Private Incentives for Sustainable Agriculture: Synthesis

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- Environmental Economics and Policy
- Farm Management
- Land Economics/Use
- Production Economics

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1. Introduction

While the benefits from adopting more sustainable food production practices accrue broadly to society as a whole, realizing those benefits requires a host of private decisions by farmers. These decisions are complex, involving a myriad of choices concerning what agricultural output to produce, where, when and how. The environmental and economic outcomes of those decisions are often spatially variable, uncertain and risky, because of the unpredictable influences of weather, pests, diseases, and markets.

This paper is a summary and synthesis of a series of three papers dealing with private incentives for sustainable agricultural outcomes. There is a large and still growing literature on ameliorating the undesirable environmental consequences of agriculture. Much of that literature deals with public policy and regulatory approaches to eco-friendly farming. While we have learned much from that past research and analysis, the premise of this suite of papers is that private decisions are crucial for putting global agriculture on a more sustainable footing. To meet the increasing demand for food as a result of population growth and economic development, agriculture must continue to grow more food. The challenge is to do this while protecting natural resources and the environment and also enabling farmers to make a decent living.

The three reports from which this summary is drawn cover a broad range of inter-related issues. Paper 1 presents important principles and concepts that are broadly relevant to the issue of private incentives for sustainable agriculture. This includes research on the drivers of decisions by farmers to change their farming practices, the farm-level economics of changing farming practices, the links from on-farm changes to broader environmental impacts, mechanisms available to encourage changes in farm management, and some of the challenges in ensuring that change leading to enduring environmental benefits is achieved. Paper 2 applies those principles and concepts to the issue of water pollution caused by agriculture, while paper 3 does the same for sequestration of carbon in agricultural soils, as a tool for combatting climate change. In both cases, the papers outline the relevant farming practices available, evidence about their technical effectiveness at delivering environmental benefits, their economic performance for farmers, existing policy approaches, and potential opportunities for private agribusiness firms such as PepsiCo to help enhance the delivery of public benefits.

2. Key Principles and Concepts

In recent years there have been a number of private initiatives to promote sustainable agriculture. These private initiatives often take the form of corporate social responsibility or sustainability targets set by food retailers, processors and others in the food supply chain. Often, the benefits from these initiatives accrue to consumer-facing retailers by way of improved market shares and profitability arising from selling sustainably branded food products that realize a price premium. Meanwhile, much of the cost of compliance may fall on those earlier in the value chain. Farmers selling segregated, sometimes farm-branded, local and often perishable produce into food supply chains may benefit from enhanced farm-gate prices. But others selling non-segregated bulk produce are more vulnerable, possibly bearing the farm-level costs of compliance with little or no ability to realize a higher price for their product. As a result, depending on the context, it is possible for the farm-level costs of sustainable production to exceed the farm-level benefits, even though the benefits to society more broadly are large.

Understanding Farmers' Decisions to Change Their Practices

The process of farmers adopting a new practice is a process of farmers learning about the performance of the practice, and about how best to implement it. Adoption is often a continuous process and may occur in a gradual or stepwise manner, sometimes ending in only partial adoption (Byerlee and Hesse de Polanco 1986). Landholders often change and modify the practice or technology to adapt it to their own circumstances.

The goals of farmers (and their families) are heterogeneous, and can include the following: (i) material wealth and financial security; (ii) environmental protection and enhancement (beyond that related to personal financial gain); (iii) social approval and acceptance; (iv) personal integrity and high ethical standards; and (v) balance of work and lifestyle. Of course, the first of these goals is particularly important to most farmers

Social and demographic factors that influence the adoption of new farming practices include the following.

i. The existence and strength of farmers' social networks and local organizations (e.g., Sobels et al. 2001)
ii. The physical proximity of other adopters (e.g., Ruttan 1996)
iii. A history of respectful relationships between landholders and advocates for the innovation, including scientists, extension agents, other landholders, and private companies (e.g., Marshall 2004)
iv. Access to and reliance on off-property income (Kebede 1992)
v. Property size – larger areas increase the overall benefits of adoption of beneficial innovations and so increase the likelihood of adoption
vi. Education level of the farmer (e.g., Feder et al. 1985), although the evidence is somewhat mixed (e.g., Marsh et al. 2006).

The adoption of a particular practice depends on its "relative advantage", meaning "the degree to which an innovation is perceived as being better than the idea [or practice] it supersedes" (Rogers 2003, p. 229). Relative advantage depends on a range of economic, social and environmental factors, including:

i. The short-term input costs, yields and output prices of the innovation or of other activities that it affects
ii. The innovation's impact on profits in the medium-to-long term
iii. Adjustment costs involved in adopting the innovation
iv. The innovation's impacts on the riskiness of production
v. The innovation's complexity
vi. Government policies
vii. The profitability of the prior practice that the innovation would replace
viii. The perceived environmental credibility of the practice.

Any population of farmers is heterogeneous. This makes it unlikely that any practice will be adopted universally. It also means that there may be benefits from targeting an initiative to a sub-set of farmers who are most likely to adopt the new practice at large scale (Morrison et al. 2012).

Farm-level Economics of Practice Change

Figure 1 illustrates a potential relationship between the proportion of adoption (e.g., the proportion of land in a region on which the practice is used) and the marginal cost of adopting the practice. In this figure, the land is ranked (left to right) from lowest to highest marginal cost (to the farmer) of adoption. This graph illustrates a situation where there is a small negative cost of adoption (i.e., a small private benefit) on about 25 percent of the land, and an increasingly large cost of adoption to the farmer (or farmers) as the level of adoption rises towards 100 percent.

[Figure 1: *The Private Costs of Adoption*]

This broad pattern is likely to be realistic for some environmental practices. In such cases, adoption at low percentages (up to 25 percent in this figure) generates some private benefit for the farmers, but the level of benefits is low so that they provide little incentive to encourage uptake. The economics may not be a barrier to adoption at these levels, and if so it might be feasible for an initiative to generate some additional change. If the farmers in this example are somewhat altruistic and are willing to bear some private costs to provide public benefits (e.g., Chouinard et al. 2008), it may be possible to persuade them to adopt the practice on up to say 50 percent of land in the region. Beyond that level of adoption, costs of adoption rise rapidly so additional adoption is unlikely to occur. The general shape of the graph in Figure 1, with increasing marginal costs as adoption increases, would be common for many environmental practices, although the cost intercept and the steepness of the rising marginal costs would likely vary between practices and between regions.

For decisions about the intensity of agricultural input use, it is very common to observe that there is a “flat payoff function.” In other words, either side of the optimal input rate, there is a wide range of input levels that provide very similar levels of profit (or another measure of payoff) (Pannell 2006). The width and flatness of the profit plateau vary, but the presence of a profit plateau is almost universal in economic production models with continuous decision variables.

Figure 2 shows profit as a function of the nitrogen application rate for several soil types in the central wheatbelt of Western Australia, as represented in the whole-farm bioeconomic model, MIDAS (Morrison et al. 1986; Kingwell and Pannell 1987). The fertilizer range that provides profit within 5 percent of the optimum is +77 to −51 percent of the optimal fertilizer rate for sandy loam over clay—i.e., any rate between 24 and 88 kg/ha of N (nitrogen) gives almost the same profit. Equivalent ranges for the other soils are +75 to −46 percent for shallow sandy loam over clay, and +55 to −42 percent for deep yellow sand. A flat payoff function like this means that it would be possible for farmers to reduce their fertilizer rates below the economically optional rates without suffering a substantial reduction in profits. Results broadly similar to this are typical almost everywhere that nitrogen fertilizer is applied (see also Chai et al. 2019).

[Figure 2: *Nitrogen Response Functions*]

3. Improving Water Quality

Improving water quality is consistently ranked as a top environmental concern in public opinion surveys across most OECD countries (OECD, 2012). Over decades, policy actions and major investments in OECD countries have helped to drastically reduce water pollution from urban centers, industry and sewage treatment works. Progress in reducing agricultural water pollution has been more challenging, in part because it principally originates from farms spread across the landscape (diffuse-source pollution), as opposed to more spatially confined sources, such as urban centers, factories and sewage treatment works (point-source pollution).

The main water pollutants from agriculture are nutrients, soil sediments, and pesticides. Our main focus in this report is on nutrient pollution. The key nutrients causing problems in receiving water bodies are nitrogen and phosphorus (Carpenter et al. 1998; Johnson et al. 1997; Randall et al. 1997). The main source of these nutrients is inorganic fertilizer, but animal manure can also contribute to nutrient pollution in some cases (Alexander et al. 2008).

Nitrogen is relatively mobile in soil water, and often is leached below the root zones of crops and pastures, resulting in it accumulating in groundwater or being discharged into streams. Nitrogen can also reach streams via surface run-off or drains. Phosphorus is less mobile because it tends to bind onto soil particles (Holtan et al. 1988). For that reason, much of the phosphorus that moves from agricultural fields into water bodies does so attached to sediment that has been made mobile in surface water due to soil erosion (Muukkonen et al. 2009; Farkas et al. 2013).

