Private Incentives for Sustainable Agriculture: Improving Water Quality

David J. Pannell[a], Philip G. Pardey[b], Terrence M. Hurley[b], and Jeffrey Coulter[c]

[a]UWA School of Agriculture and Environment, The University of Western Australia, Crawley, WA 6009, Australia
[b]Department of Applied Economics, University of Minnesota, St. Paul, MN 55108, USA.
[c]Department of Agronomy and Plant Genetics, University of Minnesota, St. Paul, MN 55108, USA.

*E-mail address: david.pannell@uwa.edu.au

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David Pannell is Professor of Agricultural and Resource Economics in the School of Agriculture and Environment and Director of the Centre for Environmental Economics and Policy at the University of Western Australia. Philip Pardey is Professor of Applied Economics and Director of the International Science and Technology Practice and policy (InSTePP) center at the University of Minnesota, and Terrence Hurley is the Austin A. Dowell Professor of Applied Economics at the University of Minnesota. Jeffrey Coulter is an Associate Professor and Extension Agronomist in the Department of Agronomy and Plant Genetics at the University of Minnesota. The authors thank Yuan Chai and Connie Chan-Kang for their assistance in the preparation of this report. The research leading to this paper was funded by PepsiCo Inc. The content of this paper is the sole responsibility of the authors.

Private Incentives for Sustainable Agriculture: Improving Water Quality

This is the second in a series of papers concerned with harnessing private incentives to enhance the sustainability of agricultural production. Paper 1 outlines key principles and insights from existing research on the general requirements to achieve changes in agriculture to enhance sustainability. This paper builds on those insights by examining the opportunities to reduce water pollution arising from agriculture, including opportunities for private agribusiness firms to contribute by virtue of their ability to influence the actions of farmers.

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, with substantial gains for the economy, human health, the environment and social values linked to water. In the light of this success, the focus has now switched in many countries to addressing agricultural water pollution (e.g., Rogowski 1990; Oliver and Kookana 2012).

Progress in reducing agricultural water pollution has been more challenging, in part because it principally originates from farms spread across the landscape (non-point or diffuse-source pollution), as opposed to more spatially confined sources, such as urban centers, factories and sewage treatment works (point-source pollution). The diffuse nature of agricultural pollution sources makes it relatively difficult to identify and quantify the various sources, to predict the effectiveness of mitigation actions, and to change the pollution behaviors of all relevant landholders. Predicted intensification of agricultural production in North America, Turkey, Korea, Australia and New Zealand in coming years could further heighten pollution pressures on water systems (OECD 2012).

The next section summarises research evidence about water pollution that results from crop production. Then there are several sections covering the on-farm practices that can reduce that pollution. First, the various available practices are described. The issues around farmers deciding to adopt those pollution-reduction practices are addressed, including their economic benefits and costs to the farmers. Strategies used by governments to promote reductions in agricultural water pollution are presented, and then potential opportunities for PepsiCo to contribute in this area are assessed. We outline the information that would be needed by an organization to evaluate and underpin the strategies for increasing adoption. The geographic focus of the paper is the United States, particularly the Cornbelt, but examples are also provided from various other locations around the world.

1. Water Quality and Crop Production

Key Agricultural Water Pollutants

The main water pollutants from agriculture are nutrients, soil sediments, and pesticides, with less important pollutants including oil products and veterinary products. 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 following the microbial transformation of soil ammonia into nitrates (NO_3, a soluble salt form of nitrogen) can be leached below the root zones of crops and pastures, where it can accumulate in groundwater or be discharged into streams. Nitrogen can also reach streams via surface run-off or drains.[1]

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). Apart from delivering phosphorus, soil sediments

[1] As Keeney and Hatfield (2008, p. 1) describe, "Many ecological problems occur when N is separated from its most common partner, carbon … Nitrification, denitrification, nitrous oxide formation, leaching of nitrate, and volatilization of ammonia are fates of the mobile N atom." In addition to leaching, applied nitrogen can volatize into the atmosphere as the gases nitrous oxide (N_2O), dinitrogen (N_2) or ammonia (NH_3).

also cause other environmental problems, particularly turbidity (see Impacts of Agricultural Water Pollutants below).

