapter 13 in this volume). One way forward may be to target certain high-benefit practices. For example, reducing fertilizer application reduces nitrate and phosphorus movement to water as well as nitrous oxide emissions to the atmosphere, so a payment motivated by demand-side desire for fewer eutrophic lakes may generate reduce greenhouse gas emissions at no added cost (Reeling and Gramig 2012).

## Summary

Evidence from economic research related to the KBS LTER indicates the potential for crop farming to be managed for higher levels of nonfood services. For example, crop management systems affect greenhouse gas fluxes and water-borne nutrients, which affect climate- and water-quality–regulating services. Land use and cover affect the abundance of natural enemies of crop pests, affecting the biocontrol regulating service.

Economic values of these and other ecosystem services have been calculated using both supply-side and demand-side approaches. KBS LTER research has used land prices and recreational hunting travel costs to estimate the values of recreational and provisioning services. The costs of supplying enhanced services by modifying crop management have been estimated using three methods. Where clear links exist between a specific ecosystem service and changes in input costs or crop yields, marketed products (such as reduced loss of grain yield from natural enemy predation of crop pests), agricultural input costs, and product prices have been used to estimate values of the ecosystem service. Where experimental data exist on how cropping practices link to multiple ecosystem services, trade-off analyses offer a limited way to rule out systems that are inefficient at generating desired services, as in the example provided here from the MCSE.

Understanding the economic value of complex changes in agroecological systems at large scales calls for a third method based on eliciting information from the people who would incur the costs and benefits of those changes. Data from farmers, such as those from contingent valuation surveys, can capture the costs of adopting modified cropping practices in a way that reflects the true heterogeneity of farm resources and people. Estimates of economic value become possible by linking such supply-side data on cost to provide ecosystem services with demand-side data on how much members of the public would willingly pay for those services. Evidence shows that the public is willing to pay for many ecosystem services at rates that many farmers find acceptable, but a challenge is to find practical ways to design an efficient and fair payment system for farmers supplying those services.

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