Water pollution causes a range of impacts, including harm to aquatic ecosystems; damage to commercial freshwater and marine fisheries; losses or treatment costs to farms and other industries that use the water; reduction of social values associated with water systems, such as recreation and aesthetics; and impacts on human health, usually through drinking or bathing in contaminated water (OECD 2012). Nutrients support high growth of algae, which causes deoxygenation of water, leading to fish kills, loss of biodiversity and development of hypoxic zones (zones where oxygen is so depleted that plant and animal life do not persist) (Diaz and Rosenberg 2008; Rabalais et al. 2002). A prominent example is the dead zone in the Gulf of Mexico, which grew to 8,800 square miles in 2017.

Farming Practices that Reduce Water-Quality Impacts

There is a broad range of practices that may help to reduce water pollution from agriculture. This report focuses mainly on practices that have been used and researched in the U.S. Cornbelt.

Fertilizer practices. These can include changing the type of fertilizer, the rate of fertilizer application, the timing of application and the location of the application. Research has shown that some farmers apply fertilizer rates in excess of the rates that would be optimal for their circumstances (e.g., Leslie et al. 2017). Even where recommended rates are used, in most cases rates could be reduced to some degree at very little cost to farmers (in terms of reduced net profit) (Pannell 2006; Chai et al. 2019).

Variable-rate technologies. These are systems for spatially varying input rates (such as for fertilizers) within a field depending on localized conditions. It has been suggested that they could help to reduce the overall application of fertilizers. However, modeling by Watkins et al. (1998) showed no difference in N losses between uniform and variable-rate nitrogen fertilizer strategies in a potato, wheat, barley rotation. Moreover, these technologies are usually not financially highly attractive to farmers, in part as another consequence of flat payoff functions (Pannell 2006). Studies by Paz et al. (1999), Babcock and Pautsch (1998) and Thrikawala et al. (1999) found that VRT had modest benefits in terms of fertilizer savings and yield increases in corn. VRT would need to be cheap to purchase if their benefits are to outweigh their costs, but currently, they remain expensive and somewhat complex to use. The high costs mean that in most cases their financial performance for farmers is negative (Watkins et al. 1998) or at best only marginally positive (Schimmelpfennig 2016). *Zero-till and conservation agriculture.* Conservation agriculture combines zero-till with retention of crop residues, to reduce soil erosion. It can potentially reduce nutrient losses from fields, particularly phosphorus. Its performance at reducing nutrient losses from farmers' fields can be expected to vary depending on soil type, slope, and nutrient type.

Cover crops. "Cover crops are grasses, legumes or forbs planted to provide seasonal soil cover on cropland when the soil would otherwise be bare—i.e., before the crop emerges in spring or after fall harvest" (Minnesota Department of Agriculture 2017). Cover crops can reduce soil N leaching, reduce soil erosion, improve soil quality, and enhance habitat for some animals, but once again their N-runoff reducing impacts are often site- and season-specific, adding to the difficulties of scaling the use of these technologies in a cost-effective fashion.

Nitrification inhibitors. These are chemicals that can be applied to agricultural fields to reduce nitrate leaching and also to reduce emissions of nitrous oxide, a potent greenhouse gas. By reducing losses of N from the soil, they can potentially increase soil fertility, reduce the need for fertilizer and increase crop yields. Although nitrification inhibitors seem promising, they have various complexities and challenges in practice. Notably, the dairy industry in New Zealand tested a type of nitrification inhibitor but voluntarily ceased using it when residues appeared in food products.

Land retirement. This takes agricultural land out of production and either restores it to a more natural condition or allows it to regenerate naturally. The USDA has used retirement of cropland as a method for achieving biodiversity and water-conservation goals (Ribaudo et al. 1994).

Buffer strips. Grass buffer strips are uncultivated zones with dense vegetation adjacent to the field or adjacent to streams. Surface runoff flows into the buffer strip and the vegetation removes sediment and nutrients before the water reaches the waterway. Kovacic et al. (2000) found that placement of buffer strips between the wetlands and the river removed an additional 9 percent of the original nitrates. In addition to placement on the landscape, the width of a buffer strip influences its effectiveness in reducing N-runoff. Typically wider strips reduce more N-runoff, although the additional benefit gets smaller as the width becomes greater. Buffer strips come at a significant cost to farmers, including the cost of establishing and maintaining the buffers and the opportunity cost of forgoing production on the buffered land. In many cases, the optimal balance between pollution reduction and cost occurs at a width that delivers less than 100 percent mitigation of pollution (e.g., Sieber et al. 2010; Mtibaa et al. 2018).

Flood-plain restoration (two-stage ditches). In a two-stage ditch, small flood-plains are constructed adjacent to the water channel. During times of high water flow, water rises and covers these floodplains. The wider expanse of water causes water velocity to decrease, providing more time and space for N removal processes such as denitrification, although their overall effect is modest (Roley et al. 2016).

Wetlands. Constructed or restored wetlands can be used to reduce nitrogen concentrations in drainage waters before it is discharged to a larger water body. Kovacic et al. (2000) found that wetlands in Illinois decreased nitrate concentrations in water by around 28 percent.

Bioreactors. These “convert nitrate – the form of nitrogen in farm drainage water – to nitrogen gas” (Christianson 2016). The conversion is done by bacteria, who feed on carbon sources, such as wood chips or crop residues, and convert nitrate in the process. Farmers install a trench of woodchips (or another carbon source) and direct nitrogen-rich drainage waters into it. Nitrogen pollution in the water reaching a stream is reduced by 15 to 90 percent (Christianson 2016).

There has been an overall increase in the uptake of farm management practices and systems beneficial to water quality, to a large extent encouraged by recent policy changes across many OECD countries. This is mainly because of the effort to decouple farm support from production and the strengthening of agri-environmental programmes with a positive effect on water quality (OECD 2012). Unfortunately, this increased adoption has not yet been translated into improved water quality in most cases, because of time lags in the system (OECD 2012).

In considering efforts to further increase the adoption of these practices, fertilizer practices appear to have the most favorable prospects. They are easy to trial, have low up-front costs, and are not complex. In fact, the average rates of nitrogen fertilizer applied in the United States are already slightly below biologically determined benchmark application rates for corn, cotton, winter wheat and spring wheat (Wade et al. 2015). However, some farmers apply more than recommended, sometimes much more.

> "Nitrogen is applied at more than the benchmark rate on 36 percent of corn acres by an average rate of 39 lbs per acre; on 19 percent of cotton acres by an average rate of 40 lbs per acre; on 22 percent of spring wheat acres by an average rate of 30 lbs per acre; and on 25 percent of winter wheat acres by an average rate of 24 lbs per acre." (Wade et al. 2015, p. 19)

Wade et al. (2015) defined four criteria relevant to the environmental impacts of nitrogen (no fall application of nitrogen, at least some post-plant application of nitrogen, nitrogen application rates no larger than the benchmark rate, and nitrogen either injected or incorporated below the soil surface). They found that the percentages of farmers meeting all four of these criteria and also using no-till were very low: 1 percent for cotton, 2 percent for corn, zero for spring wheat, and 4 percent for winter wheat.

Farm-Level Economics of Practices to Improve Water Quality

Farmers are not motivated solely by profit (Chouinard et al. 2008) but the profitability or costliness of a farming practice is a key factor influencing its uptake by farmers (Pannell et al. 2006). An initiative might promote a practice because of its benefits for water pollution, for example, but farmers will need to weigh up issues related to farming logistics (e.g., sowing time), weed and crop management, soil moisture, and so on. For example, cover crops have multiple impacts. As well as reducing erosion and the surface run-off of water, they can fix nitrogen that becomes available to subsequent crops, they may help to suppress weeds, they make break cycles of crop diseases, and they may serve as trap crops for pests. They may influence other aspects of crop management, including rotation selection, fertilizer rates, and weed control methods. All of these things influence the economics of using a cover crop. Although it is possible to discern some general trends, the farm-level economics of specific practices are highly variable and case-specific. Published estimates of the costs and benefits to farmers are helpful but are only indicative, and may not be appropriate for analysis at scale given the site- and season-specific nature of the estimates. .

Fertilizer practices. As noted earlier, the typical shape of the relationship between fertilizer rates and profits is such that changing the rate over quite a wide range makes little difference to farm profit. This means that there is an opportunity for farmers who are willing to make a small financial sacrifice in order to benefit the environment can make a disproportionately large contribution: their percentage reduction in fertilizer use and costs would be much more than their percentage reduction in profit. In addition, a number of farmers apply more fertilizer than the rates that would maximize their profits (around 30 percent of crop farmers in the United States – Wade et al. 2015). In these cases, reducing fertilizer rates would, on average, provide a win-win outcome: more profit for farmers and better environmental outcomes.