Pesticides are widely used in agriculture, and a subset of them are found to contaminate water bodies and/or groundwater. They vary widely in their toxicity to humans and their ecological impacts. For particular pesticides, there are concerns about impacts on human health due to skin contact or ingestion in water (Ongley 1996).

Impacts of Agricultural Water Pollutants

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).

High concentrations of nutrients can cause eutrophication of receiving water bodies, which could be used for public water supplies or be of high ecological value. Nutrients support high growth of algae, which cause 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. This is caused largely by nutrients originating on farms within the Mississippi River watershed, as well as from towns along the river. Alexander et al. (2008, p. 822) estimated that:

> … agricultural sources in the watersheds contribute more than 70% of the delivered N and P. However, corn and soybean cultivation is the largest contributor of N(52%), followed by atmospheric deposition sources (16%); whereas P originates primarily from animal manure on pasture and rangelands (37%), followed by corn and soybeans (25%), other crops (18%), and urban sources (12%).

As well as its impact on ecological values, the Gulf of Mexico dead zone has severely damaged the region's shrimp industry (*Scientific American*, undated). More broadly, seafood and tourism industries in the region bear large costs each year as a result of the dead zone.

Nitrogen (in the form of nitrates) in groundwater can cause localized impacts on human health if the groundwater is drunk. Infants are particularly susceptible and can develop blue baby syndrome (methemoglobinemia) due to a loss of hemoglobin in the blood (Fan and Steinberg 1996). At extremely high levels of nitrates in water, adults can experience a rise in the level of nitrosamines, which are well-documented to be cancer-causing (McCasland et al. 2012).

Soil erosion on farms contributes to siltation of rivers and dams in many parts of the world. Dams are particularly susceptible because sediment can drop out of still water. As this sediment accumulates over time, the water-storage capacity of a dam falls. All river water contains some sediment, and dams are designed to cope with accumulation of sediment for an expected lifespan (e.g., 100 years). Large reservoirs in the United States lose storage capacity at an average rate of around 0.2 percent per year. However, in some cases, the loss of capacity is much more rapid.

> Chixoy is one of a number of very expensive hydrodams built in Central America during the 1970s and 1980s with loans from the World Bank and Inter-American Development Bank despite the very high and accelerating rates of erosion in their watersheds. These dams are now rapidly filling with sediment, leaving small, impoverished countries like Guatemala, Honduras and Costa Rica with huge debts and in desperate need of building new power plants to reduce their dependence on their white-elephant dams. (McCully 1996).

Turbidity is cloudiness or murkiness of water. Rivers become turbid for various reasons, but in many rivers, fine sediment eroded from agricultural soils is the main cause. Turbidity adversely affects the aesthetic qualities of rivers and lakes, with impacts on tourism and recreation. It can increase the cost of water treatment, and harm aquatic life by reducing food supplies, degrading spawning beds, and affecting gill function in fish (Minnesota Pollution Control Agency 2008).

Pesticides in water bodies can cause disruption of ecological systems. For example, Beketov et al. (2013) examined the effect of pesticides on stream invertebrates in Europe and Australia. They found that the species richness and family richness of streams were negatively correlated with their pesticide concentrations, with reductions of up to 42 percent.

Public health impacts from pesticides in water vary widely depending on the pesticide and its concentration. As with nitrates, pesticides can be found in many drinking-water wells in farming areas. Impacts include affecting the nervous system, irritating the skin or eyes, causing cancer and affecting the hormone or endocrine system in the body (McCasland et al. 2012).

2. Practices that Reduce Water-Quality Impacts from Cropping

We group the practices into two broad categories: in-field and off-field practices. In-field practices include adjustments to fertilizer practices, variable-rate technology, no till, conservation agriculture, cover crops, and nitrification inhibitors. Off-field practices include buffer strips, land retirement, flood-plain restoration, artificial wetlands, and bioreactors. Some of these practices can be effective against multiple pollutants (Table 1). The suitability of each practice varies by country and by the locale within a country. Here we focus mainly, but not exclusively, on practices that have been used and researched in the U.S. Cornbelt.

[Table 1. *Summary of key practices for reducing water pollution from agriculture*]

Fertilizer Practices

Raun and Johnson (1999) saw fertilizer management options as the best prospect for reducing nutrient pollution of waterways. They argued that low nitrogen-use efficiency (NUE) of many crops provides a key opportunity. Research by Olson and Swallow (1984) estimated that only about 30 percent of the nitrogen applied to winter wheat in the United States was recovered in the harvested grain.

There are various ways in which fertilizer usage can be modified to reduce water pollution. For example, the 4R nutrient management stewardship system involves four aspects of fertilizer management: the *right* type of fertilizer, rate of fertilizer application, timing of application and location of application. The goal of the approach is "… to match nutrient supply with crop requirements and to minimize nutrient losses from fields. Selection of BMPs [best management practices] varies by location, and those chosen for a given farm are dependent on local soil and climatic conditions, crop, management conditions and other site specific factors" (http://www.nutrientstewardship.com/4rs/ accessed 7 December 2018).

One effective strategy that farmers can adopt is simply to reduce their fertilizer rates. Leslie et al. (2017) found that many farmers in Ontario, Canada apply more than the recommended rate of phosphorus fertilizer, although not of nitrogen. Even where recommended rates are used, an option is to apply lower-than-recommended rates. While this results in lower revenue from crop production, it also reduces costs. In most cases, there are wide ranges of fertilizer rates for which marginal revenue and marginal cost are very similar, so that rates can be reduced at little net cost (Pannell 2006). In paper 1 this was described as the phenomenon of "flat payoff functions".

Another strategy is to delay application of part of the planned nitrogen until a foliar application later in the season to better match the timing of when nitrogen is needed by the plant (Dinnes et al. 2002). In wheat, this strategy has the advantage of increasing grain protein (Woolfolk et al. 2002).

Variable-rate Technology

Variable-Rate Technologies (VRTs) are systems for spatially varying input rates (such as for fertilizers) within a field depending on localised conditions. Raun and Johnson (1999) discuss application of prescribed rates consistent with varying plant needs within the field. This reduces nitrogen leaching from areas where a standard uniform rate would have been higher than needed by crop plants. However, its overall effect on nitrogen losses from a field is not necessarily positive. Modelling 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.

Zero-till and Conservation Agriculture

Conservation agriculture combines reduced tillage with retention of crop residues, to reduce soil erosion. It can potentially reduce nutrient losses from fields, particularly phosphorus, although the evidence is not clear cut. Its performance at reducing losses can be expected to vary depending on soil type, slope and nutrient type. For example, Cooper et al. (2017) found that non-inversive tillage was not effective at reducing nutrient losses from a low-gradient site in Norfolk, UK. Similarly, a study of mainly wheat-based crop rotations grown on sandy soils in Denmark found that decreased soil tillage intensity did not reduce nitrate leaching relative to plowing (Hansen et al. 2010). Sharpley and Smith (1994) report the same results for wheat-based systems planted in seven water sheds located throughout Kansas, Texas and Oklahoma. In contrast, Deasy et al. (2009) reported reductions in losses of phosphorus in surface runoff in the United Kingdom, a finding shared by a review of European studies conducted by Soane et al. (2011), who also noted reduced loss of nitrate via leaching, but only in some cases.

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. Cover crops can reduce soil N leaching, reduce soil erosion, improve soil quality, and enhance habitat for some animals.

Experimental results from the literature showed that cover crops can reduce both the mass of N leached, and the N concentration of the leachate by 20 to 80 percent compared with no cover crop (Meisinger et al. 1991). Management factors that improve N conservation include selection of a species with vigorous fall growth and early establishment. From a water-pollution perspective, grasses and brassicas perform better as cover crops than do legumes, because they are two to three times more effective at reducing N leaching. Summarizing the results from a meta-analysis of 14 (mainly corn, but some barley, wheat, potato and sugarbeet) studies that compared nitrate leaching between bare fallow and cover-crop rotation systems, Tonitto et al. (2006) concluded that the use of non-legume cover crops reduced nitrate leaching by 70 percent on average, with a 40 percent reduction in legume-based systems. In a more recent, comprehensive review of the relevant literature (795 references in total, of which 256 related to the nitrate-water nexus during fallow periods), Justes (2017, p.2) concluded that cover crops reduced the leaching of nitrates into aquifers in the range of 20 to 90 percent. However, they underscored the exceptionally large variation in the extent of leaching, stressing that the intensity of this biophysical process "… is determined by the interaction between the cropping system, farming practices and soil and climate factors; this intensity is therefore site-specific and, more particularly, year-specific, due to its reliance on rainfall amounts" (p. 20)[2].