Relatively high-cost options. Some of the practices outlined earlier would make worthwhile contributions to reducing water pollution, but would do so at a relatively high cost for many farmers. Recognizing that the economic costs and benefits are heterogeneous, the practices that would tend to fall into this category include cover crops, land retirement, buffer strips, flood-plain restoration, wetlands, and bioreactors.

Less effective options. Some of the practices are not unattractive to many farmers from a financial perspective (neutral, slightly positive or somewhat positive), but make only minor contributions to reducing water pollution. Based on the evidence available to date, practices that tend to fall into this category include variable-rate technologies, zero-till and conservation agriculture. Although these are likely to make only a small difference to water pollution on a per acre basis, they may be applied over very large areas and so make a worthwhile contribution in aggregate.

Watershed-Level Economics of Practices to Improve Water Quality

Some studies have integrated biophysical and socio-economic aspects at the watershed (or water catchment) scale to analyze strategies at that scale or the spatial allocation of strategies within the watershed. Noteworthy for the United States is the work of Cathy Kling and her collaborators. It focuses on identifying economically optimal usage of nutrient mitigation actions in the Upper Mississippi River Basin using an economic model informed by biophysical simulation models (Kling et al. 2006; Kling et al. 2014). They estimate the cost of each management practice considered in each of 119 sub-watersheds, and the reduction in pollutants for each practice in each sub-watershed. Pollutants considered are sediment, nitrogen (in two forms) and phosphorus (in two forms). Their work highlights the great difficulty of achieving ambitious pollution reduction targets. They concluded that, even if comprehensive packages of mitigation actions are taken up broadly by farmers, the percentage reduction in nitrogen losses in the Upper Mississippi River Basin would be around half the target set by the Committee on Water Implications of Biofuels Production in the United States (2008).

Gourevitch et al. (2018) took a different approach, building the social cost of nitrogen application into the function that relates nitrogen rate to net returns from corn production. The analysis was repeated for each county in Minnesota. The social costs considered were groundwater nitrate (NO_3^-) contamination, air pollution by small particulate matter (PM2.5) formed from ammonia (NH_3) and N oxides (NO_x), and global climate change from nitrous oxide (N2O) emissions. They examined the effect of these pollution issues on the optimal nitrogen rate. Figure 3 shows results for a pollution cost of $0.50 per kg of nitrogen applied (a “mid-range” estimate).

[Figure 3. *Private net returns and public net returns for nitrogen fertilizer*]

Factoring in the $0.50 pollution cost reduced the optimal N rate from 165 to 137 kg per ha. Notably, the reduction in private net returns that results from this reduction of N rate is small, at around $6 per hectare (equivalent to just 0.3 percent of the net revenue). A 26 percent reduction in N rate would result in a loss of only 1 percent of net revenue. This would be optimal for a pollution cost of just under $1.00 per kg of N.

Strategies Used by Governments to Promote Water-Quality-Improving Practices

Billions of dollars are spent around the world in public programs to improve water quality. OECD (2012) made a number of recommendations to improve water-quality outcomes from policy efforts, including the following:

- Remove perverse incentives. In many cases, subsidies provided to farmers create an incentive to increase agricultural input usage, worsening water pollution problems
- Properly enforce existing regulations
- Improve the spatial targeting of policies to areas where water pollution is most acute
- Use economic analysis to assess policy options
- Establish improved information systems to support farmers, water managers, and policymakers.

A variety of mechanisms and approaches are used to try to limit water pollution, including the following:

- economic incentives (taxes and subsidies)
- environmental regulations (specific rules backed by penalties)
- farm advice and education (information provision), sometimes combined with measures to build farmer capacity or social capital
- economic instruments, such as water-quality trading schemes, and
- voluntary standards, sometimes associated with accredited standards (like organic farming).

Policies vary from country to country. Shortle and Uetake (2015) provide an overview of agri-environmental policies relevant to water quality in the United States (Table 1).

[Table 1: *Policies used to address water quality*]

4. Soil Carbon Sequestration

Conversion of land to agriculture has decreased soil carbon by about 40 to 60 percent compared to pre-agriculture levels (Sanderman et al. 2010). Globally, this loss of soil carbon (C) has resulted in at least 150 billion tonnes of carbon dioxide being emitted to the atmosphere (compared with about 10 billion tonnes per year from fossil fuels).

Given the large areas of land used for agriculture, even modest increases in soil C levels would make worthwhile contributions to offsetting emissions. However, currently, there is much uncertainty and debate as to the potential of agricultural soils to store additional C, the rate at which it can be done, the 'permanence' of the stored C, the economics of measures to increase soil C, the best policy approaches to the issue, and how best to measure changes in soil C.

When a sequestering practice is adopted, carbon storage typically increases, but at a diminishing rate through time until it plateaus at a new steady-state equilibrium (Figure 4) (Gramig 2012; West et al. 2004). Consequently, only a finite amount of sequestration is possible on any piece of land. Furthermore, this finite opportunity can only be exploited once and is reversible (Figure 4). To retain stored carbon, the sequestering practice must be continued; reverting to the previous practice re-emits the carbon (Figure 4). For these and other reasons, sequestration creates some particular challenges for initiatives and programs, which we will explore later.

[Figure 4: *Stylized dynamics of carbon sequestration*]

The geographic focus here is mainly on broadacre agriculture in developed countries, such as the United States, Canada, and Australia, but many of the issues raised are more broadly relevant to agricultural systems around the world.

Farming practices for sequestering carbon

This report focuses mainly on practices that have been used and researched in the U.S. Cornbelt, but it includes some practices that are more relevant to other farming systems.

Increasing the growth of agricultural plants. An increase in plant growth will result in an increase in the amount of carbon captured by photosynthesis. This could come about through the planting of a more productive plant species (e.g., Subak 2000), or by boosting the growth of existing species by fertilization or irrigation. In the case of the latter, this may increase both the amount of product removed (harvested) and the amount of unremoved residue kept on site and returned to the soil.

Retaining unharvested biomass within the field. Retention of crop residues within the field, rather than removing, grazing or burning them, can contribute to raising soil carbon levels (e.g., Liu et al*.* 2014). Residue retention can have agronomic benefits through reduced evaporation and erosion (Incerti et al*.* 1993). On the other hand, in some cases, removal or burning of crop residues can improve weed control and delay herbicide resistance (Walsh et al. 2013; Lyon et al. 2016), and removed crop residues can have other uses (e.g., as animal fodder or as a fuel source).

Replacing crops with pasture. Including pasture phases in cropping rotations, or increasing the duration of pasture phases, can sequester carbon (e.g., Chan et al. 2011). This is because pastures generally return more carbon to the soil. Depending on the tillage practices used in the cropping, it may also result in reduced cultivation. There is, however, a drawback: pasture is usually grazed by ruminant livestock that emit methane, a much more potent greenhouse gas than CO_2.

Replacing annual pastures with perennial pastures. Sequestration in pastures is greatest if the pasture species is perennial (a plant that lives for several years, rather than re-growing annually). As well as having a longer growing season and thus greater assimilation of carbon through photosynthesis, perennial species tend to allocate a greater proportion of photosynthate below ground. Given current technologies, the viability of substituting annuals with perennials is likely to be greater in pastoral rather than cropping systems.

Cover crops. Cover crops are planted to provide soil cover before the crop emerges in spring or after fall harvest (Minnesota Department of Agriculture 2017). Using cover crops to effectively extend the area cropped per year (without extending the area in crops) can increase the assimilation of carbon via photosynthesis. This can increase contributions of organic matter to the soil, and the water buffering potential of cover crops can also reduce soil erosion which would otherwise result in carbon losses.

Importing organic matter to the field. Applications of organic materials such as compost or manure to soil represent a direct input of carbon (Lal 2004). Adding organic material sourced from offsite does not necessarily result in *net* additional sequestration though, because in many cases this material would otherwise be returned back to the soil anyway, just at another location (e.g., Powlson et al. 2011). Indeed, because of this effect, manure application is not deemed a GHG mitigating practice in some countries, including Canada and France (Minasny et al. 2017).

Applying biochar. Biochar is created by the pyrolysis of feedstock biomass (typically straw, woody materials or manure) to produce carbon-rich biochar, which is similar to charcoal. There has been interest in applying biochar to agricultural soils. Compared to the unprocessed feedstock biomass, biochar is relatively stable and inert (Krull et al. 2009). Despite academic interest (Sanroman et al. 2017), widespread commercial adoption of biochar application by farmers is yet to occur, in part because fertility benefits of biochar have been found to be highly variable (Bach et al. 2016).

Zero-till, reduced tillage. Cultivation releases carbon stored in the soil, primarily through changes in soil structure that enhance the degradation of soil organic matter, and also potentially, by increasing erosion rates (Sanderman et al. 2010). Hence the adoption of zero-till or reduced tillage can reduce the return soil carbon back to the atmosphere, and thus increase soil carbon levels. It is the most commonly discussed way of sequestering carbon in agriculture (e.g., West and Marland 2002; Antle et al. 2007; Lal 2015) but its efficacy as a means of sequestering carbon has been called into question.

Land Retirement or afforestation. Retirement of agricultural land to a more natural vegetated state can increase carbon sequestration in various ways. Less carbon is removed in harvested material, meaning more carbon is available to enter the soil, and there can also be an end to actions that destabilize soil organic matter, like tillage. Retired land is less likely to be bared and thus exposed to erosion. Land retirement can also reduce emissions of the other greenhouse gases (N_2O and CH_4) associated with agricultural production.

Farm-level economics of sequestering farm practices

"The economic potential to sequester carbon is much lower than the technical potential reported in soil science studies" (Lewandrowski et al. 2004, p.i).

Just like the practices for reducing water pollution discussed earlier, the farm-level economic performance of practices that sequester soil carbon is a key driver of their adoption and is heterogeneous between farmers and locations.

Economically attractive options that are already widely adopted. There are limited opportunities for worthwhile initiatives with these options because they are already standard practice for many farmers. They include increasing the growth of agricultural plants; retaining unharvested biomass within the field; and zero-till, reduced tillage. In regions where their adoption is moderate (say 20 to 60 percent), rather than being close to 100 percent, there could be opportunities to influence farmers who have not yet adopted but would benefit from doing so, or who would be willing to bear some net costs to generate environmental public benefits.

Options that are technically effective but costly in many or most situations. Options in this category are land retirement or afforestation and applying biochar. Biochar may be cost-effective for farmers in certain situations, but high uncertainty about its effects on crop yields creates a risk that is unattractive to many farmers.

Options with zero or low technical effectiveness. This includes the three options that are already widely adopted—increasing the growth of agricultural plants; retaining unharvested biomass within the field; and zero-till, reduced tillage—plus cover crops. Present cover crop options have the added disadvantage of low economic attractiveness to most farmers, at least in the U.S. context, reflected in the fact that it is adopted by only around 2 percent of farmers.

Options with other disadvantages. Even in situations where these options are economically attractive to farmers, they would be unsuitable for promotion for other reasons. One such option is replacing crops with pastures. The problem is that pastures are generally used to run ruminant livestock, which emit high levels of methane, a potent greenhouse gas. A second case is retiring cropland from production, replacing it with forest or natural vegetation. The problem this time is evidence of high levels of "indirect leakage"; the reduction in the supply of grain causes an increase in global grain prices or creates market opportunities, resulting in increases in crop production and associated emissions in other locations. Thirdly, if biomass is imported to a field, it may increase carbon sequestration locally, but potentially at the cost of reduced carbon sequestration elsewhere. The carbon is just transferred from one location to another.

Aside from removing greenhouse gases from the atmosphere, increasing soil C content can also contribute to other improvements, including: reduced soil erosion, improved nutrient retention, improved pH buffering, improved soil structure, increased water infiltration and increased water-holding capacity (Incerti et al*.* 1993; Sanderman et al*.* 2010; Meyer et al*.* 2015; Murphy 2015; Petersen and Hoyle 2016). If these changes provided large enough benefits for farmers, they could help to encourage adoption of carbon sequestering practices. However, placing an economic value on these agronomic benefits is challenging and has seldom been attempted. An exception is Petersen and Hoyle (2016) who estimated agronomic benefits of around AUD$1 to 2/ha/year per tonne of soil C in the wheatbelt of Western Australia – probably too small to motivate most management changes.

Strategies Used by Governments to Promote Sequestration

In most existing policies and schemes to promote sequestration, landholder participation is voluntary, with participation incentivized financially, either through direct payments or through the provision of carbon credits or offsets that can be on-sold to emitters (Thamo and Pannell 2016). In some cases, governments directly purchase the credits (e.g., Australia's Emission Reduction Fund). In other cases, polluters can buy the sequestration credits in lieu of paying a carbon tax (as was possible under Australia's now-defunct Carbon Farming Initiative and carbon tax) or to allow them to operate within an emissions cap (e.g., the Specified Greenhouse Gas Emitters Regulation in Alberta, Canada).

Even in the absence of mandated requirements to offset emissions, polluters can buy sequestration credits to voluntarily offset their emissions (e.g., for marketing purposes). As of 2017, voluntary carbon projects aimed at offsetting emissions were underway in 83 countries (Hamrick and Gallant 2018). Agriculture, forestry and land-management projects constitute around 13 percent of these projects by number and 23 percent by volume of offsets in CO_2 equivalents (Hamrick and Gallant 2018). Unfortunately, the offsets polluters use to voluntarily reduce emissions tend to be regarded as lower quality than offsets supplied to mandatory schemes, and they usually trade at much lower prices.

5. Challenges in ensuring that apparent environmental benefits are real

There are several well-recognized problems that are likely to affect agri-environmental initiatives and may require management.

Additionality

This is relevant in cases where farmers are rewarded in some way for providing public environmental benefits. The principle is that farmers should not be rewarded for undertaking actions that they were going to undertake even without the rewards. Doing so uses up scarce resources that could be used more productively to promote practice change that is actually additional (Claassen et al. 2014).

Assessing additionality is widely recognized as an important requirement (and a considerable challenge) in relation to sequestration of carbon in soils (Thamo and Pannell 2016). However, it is just as relevant to reducing water pollution and to other types of environmental benefits where a financial reward is used to encourage adoption of the practice.

In Australia, zero-till is routinely used by most farmers (Llewellyn et al. 2012). Any program offering payments to farmers who adopted zero-till would generate very little additional adoption. In the United States, Wade et al. (2015) reported that just under 40 percent of the area of four major crops was sown using zero-till or strip-till in 2010-11. In this case, the prospects of a program generating addition adoption are greater, but care is required.

Leakage

In some cases, an action that is intended to mitigate a pollutant can inadvertently result in other emissions that partly or fully offset the original gains. This has particularly been noted as a problem with carbon sequestration. For example, Montserrat and Sohngen (2009) estimated that in some voluntary offset schemes, up to 90 percent of the claimed emissions 'savings' may have been shifted or 'leaked' to another location. Thamo and Pannell (2016) argued that direct leakage (in the form of emissions of methane from ruminant livestock) could exceed the original reduction in emissions resulting from a switch from cropping to pasture.

Permanence

As CO_2 emissions reside in the atmosphere for 300+ years (Archer 2005), if a unit of carbon sequestration is to fully offset a unit of emissions, the sequestration needs to be permanent. In programs where farmers receive payment for sequestering carbon, this need has been recognized in two ways: (a) by imposing a condition that sequestration activities must be maintained for a specified time frame, and (b) discounting the payments offered to farmers for shorter time frames (Thamo and Pannell 2016). Understandably, farmers are reluctant to commit land to be permanently devoted to a particular activity, in case the economics of agriculture change in such a way that other production activities become far more attractive.

Transaction costs

A common theme in the literature is that satisfying requirements for additionality, non-leakage and permanence increases transaction costs, particularly the costs of monitoring and measuring. For example, ensuring permanency requires on-going monitoring of land management, to ensure that the sequestering activity is being continued. This monitoring is relatively difficult for soil carbon given that some of the methods for increasing soil carbon are not easily detected by cheap monitoring methods, like remote sensing. This creates a dilemma. The transaction costs of monitoring and measuring to accurately assessing additionality, leakage and permanence are likely to be so high that they counteract much of the benefit of the program and they discourage participation. On the other hand, simplifications that sufficiently avoid most of the transaction costs are likely to leave the sequestration program ineffective because of lack of additionality, excessive leakage or impermanence.

Time lags

Time lags between promoting a more sustainable practice and delivering the ultimate environmental improvements are often much longer than commonly appreciated. There are various components to these time lags, including the time lag until adoption (typically 10 to 20 years), and the time lag until the practice has the desired effect. If a new technology or crop management approach is required, the research lag involved can also be quite considerable, typically decades, not years (Pardey and Beddow 2013).

While changes to surface hydrology can occur very rapidly, changes to groundwater hydrology may take decades or even a century to reach a new equilibrium (Meals and Dressing 2008). Similarly, it typically takes decades (e.g., 30-50 years) for a new equilibrium level of soil carbon to be established after a change in management (West et al. 2004). It may be possible to identify and target changes that would deliver environmental benefits with relatively short time lags.

6. Potential Opportunities for PepsiCo to Contribute

The set of opportunities available to PepsiCo differs from the set used by governments. Some of the options available to governments are not available (e.g., regulatory constraints) while PepsiCo has some options not available to governments. Opportunities are considered here in five areas: targeting, informing and persuading, empowering, coordinating and incentivizing.