Based on a set of simulated results, Kladivko et al. (2014), estimated that 34 to 81 percent of the agricultural land in 10 counties in the upper mid-western United States could have cover crops integrated into their corn and soybean cropping systems. They estimated that cover crops have the potential to reduce NO_3 loadings to the Mississippi River by approximately 20 percent. However, these conclusions are based solely on bio-physical criteria, with no account taken for the economic performance of cover crops.

Comparing the N-runoff reduction efficacy of cover crops vis-à-vis other options, Roley et al. (2016) found that cover crops remove less N per acre of practice than two other conservation practices (constructed wetlands and two-stage ditches, which are described below), but that they compare favorably nevertheless because they are applied to a larger surface area which offsets the low removal rates per acre. Zhang et al. (2017) found that cover crops reduced surface runoff flow volume by 32 percent relative to no cover crop, regardless of drainage treatment. As a consequence of this, "… cover crops are best adapted to warm areas with abundant precipitation. Water use by cover crops can adversely impact yields of subsequent dryland crops in semiarid areas" (Dabney et al. 2001, p. 1222).

[2] More specifically Justes (2017, pp. vii-viii) noted that "… the effectiveness of nitrate-trapping "catch [or cover] crops" depends on a number of highly variable factors. The amount of nitrate present in the soil during fallow periods depends not only on farmers' fertilizing practices, but also on the soil characteristics and the weather conditions that year. It is these factors that determine the rate at which organic nitrogen in the soil is biotransformed into nitrate, firstly by mineralization and secondly by nitrification. Moreover, the cover crop planting and destruction conditions, which determine their effectiveness, vary considerably and depend on local conditions as well as on the cropping systems."

The potential nitrate-reducing benefits from the use of cover crops are not without their complications. As Wortmann et al. (2013, p.22) observed "… cover crops use soil water that might otherwise be stored for the next crop, … stand establishment is often difficult due to sowing with less than optimal times and conditions, [s]eed is often sown into heavy crop residues, [and] the time between establishment and harvest of the main crop and the onset of cold weather is often short."

Nitrification Inhibitors

Nitrification inhibitors are chemicals such as nitrapyrin that can be applied to agricultural fields to delay the microbial action on soil ammonia thus reducing nitrate leaching, urease that delays urea fertilizer dissolving in soil water, or polymer coated urea that can 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. Ruser and Schultz (2015) propose that "compared to other measures NIs have a high potential to reduce N_2O emissions from agricultural soils. … From the published data … a reduction potential of approx. 35% seems realistic" (Ruser and Schultz 2015, p. 171).

Although nitrification inhibitors seem promising, they have their complexities and challenges in practice. Firstly, mitigation effects observed on farm are well below those gains observed in experiments. This reflects the cost of inhibitors, which drives lower rates of application in commercial applications. Secondly, inhibitors degrade over time after they have been applied, and may no longer be available at the time when the loss of nitrogen to the environment is highest. This varies spatially, according to regional differences in weather patterns. Thirdly, reductions in nitrogen loss may boost pasture growth, motivating landholders to intensify their farm, which may worsen the off-site impacts of agriculture. Lastly, residues may appear in food products. In New Zealand, the N inhibitor DCD began to appear in minute traces in milk and raised public concerns. Although DCD is not considered harmful, there is a lack of unequivocal evidence that it is completely safe, so the dairy industry in New Zealand has voluntarily ceased using DCD.

Land Retirement

Land retirement 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 of achieving biodiversity and water-conservation goals (Ribaudo et al. 1994). The retired land could be selected for retirement because it has high value for biodiversity conservation, high potential to reduce water pollution (e.g., it is riparian land) or it has a low opportunity cost (a low value for commercial agriculture).

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. They may also provide habitat for wildlife.

Buffer strips can be of varying widths (e.g., 5 meters to 30 meters), and there is a trade-off between financial costs and environmental benefits as the width is increased. Typically, benefits (in terms of reduced pollution) increase at a decreasing rate as the width increases (Mtibaa et al. 2018; Sieber et al. 2010).

Buffer strips can be used in combinations with wetlands and two-stage ditches. They can reduce delivery of sediment, phosphorus, and nitrogen in surface water into waterways. However, in some areas, the main source of nutrients into waterways is via tile drains, and in these cases, nutrient-laden water bypasses grass buffers, resulting in minimal N removal by those buffers (Roley et al. 2016).

Fennessy and Cronk (1997) showed that a riparian buffer zone of 20 to 30 m width can remove up to 100 percent of incoming nitrate. Denitrification is the major pathway of removal and rates depend on nitrate loadings, carbon availability, and hydrology. 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.

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. Benefits relative to simpler drain systems are that the wider expanse of water causes water velocity to decrease and that the overall drainage ditch remains more stable, reducing the need for dredging. From a pollution perspective, the two-stage ditch provides more time and space for N removal processes such as denitrification (Roley et al. 2016).

Using storm-flow simulations, Roley et al. (2012) found that two-stage ditches removed some nitrates from the surface water entering streams, but not to a large extent. In a high-nitrogen site, less than 10 percent of the nitrogen load was removed from storm flow, on average.

Wetlands

Constructed or restored wetlands can be used to reduce nitrogen concentrations in drainage waters before it is discharged to a larger water body. "Wetland sediments are typically anoxic and rich in organic matter, which promotes denitrification" (Roley et al. 2016). For effectiveness, wetlands require that the rate of water flow is slow enough to be processed.

Kovacic et al. (2000) found that wetlands in Illinois decreased nitrate concentrations in inlet water by around 28 percent compared with the outlets. Total phosphorus removal was much less - only 2 percent, with highly variable results between wetlands and years.

Bioreactors

Bioreactors "convert nitrate – the form of nitrogen in farm drainage water – to nitrogen gas, which is environmentally benign and makes up more than three-fourths of the air we breathe" 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). A bioreactor costs around $11,000 to $15,000 to install, and the wood chips last 7-15 years before needing replacement (crop residues last a shorter time). A bioreactor of this size can service the drainage discharge for an area of around 50 to 75 acres at a cost of around $0.80 per lb of N removal (calculated using "Bioreactor Evaluation", a routine developed by the Illinois Natural Resources Conservation Service).

Combinations of Strategies

In order to meet targets for water-pollution reduction, in most environments, it will be necessary to implement combinations of mitigation strategies, and to do so broadly across watersheds. Targeting of effort will help to lower costs and increase effectiveness, but nevertheless, the overall effort required will often be large. For example, the Committee on Water Implications of Biofuels Production in the United States (2008) made the following observation.

> There is a target of 45% nitrogen-load reduction proposed to remediate hypoxia in the Gulf of Mexico (GoM). Our results indicate that nitrogen-management practices (improved fertilizer management and cover crops) fall short of achieving this goal, even if adopted on all cropland in the region. The goal of a 45% decrease in loads to the GoM can only be achieved through the coupling of nitrogen-management practices with innovative nitrogen-removal practices such as tile-drainage treatment wetlands, drainage-ditch enhancements, stream-channel restoration, and floodplain reconnection. (Committee on Water Implications of Biofuels Production in the United States 2008).

Kling et al. (2006) modeled a particular management strategy for the Upper Mississippi River Basin, consisting of the following actions:

i. Retire all cropland within 100 feet of a waterway
ii. Retire additional cropland until 10 percent is retired, based on an NRI erosion index
iii. Terrace remaining cropland with slopes above 5 percent
iv. Implement contouring on all remaining cropland with slopes above 4 percent
v. Install grassed waterways on remaining cropland with 2 to 4 percent slopes

vi. Implement conservation tillage (20 percent no till, 80 percent mulch till) on all non-retired cropland with slopes >= 2 percent

vii. Reduce nitrogen fertilizer rates on all corn acres by 10 percent.