Targeting

A targeted approach to abatement of water pollution (or potentially carbon sequestration if suitable options were available) is more cost-effecting than untargeted broad-brush approaches. Targeting could include identifying those countries, regions, sub-watersheds, farms/farmers, fields, crops or production systems for which the opportunities to reduce water pollution are particularly high, either because the public environmental benefits of adopting new sustainable practices are particularly high or because the costs of doing so are particularly low. Similarly, targeting could include identifying those countries, regions, farms/farmers, fields, crops or production systems for which the opportunities to sequester carbon in soils are relatively high because the overall costs of doing so are relatively low.

Informing and Persuading

PepsiCo could lead an initiative to collate and analyze information and provide it to farmers, potentially directly, or potentially via other existing extension channels. We know that many farmers are willing to adopt sustainable practices if they are win-win (in terms of their public and private net benefits) and that some can be persuaded to adopt them if they are win-neutral or even win-lose to some extent. The targeting exercise (see above) could contribute in this respect by providing convincing information that particular practices in particular locations are most likely to generate public benefits.

Another innovation that could be brought in here is a set of insights from the relatively new discipline of Behavioural Economics, which has developed a number of insights into how the form of a communication can influence peoples' willingness to comply with its advice or request. PepsiCo could invest in experiments in communication options, informed by Behavioural Economics, to identify approaches that are most effective in stimulating farmers' adoption of sustainable practices.

For soil carbon, the biggest challenge with a strategy based on informing and persuading farmers is the lack of compelling practices that are worth farmers adopting. It appears that all of the available practices have significant challenges or limitations. If sequestration of carbon in soils is adopted as a target, it may be best to start with efforts to enhance the suite of available technical options (see the *Empowering* section below).

Empowering

This approach is about improving the technical options available to farmers. It could involve PepsiCo investing in highly targeted R&D, perhaps in partnerships with other private and public agencies, to improve existing technical options or develop new options, with the objective being to reduce their private costs and/or increase their benefits.

The spatial information from the *Targeting* component (above) and insights about adoption of new practices could be used in an analysis to identify specific contexts where investment in technology development is most likely to lead to adoption of new practices, and where adoption is most likely to deliver environmental benefits.

Coordinating

PepsiCo could use its networks, reputation, and profile to influence and organize other firms and agencies so that efforts to pursue sustainable agricultural outcomes are well aligned and mutually reinforcing. PepsiCo could add value to the analyses of this project by sharing and advocating them with other relevant firms and agencies.

For example, the spatial analysis and insights into adoption could be shared with other firms and agencies with encouragement for them to target their extension efforts in productive ways. Results of the Empowering analysis could be shared with research agencies to encourage them to target their research efforts appropriately. The spatial analysis could be shared with policy agencies to assist them to identify priorities for incentive payments or regulation. This could be supported by advocating for a targeted approach to policy, rather than relatively untargeted approaches which are often used in practice. The strategy might also involve PepsiCo coordinating with other firms or NGOs to jointly fund initiatives, such as the analyses needed to support targeting of effort, or the R&D to develop new technologies.

Incentivising

PepsiCo may be able to contribute to incentivization of farmers by informing and collaborating with public regulatory and funding bodies (part of the *Coordinating* strategy), or by designing their contracts with farmers in innovative ways that provide the required incentives.

Another potential strategy would be to compensate farmers for any economic losses or increased production or economic risks that they suffer as a result of reducing nitrogen fertilizer rates by specified percentages. The compensation could potentially come from private or public sources. Because of flat payoff functions, worthwhile reductions in nutrient pollution would be achievable at low cost. Such a scheme would require careful design to ensure that claimed fertilizer reductions were legitimate.

7. Conclusion

There are likely to be opportunities for private agribusiness firms to influence the decisions of farmers regarding their adoption of sustainable agricultural practices. Such firms have some advantages relative to traditional conservation programs delivered by governments. We see potential in harnessing the private incentives of agribusiness firms to enhance the private incentives of farmers to deliver public environmental benefits.

Nevertheless, there are challenges in delivering a successful initiative in this space. Although there are many different farming practices that have been advocated for addressing water pollution or carbon sequestration, care and good information is needed when selecting which practices to promote. Some practices are costly, risky or inconvenient for farmers. Some are attractive to farmers but are already widely adopted in certain regions or farming systems. Some have other disadvantages, such as causing leakage of CO_2 at other times or in other places, counteracting the initial sequestration. Some are not very effective at reducing water pollution or sequestering carbon.

In this report, we have identified options and strategies that would help PepsiCo to identify approaches that are most likely to deliver genuine improvements in water quality or carbon sequestration. Key principles identified in the analysis include the following.

i. The importance of understanding how well the pollution mitigation practices fit with farmers own needs and preferences. For example, are the practices financially beneficial or costly, are they complex and inconvenient to implement, and are they risky?
ii. The benefits of taking a targeted approach. Analyses to select particular locations, farming types and pollution mitigation practices can potentially improve the impacts substantially.
iii. The need to look beyond superficial assessments of apparent environmental benefits. In considering initiatives, it is important to consider the risks of non-additionality, leakage, and non-permanence. Realistic time lags until the delivery of benefits should be appreciated.

Of the many options and ideas discussed in these reports, those that seem most likely to be worth further consideration are as follows.

i. Targeting farmers who apply fertilizer rates that are above-recommended rates – around 30 percent of U.S. crop farmers.
ii. Exploiting "flat payoff functions" for fertilizer, which mean that farmers can reduce fertilizer rates below economically optimal rates and bear only a very minor economic sacrifice. We saw an example where a 26 percent reduction in nitrogen rate would result in a loss of only 1 percent of net revenue. One potential strategy is to make good any economic losses that farmers suffer as a result of reducing nitrogen fertilizer rates by specified percentages (i.e., pay them the difference). Because of flat payoff functions, the losses, on average, will likely be small for moderate rate reductions.
iii. Look for low-cost opportunities to encourage zero-till. This means targeting efforts to areas with moderate existing adoption and aiming to influence farmers who are on the borderline of adoption.
iv. Collate and analyze information and provide it to farmers, potentially directly, or potentially via other existing extension channels. This could relate to over-fertilizing, flat payoff functions, or zero-till, for example.

v. Invest in experiments in communication options, informed by Behavioural Economics, to identify approaches that are most effective in stimulating farmers' adoption of sustainable practices.

vi. Invest in highly targeted R&D to improve existing technical options or develop new options, with the objective being to reduce their private costs or increase their benefits.

vii. Coordinate with other firms or Non-Government Organisations to jointly fund initiatives, such as the analyses needed to support targeting of effort, or R&D to develop new technologies.

viii. Include conditions in contracts with farmers to mandate particular sustainable practices, such as reduced fertilizer rates. Investigate the acceptability of such conditions for different practices in different situations, along with the design details of these contracts to ensure they do not undercut farmers' profitability over the longer run.

References

Alexander, R.B., Smith, R.A., Schwarz, G.E., Boyer, E.W., Nolan, J.V. and Brakebill, J.W. (2008). Differences in phosphorus and nitrogen delivery to the Gulf of Mexico from the Mississippi River Basin. *Environmental Science and Technology* 42(3), 822-830.

Antle, J.M., Capalbo, S.M., Paustian, K. and Ali, M. (2007). Estimating the economic potential for agricultural soil carbon sequestration in the Central United States using an aggregate econometric-process simulation model. *Climatic Change* 80, 145-171.

Archer, D. (2005). Fate of fossil fuel CO_2 in geologic time. *Journal of Geophysical Research* 110, C09S05.

Babcock, B.A. and Pautsch, G.R. (1998). Moving from uniform to variable fertilizer rates on Iowa corn: Effects on rates and returns. *Journal of Agricultural and Resource Economics* 23(2), 385-400.

Bach, M., Wilske, B. and Breuer, L. (2016). Current economic obstacles to biochar use in agriculture and climate change mitigation. *Carbon Management* 7, 183-190.

Butt, T.A. and McCarl, B.A. (2004). On-farm carbon sequestration: can farmers employ it to make some money. *Choices* 19, 27-32.

Byerlee, D. and Hesse de Polanco, E. (1986). Farmers' stepwise adoption of technological packages: evidence from the Mexican Altiplano. *American Journal of Agricultural Economics* 68, 519-527.

Carpenter, S.R., Caraco, N.F., Correll, D.L., Howarth, R.W., Sharpley, A.N. and Smith, V.H. (1998). Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications* 8(3), 559-568.

Chai, Y., Pannell, D.J., Pardey, P.G., and Hurley, T.M. (2019). Harnessing flat payoff functions for nitrogen fertilizer to address water pollution. Contributed Paper to 63rd annual conference of the Australasian Association of Agricultural and Resource Economists, Melbourne, Australia (in preparation).

Chan, K.Y., Conyers, M.K., Li, G.D., Helyar, K.R., Poile, G., Oates, A. and Barchia, I.M. (2011). Soil carbon dynamics under different cropping and pasture management in temperate Australia: Results of three long-term experiments. *Soil Research* 49, 320-328.

Chouinard, H.H., Paterson, T., Wandschneider, P.R., and Ohler, A.M. (2008). Will farmers trade profits for stewardship? Heterogeneous motivations for farm practice selection. *Land Economics* 84(1), 66-82.