Their model predicted substantial improvements in water quality if this strategy was fully implemented: an overall reduction in sediment load reductions of 63 percent, a total nitrogen reduction of 22 percent, and a total phosphorus reduction of 32 percent. Note that even this combination of actions would not be sufficient to meet the 45 percent target for nitrogen-load reduction referred to by the Committee on Water Implications of Biofuels Production in the United States (2008).

3. Adoption of Water Quality-Improving Practices in Agriculture

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, both in terms of the numbers of farmers and the agriculture land area covered under these programmes. (OECD 2012). Unfortunately, this increased adoption has not yet translated into improved water quality in most cases, because of time lags, and perhaps (setting aside the more recent years) because of high prices for agricultural commodities, which encourage high input rates (OECD 2012).

In Paper 1, factors that are likely to influence the adoption or non-adoption of beneficial practices for water quality were outlined. Here we examine each of the practices for improving water quality (described above) and comment on potential barriers to, or drivers of, their adoption by farmers. The key adoption factors that we consider for each practice are: (a) ease of trialing the new practice (including the observability of results, risk of trial failure, ability to implement on a small scale); (b) up-front costs of implementing the practice; (c) profitability of the practice; (d) time lags between implementation and benefits; (e) riskiness of the practice and uncertainty about its benefits; (f) complexity of the practice.

Table 2 shows a subjective score for each group of practices for each of the adoption factors. Scores range from 1 (highly unfavorable for adoption) to 7 (highly favorable for adoption). The first two adoption factors (ease of trialing and up-front costs) mainly influence how quickly adoption occurs, although in extreme cases (trialing impossible, up-front costs enormous) they can mean that adoption never gets off the ground at all. The other four factors (profitability of the new practices, the time lag until it delivers benefits, riskiness of the new practice and complexity of the new practice) are key drivers of the eventual level of adoption.

[Table 2. *Assessment of key adoption factors for each practice for reducing water pollution from agriculture*]

Of the practices discussed, fertilizer practices have the most favorable adoption assessments overall. They are easy to trial, have low up-front costs, and are not complex. They may result in slightly lower profits than business-as-usual fertilizer practices, so are given a profitability score of 3 (4 being neutral). In fact, the average rates of nitrogen fertilizer applied in the United States are already slightly below 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. Across corn, cotton, spring wheat, and winter wheat, around 30 percent of the cropped area received more nitrogen fertilizer than a realistic benchmark rate, and farmers spent almost $1 billion on excess nitrogen application (Wade et al. 2015).

Other relevant nitrogen fertilizer practices include timing and placement. 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) and estimated the proportion of farmers who met all four. The numbers were mostly very low.

> "For cotton, 24 percent of acres receiving nitrogen fertilizer were on farms that met all four nitrogen application criteria; 1 percent of these met these criteria and used no-till. In corn, 6 percent met all four nitrogen criteria; 2 percent also used no-till. For spring wheat, 2 percent of treated acres met all four criteria and no farmers reported using these practices while also using no-till. Farmers planting winter wheat met three of the

> four nitrogen criteria (all except fall application) on 10 percent of treated acres; 4 percent met all these criteria and used no-till.” (Wade et al. 2015, p. 20)

Trialing of variable-rate technology is feasible but requires farmers to access the technology prior to purchasing it. The technology is somewhat costly, and benefits are low because of flat payoff functions (see Paper 1), so profitability is low.

Zero-till (or no-till) can be trialed relatively easily, in terms of in-field actions, but assessing the on-farm and ultimate water quality effects of the practice is difficult, in part because of the time delays until benefits are observed. The low ease-of-trialing score reflects the difficulty of learning from a trial, rather than the difficulty of physically conducting a trial. Depending on the farming system and context, it may slightly reduce or somewhat increase profits. Where water availability to crops is a problem, zero-till can reduce the riskiness of cropping by retaining more soil moisture through reduced evaporation. Wade et al. (2015) reported that just under 40 percent of the area of four major crops in the United States was sown using zero-till or strip-till in 2010-11.

Of these adoption factors, the main ones influencing adoption of cover crops is their negative influence on profitability and the increased complexity that they bring to the farming system. Roesch-McNally et al. (2017) describes how some farmers have overcome barriers to adoption of cover crops, but finds that the barriers are still significant. Wade et al. (2015) found that cover crops were used on less than 2 percent of U.S. cropland.