Christianson, L. (2016). Reducing water pollution with microbes and wood chips. *The Conversation*, https://theconversation.com/reducing-water-pollution-with-microbes-and-wood-chips-58852 (accessed 28 January 2018).

Claassen, R., Horowitz, J., Duquette, E., and Ueda, K. (2014). Additionality in U.S. agricultural conservation and regulatory offset programs (USDA Economic Research Report No. 170). Washington, DC: United States Department of Agriculture.

Committee on Water Implications of Biofuels Production in the United States (2008). Water Implications of Biofuels Production in the United States, National Academies Press, Washington D.C. https://www.nap.edu/read/12039/chapter/6 (accessed 7 December 2018).

Deloitte Access Economics (2013). Economic Contribution of the Great Barrier Reef, Great Barrier Reef Marine Park Authority, Townsville.

Diaz, R.J. and Rosenberg, R. (2008). Spreading dead zones and consequences for marine ecosystems. *Science* 321(5891), 926-929.

Farkas, C., Beldring, S., Bechmann, M. and Deelstra, J. (2013). Soil erosion and phosphorus losses under variable land use as simulated by the INCA-P model, *Soil Use and Management* 29(s1), 124-137.

Ferguson, R.B. (2015). Groundwater quality and nitrogen use efficiency in Nebraska's Central Platte River Valley. *Journal of Environmental Quality* 44, 449-459.

Gourevitch, J.D., Keeler, B.L. and Ricketts, T.H. (2018). Determining socially optimal rates of nitrogen fertilizer application. *Agriculture, Ecosystems and Environment* 254, 292-299.

Gramig, B.M. (2012). Some unaddressed issues in proposed cap-and-trade legislation involving agricultural soil carbon sequestration. *American Journal of Agricultural Economics* 94, 360-367.

Hamrick, K. and Gallant, M. (2018). Voluntary Carbon Markets Outlooks and Trends, January to March 2018, Forest Trends, Washington D.C.

Holtan, H., Kamp-Nielsen, L. and Stuanes, A.O. (1988). Phosphorus in soil, water and sediment: an overview. *Hydrobiologia* 170, 19-34.

Incerti, M., Sale, P.W.G. and O'Leary, G.J. (1993). Cropping practices in the Victorian Mallee. 1. Effect of direct drilling and stubble retention on the water economy and yield of wheat. *Australian Journal of Experimental Agriculture* 33, 877-883.

Johnson, L.B., Richards, C., Host, G.E., and Arthur, J.W. (1997). Landscape influences on water chemistry in Midwestern stream ecosystems. *Freshwater Biology*, 37(1), 193-208.

Kebede, Y. (1992). Risk behavior and new agricultural technologies: The case of producers in the central highlands of Ethiopia. *Quarterly Journal of International Agriculture* 31, 269-284.

Kingwell R.S. and Pannell D.J. (Eds) (1987). *MIDAS, A Bioeconomic Model of a Dryland Farm System*, Pudoc, Wageningen, 207pp.

Kling, C.L., Secchi, S., Jha, M., Feng, H., Gassman, P.W. and Kurkalova, L. A. (2006). Upper Mississippi River Basin modeling system part 3: Conservation practice scenario results. In Coastal hydrology and processes, ed. V. P. Singh, and Y. J. Xu. Highland Ranch, CO: Water Resources Publications.

Kling, C.L., Panagopoulos, Y., Rabotyagov, S., Valcu, A., Gassman, P.W., Campbell, T., White, M., Arnold, J.G., Srinivasan, R., Jha, M.K., Richardson, J., Moskal, L.M., Turner, G., and Rabalais, N. (2014). LUMINATE: linking agricultural land use, local water quality and Gulf of Mexico Hypoxia. *European Review of Agricultural Economics* 41(3), 431-459.

Kovacic, D.A., David, M.B., Gentry, L.E., Starks, K.M., and Cooke, R.A. (2000). Effectiveness of constructed wetlands in reducing nitrogen and phosphorus export from agricultural tile drainage. *Journal of Environmental Quality*, 29(4), 1262-1274.

Krull, E., Cowie, A. and Singh, B. (2009). Biochar: How stable is it? And how accurately do we need to know? *1st Asia Pacific Biochar Conference*. NSW Department of Primary Industries, Gold Coast, QLD, pp 27-28.

Lal, R. (2004). Soil carbon sequestration to mitigate climate change. *Geoderma* 123(1-2), 1-22.

Lal, R. (2015). Sequestering carbon and increasing productivity by conservation agriculture. *Journal of Soil and Water Conservation* 70, 55A-62A.

Lawrence, J. (2017). Reducing nutrient loss: science shows what works. Ames: Iowa State University. https://store.extension.iastate.edu/product/13960 (accessed 7 December 2018).

Leslie, J.E., Weersink, A., Yang, W. and Fox, G. (2017). Actual versus environmentally recommended fertilizer application rates: Implications for water quality and policy. *Agriculture, Ecosystems and Environment* 240, 109-120.

Lewandrowski, J., Peters, M., Jones, C., Sperow, M., Eve, M., and Paustian, K. (2004). Economics of Sequestering Carbon in the U.S. Agricultural Sector, United States Department of Agriculture, Economic Research Service, Technical Bulletin Number 1909, Washington D.C.

Liu, D.L., Anwar, M.R., O'Leary, G. and Conyers, M.K. (2014). Managing wheat stubble as an effective approach to sequester soil carbon in a semi-arid environment: Spatial modelling. *Geoderma* 214–215, 50-61.

Llewellyn, R.S., D’Emden, F.H. and Kuehne, G. (2012). Extensive use of no-tillage in grain growing regions of Australia. *Field Crops Research* 132, 204-212.

Lyon, D.J., Huggins, D.R. and Spring, J.F. (2016). Windrow Burning Eliminates Italian Ryegrass (Lolium perenne ssp. multiflorum) Seed Viability. *Weed Technology* 30, 279-283.

Marsh, S.P., Burton, M.P., and Pannell, D.J. (2006) Understanding farmers’ monitoring of water tables for salinity management. *Australian Journal of Experimental Agriculture* 46, 1113-1122.

Marshall, G.R. (2004) From words to deeds: enforcing farmers’ conservation cost-sharing commitments. *Journal of Rural Studies* 20, 157-167.

Meals, D.W. and Dressing, S.A. (2008). Lag time in water quality response to land treatment. Tech Notes 4. Environmental Protection Agency. http://www.epa.gov/sites/production/files/2016-05/documents/tech_notes_4_dec2013_lag.pdf (accessed 23 November 2018)

Meyer, R., Cullen, B.R., Johnson, I.R. and Eckard, R.J. (2015). Process modelling to assess the sequestration and productivity benefits of soil carbon for pasture. *Agriculture, Ecosystems and Environment* 213, 272-280.

Minasny, B., Malone, B.P., McBratney, A.B., Angers, D.A., Arrouays, D., Chambers, A. et al. (2017). Soil carbon 4 per mille. *Geoderma* 292, 59-86.

Minnesota Department of Agriculture (2017). Cover crops, Conservation Practices - Minnesota Conservation Funding Guide, http://www.mda.state.mn.us/protecting/conservation/practices/covercrops.aspx (accessed 28 January 2018)

Montserrat, A., and Sohngen, B. (2009). How big is leakage from forestry carbon credits? Estimates from a global model. *IOP Conference Series: Earth and Environmental Science, 6*(5), 052011.

Morrison D.A., Kingwell R.S., Pannell, D.J. and Ewing, M.A. (1986). A mathematical programming model of a crop-livestock farm system. *Agricultural Systems* 20(4), 243-268.

Morrison, M., Durante, J., Greig, J., Ward, J. and Oczkowski, E. (2012). Segmenting landholders for improving the targeting of natural resource management expenditures. *Journal of Environmental Planning and Management* 55(1), 17–37.

Mtibaa, S., Hotta, N. and Irie, M. (2010). Analysis of the efficacy and cost-effectiveness of best management practices for controlling sediment yield: A case study of the Joumine watershed, Tunisia. *Science of the Total Environment* 616-617, 1-16.

Murphy, B.W. (2015). Impact of soil organic matter on soil properties—a review with emphasis on Australian soils. *Soil Research* 53, 605-635.

Muukkonen, P., Hartikainen, H. and Alakukku, L. (2009). Effect of soil structure disturbance on erosion and phosphorus losses from Finnish clay soil. *Soil and Tillage Research* 103(1), 84-91.

OECD (2012). Water Quality and Agriculture: Meeting the Policy Challenge, Key Messages, Executive Summary, OECD, Paris.

Pannell, D.J. (2006). Flat-earth economics: the far-reaching consequences of flat payoff functions in economic decision making. *Review of Agricultural Economics* 28(4), 553-566.