The remaining practices broached above (from nitrification inhibitors through to bioreactors) have broadly similar sets of scores. Trialing, or learning from trialing, is not straightforward, up-front costs are moderate to high in most cases, and profitability is low. Several of these practices add to the complexity of farming. Moreover, they are liable to involve a lag time until they deliver benefits, a significant share of which are likely to be public benefits (externalities) rather than private benefits to the farmers

4. 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). The on-farm economics of strategies to reduce water pollution can be complex. In this report, we are focused on water quality, but farmers will need to weigh up issues related to farming logistics (e.g., sowing time), weed management, soil moisture, and so on. For example, cover crops have multiple impacts. As well as retaining moisture in the field, 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 (over both time and space) and case-specific. This makes them very difficult for policy agencies (or private firms) to estimate the farm-level viability and landscape-level efficacy of any one of these practices (or a combination thereof) for particular farms in particular locations. Published estimates of the costs and benefits to farmers are helpful but are only indicative.

Any initiative to encourage greater adoption of pollution-mitigating practices has to accept the reality that the on-farm economics of the practices are highly variable from farm to farm. Ideally, the initiative would encourage farmers with low abatement costs to reveal themselves. One tool to do this is a reverse auction, in which farmers bid for payments from an environmental organization, with the auction being won by those farmers whose bids offer the best value for money (benefit per dollar spent). This approach can be useful for targeting effort and resources (e.g., Stoneham et al. 2003), although it requires effort, expertise and information to undertake.

In the following review of evidence, unless otherwise stated, the economics of the practices are expressed without accounting for payments from government schemes to encourage their adoption.

Fertilizer Practices

Based on published economic analyses, adjustments to fertilizer strategies do appear to offer an opportunity in many cases. Some farmers fertilize to maximize yield. However, the profit-maximizing fertilizer rate is always somewhat less than the yield-maximizing rate, so rate cuts could increase profits in these cases (Pannell 2017), providing environmental benefits as a spinoff. The following statistics are for the United States as a whole.

> "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)

The benchmark rates used to derive these results were based on physical or biological criteria, rather than economic criteria (National Resource Conservation Service 2012). Given that economically optimal fertilizer rates are likely to be lower than biologically determined rates, the potential for rate reductions that would be financially beneficial to farmers may be even greater than suggested by the numbers provided by Wade et al. (2015). Christianson et al. (2013) also found that reductions in nitrogen fertilizer rates in the U.S. Midwest could be done at negative cost (i.e., increased profit).

Even where fertilizer is applied at rates that are economically optimal for the farmer, it is generally possible to reduce rates significantly without substantial reductions in profit. This reflects the phenomenon of flat payoff functions (Pannell 2006), as outlined in Paper 1. We are conducting a separate analysis of this option. However, past research reinforces that this strategy may provide opportunities. Doering et al. (1999) found that restricting fertilizer use by farmers would be relatively cost-effective, more so than a nutrient tax, use of wetlands or use of vegetation buffers. Hansen et al. (2012) also found that adjustments to nitrogen fertilizer practices are the most cost-effective way of reducing nitrogen pollution, particularly for fields in a corn-corn rotation. Similarly, Ribaudo et al. (2001) found that within-field nitrogen management in the Mississippi basin is the most efficient strategy for delivering reductions in nitrogen losses of up to 26 percent. A strategy based on interception of nitrogen outside the field in wetlands is more expensive. “The major reasons for the higher cost of wetland restoration are the opportunity cost of retiring cropland and the cost of restoring wetland functions on cropland” (Ribaudo et al. 2001, p. 195). If the target is for more than 26 percent N loss reduction, the yield (and, by extension, expected profit) losses from reduced nitrogen rates become large enough that it is optimal to include wetland restoration in the strategy.

As noted earlier, Christianson et al. (2013) estimated that there would be net financial benefits to farmers in the U.S. Midwest from reducing N fertilizer rates: $1.60 benefit per kg of N loss abatement per year. In addition they estimated that switching from fall to spring application time for the fertilizer would be beneficial for farmers, by $14 per kg of N loss abatement per year. These were the only two of the strategies that they examined that generated financial benefits for farmers.