Pannell, D.J., Marshall, G.R., Barr, N., Curtis, A., Vanclay, F. and Wilkinson, R. (2006). Understanding and promoting adoption of conservation practices by rural landholders. *Australian Journal of Experimental Agriculture* 46(11), 1407-1424.

Pardey, P.G. and J.M. Beddow. *Agricultural Innovation: The United States in a Changing Global Reality*. CCGA Report. Chicago: Chicago Council on Global Affairs, 2013.

Paz J.O., Batchelor, W.D., Babcock, B.A., Colvin, T.S., Logsdon, S.D., Kaspar, T.C. and Karlen, D.L. (1999). Model-based technique to determine variable rate nitrogen for corn. *Agricultural Systems* 61(1), 69-75.

Petersen, E.H. and Hoyle, F.C. (2016). Estimating the economic value of soil organic carbon for grains cropping systems in Western Australia. *Soil Research* 54, 383-396.

Powlson, D.S., Whitmore, A.P. and Goulding, K.W.T. (2011). Soil carbon sequestration to mitigate climate change: a critical re-examination to identify the true and the false. *European Journal of Soil Science* 62, 42-55.

Rabalais, N.N., Turner, R.E. and Wiseman Jr., W.J. (2002). Gulf of Mexico hypoxia, a.k.a. "The dead zone". *Annual Review of Ecology and Systematics* 33, 235-263.

Randall, G.W., Huggins, D.R., Russelle, M.P., Fuchs, D.J., Nelson, W.W., and Anderson, J.L. (1997). Nitrate losses through subsurface tile drainage in conservation reserve program, alfalfa, and row crop systems. *Journal of Environmental Quality* 26(5), 1240-1247.

Ribaudo, M.O., Osborn, C.T. and Konyar, K. (1994). Land retirement as a tool for reducing agricultural nonpoint source pollution. *Land Economics* 70(1), 77-87.

Rogers, E.M. (2003). *Diffusion of Innovations*. 5th ed, New York: Free Press.

Roley, S.S., Tank, J.L., Tyndall, J.C. and Witter, J.D. (2016). How cost-effective are cover crops, wetlands, and two-stage ditches for nitrogen removal in the Mississippi River Basin? *Water Resources and Economics* 15, 43-56.

Ruttan, V.W. (1996). What happened to technology adoption-diffusion research? *Sociologia Ruralis* 36, 51-73.

Sanderman, J., Farquharson, R. and Baldock, J. (2010). *Soil Carbon Sequestration Potential: A review for Australian agriculture.* CSIRO Land and Water, Canberra.

Sanroman, M.A., Lee, D.J., Khanal, S. and Ok, Y.S. (2017). Special Issue on Biochar: Production, Characterization and Applications – Beyond Soil Applications, *Bioresource Technology* 246, 1.

Schimmelpfennig, D. (2016). *Farm profits and adoption of precision agriculture*. Economic Research Report No. 217, Economic Research Service. Washington, D.C.: U.S. Department of Agriculture.

Scientific American (undated). What Causes Ocean "Dead Zones"? https://www.scientificamerican.com/article/ocean-dead-zones/ (accessed 7 December 2018).

Sesmero, J. and Gramig, B. (2013). Farmers' Supply Response, Price of Corn Residue, and Its Economic Viability as an Energy Feedstock. *BioEnergy Research* 6, 797-807.

Shortle, J. and T. Uetake (2015). Public goods and externalities: agri-environmental policy measures in the United States, OECD Food, Agriculture and Fisheries Papers, No. 84, OECD Publishing, Paris. http://dx.doi.org/10.1787/5js08hwhg8mw-en (accessed 4 October 2018).

Sieber, S., Pannell, D.J., Müller, K., Holm-Müller, K., Kreins, P. and Gutsche, V. (2010). Modelling pesticide risk: A marginal cost-benefit analysis of an environmental buffer-zone programme. *Land Use Policy* 27(2), 653-661.

Sobels, J., Curtis, A., and Lockie, S. (2001) The role of Landcare networks in rural Australia: exploring the contribution of social capital. *Journal of Rural Studies* **17**, 265-276.

Subak, S. (2000). Agricultural soil carbon accumulation in North America: considerations for climate policy. *Global Environmental Change* 10, 185-195.

Thamo, T. and Pannell, D.J. (2016). Challenges in developing effective policy for soil carbon sequestration: perspectives on additionality, leakage, and permanence. *Climate Policy* 16, 973-992.

Thrikawala, S., Weersink, A., Kachanoski, G. and Fox, G. (1999). Economic feasibility of variable-rate technology for nitrogen on corn. *American Journal of Agricultural Economics* 81(4), 914-927.

Wade, T., Claassen, R. and Wallander, S. (2015). *Conservation-practice adoption rates vary widely by crop and region,* EIB-147. Washington, D.C.: U.S. Department of Agriculture, Economic Research Service. https://www.ers.usda.gov/webdocs/publications/44027/56332_eib147.pdf (accessed 23 November 2018).

Walsh, M., Newman, P. and Powles, S. (2013). Targeting Weed Seeds In-Crop: A New Weed Control Paradigm for Global Agriculture. *Weed Technology* 27, 431-436.

Watkins, K.B., Lu, Y. and Huang, W. (1998). Economic and environmental feasibility of variable rate nitrogen fertilizer application with carry-over effects. *Journal of Agricultural and Resource Economics* 23(2), 401-426.

West, T.O. and Marland, G. (2002). A synthesis of carbon sequestration, carbon emissions, and net carbon flux in agriculture: comparing tillage practices in the United States. *Agriculture, Ecosystems and Environment* 91, 217-232.

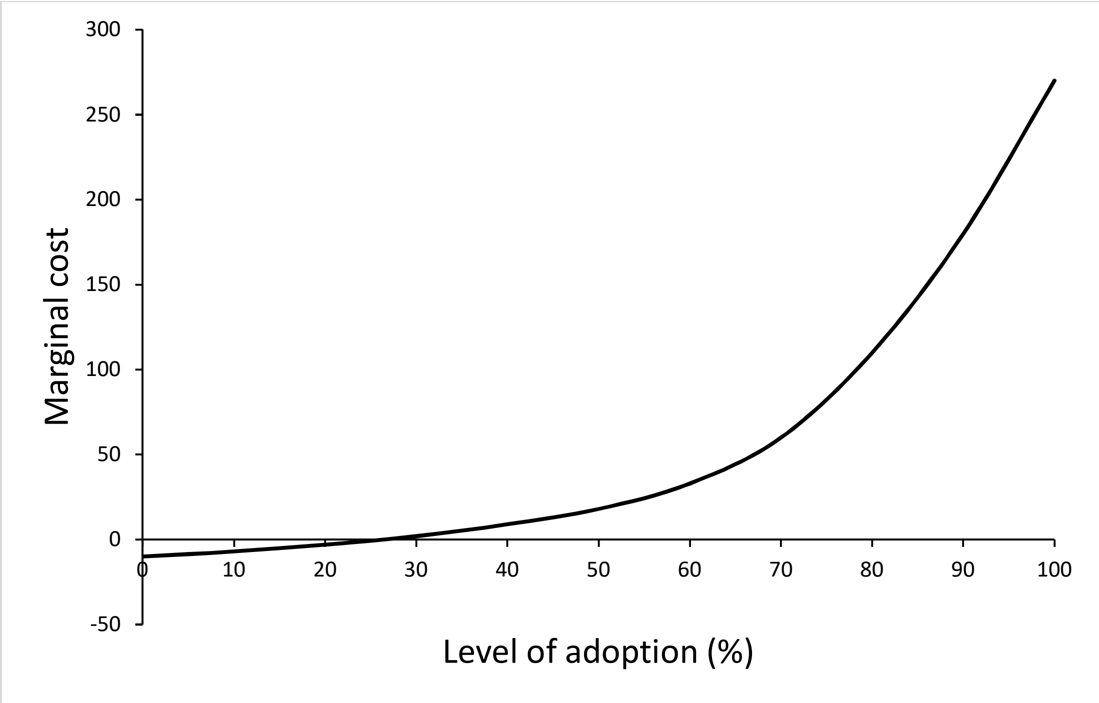
West, T. O., Marland, G., King, A. W., Post, W. M., Jain, A. K., and Andrasko, K. (2004). Carbon management response curves: estimates of temporal soil carbon dynamics. *Environmental Management* 33(4), 507-518.

Table 1. Policies used to address water quality impacts from agriculture in the United States

Instrument type	**U.S. policies related to water quality**
Regulatory requirement	Pesticides (federal); Regulated concentrated animal feeding operations (federal and state); Farming practices (e.g. nutrient management) (some states)
Environmental taxes/charges	Agricultural privilege tax (Florida)
Payments based on farming practices	United States Department of Agriculture (USDA) Environmental Quality Incentives Program (EQIP) (some states)
Payments based on agricultural land retirement	USDA Land Retirement Programs (Conservation Reserve Program, CRP)
Payments based on performance rankings	USDA (Conservation Stewardship Program, CSP)
Tradable rights/permits	Water quality trading (some states)
Facilitative	Various federal, state and local educational programs, federal and state technical assistance programs, federal organic labelling requirements

Source: Adapted from Shortle and Uetake (2015).

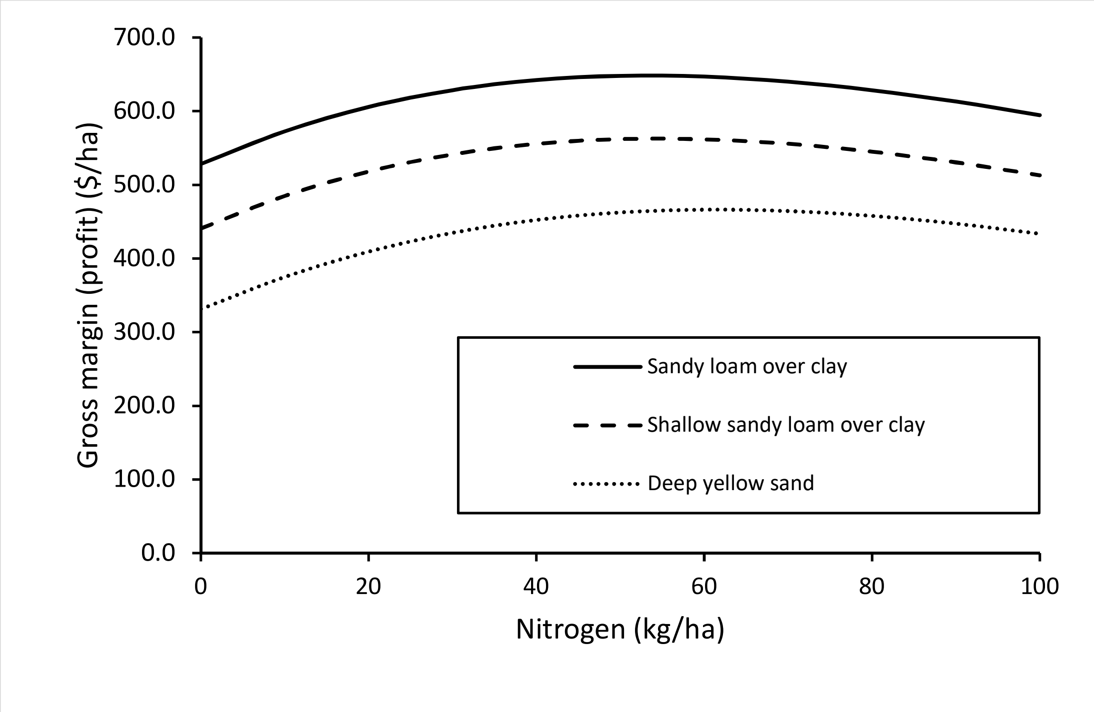
Figure 1. The private marginal cost of adoption by level of adoption (percentage of farming area)



Source: Developed by authors.

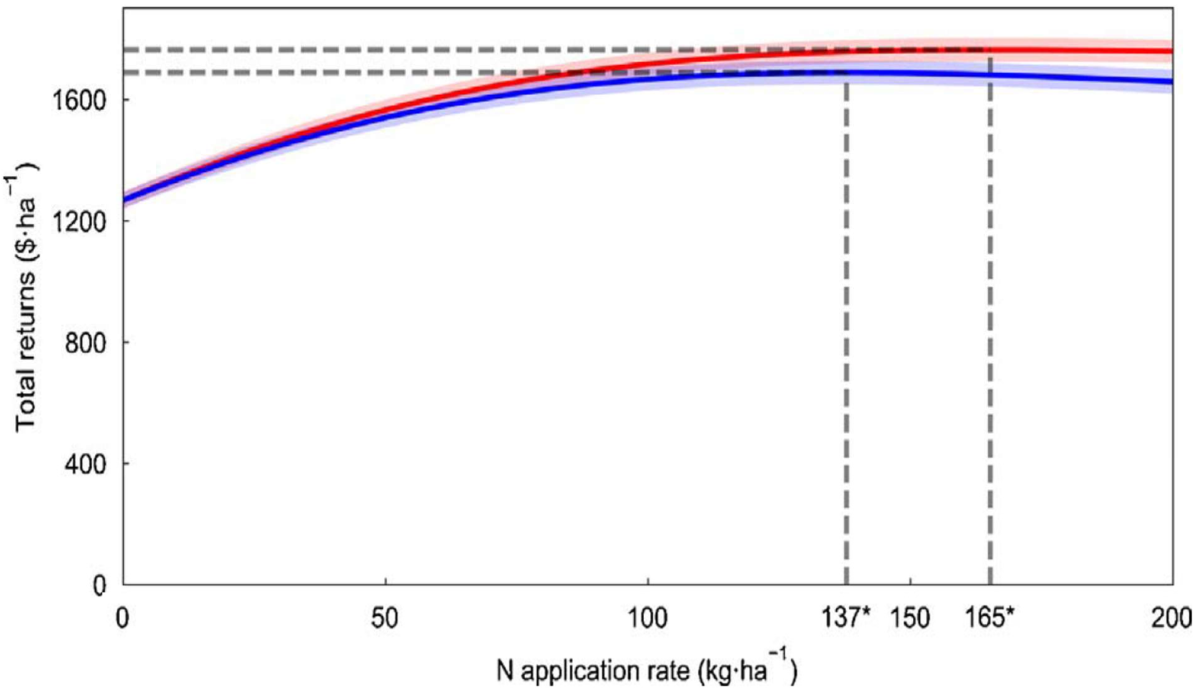
Notes: Figure illustrates case where there are small benefits (i.e., negative marginal costs) of adoption up to 25 percent of the maximum adoption potential.

Figure 2. Profit as a function of nitrogen application rate in the central wheatbelt of Western Australia



Source: Morrison et al. (1986).

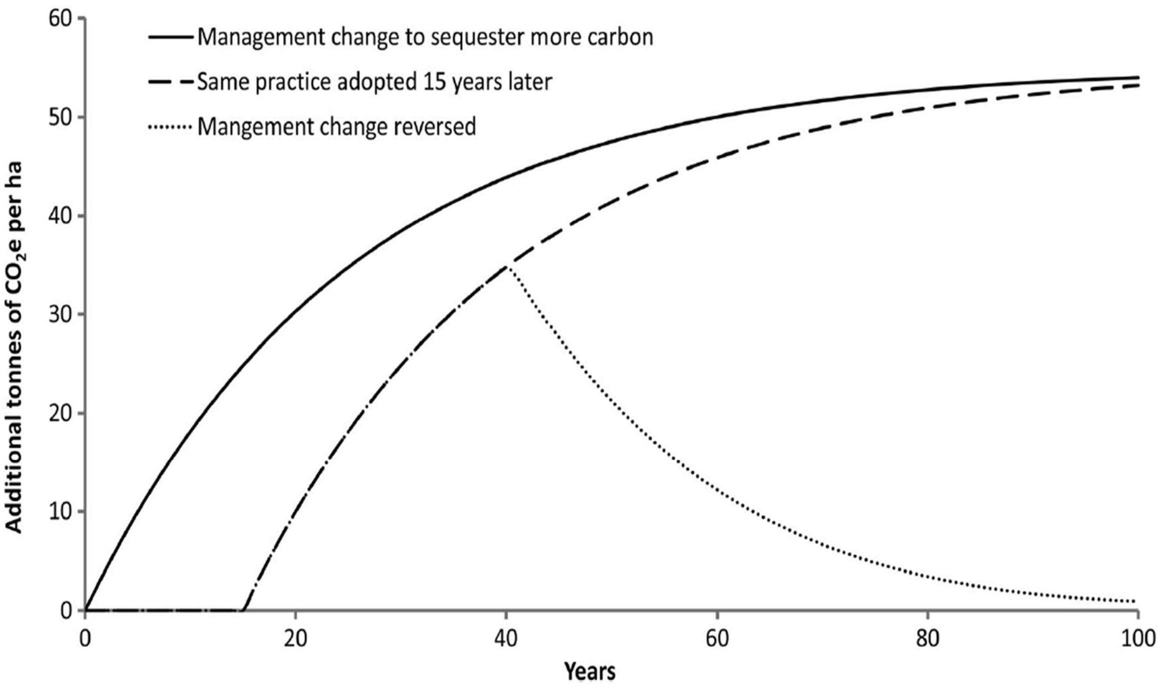
Figure 3. Private net returns (red) and public net returns for nitrogen fertilizer application to corn after soybeans in Minnesota



Source: Gourevitch et al. (2018).

Notes: The public returns (in blue) are "net", that is, the benefits after the social cost of nitrogen pollution (estimated at $0.50 per kg N) have been deducted.

Figure 4. Stylized dynamics of carbon sequestration



Source: Developed by authors.