Variable-rate Technologies

Another fertilizer-related strategy is use of variable-rate technologies (VRT). The prospects for VRT delivering substantial reductions in water pollution appear to be small. In addition to the low or zero savings in nitrogen application that were noted earlier, these technologies are usually not financially highly attractive to farmers, as another consequence of flat payoff functions (Pannell 2006; and paper 1 in this series). Earlier 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 was negative (Watkins et al. 1998).

Recent work by Schimmelpfennig (2016) using USDA’s farm-cum-field level ARMS (Agricultural Resource Management Survey) data spanning the period 1996 to 2013 are in line with the earlier findings. He reports only minor gains on average from deploying VRT; both operating profits and net returns on corn farms were raised by an estimated 1.1 percent. Around 26 percent of the surveyed farms had adopted VRT technologies in 2012. Technology uptake was biased towards larger farms (used on just 12 percent of farms less than 600 acres, and 40 percent of farms over 3,800 acres), likely a reflection of the costs involved in purchasing and operating these technologies. VRT technologies were sometimes used in conjunction with other precision agriculture technologies such as yield monitors, yield mapping, and GPS aided guidance systems. The results of these economic studies suggest that the existing modest adoption of VRT is probably driven by farmer interest in the technologies per se, rather than proven financial benefits.[3]

Zero-till and Conservation Agriculture

Zero-till (or no-till) and various forms of conservation tillage are widely practiced in countries with broad-scale commercial agriculture, including the United States, Australia, Argentina, Brazil, and Canada. Adoption rates in these countries range from around 40 percent of the 2010/11 crop (corn, soybean, wheat, and cotton) area in the United States (Wade et al. 2015) to 90 percent in the main cropping regions of Australia in 2008 (Llewellyn et al. 2012).

Adoption of zero-till in most other counties has been less extensive, and it is likely that it is less suitable to those countries for a mix of technical and economic reasons.

The economics of zero-till are complex and difficult to quantify, because the practice can have multiple impacts on the farming system (Pannell et al. 2014). There are numerous studies of the on-farm economics (e.g. Aase and Schaefer 1996; Stonehouse 1997; Janosky et al. 2002), but many are too simplistic to provide actionable results. Nevertheless, the persistence of high adoption rates in certain countries can be taken as evidence that the farm-level economics are favorable in many cases in those countries. Given the U.S. adoption rate of 40 percent, there are likely to be significant areas (e.g., an additional 10 percent) where adoption could be

[3] Lambert and Lowenberg-deBoer (2000) conducted a review of the profitability of precision agriculture drawing their evidence from 108 published studies. VRT was the most prevalent technology evaluated but in our judgement a lack of standardization of what constitutes "profitability" and variations in the crops, years (i.e., weather, etc), and production circumstances covered makes it difficult to draw any overall conclusions on the profitability of this technology from this compilation. Swinton and Lowenberg-deBoer (1998. P. 439) used a partial budgeting approach to evaluate the results from nine university field trial studies of SSF (site-specific farming) and concluded that "… VR [variable rate] fertilizer application was unprofitable on wheat and barley, sometimes profitable on corn, and profitable on sugarbeet."

increased at low cost. This differs from cases where adoption is already very high, in which case there is little room for upside. Conversely, in cases where adoption has been sustained at very low levels for some time, the economics are probably too adverse for substantial increases to be realistic.

Cover Crops

As noted above, Christianson et al. (2013) found positive economic results for fertilizer management strategies to reduce nutrient exports. They also analyzed other in-field practices, cover crops and crop rotations, but found that they were the least cost-effective of the practices studied, with mean abatement costs of \$55 and \$43 kg N^{-1} yr^{-1}, respectively. Roley et al. (2016) also found that cover crops were the least cost-effective method for nitrogen mitigation, compared with wetlands and two-stage ditches.

Nitrification Inhibitors

There has been little analysis of the economics of nitrification inhibitors, apart from the work of Graeme Doole. Looking at dairy farms in New Zealand, Doole and Parangahawewa (2011) found that, "… with an assumed 10 percent increase in pasture production in response to nitrification inhibitor application, nitrification inhibitors are a profitable innovation because greater pasture production supports higher stocking rates. Nonetheless, their overall impact on farm profit is low" (Doole and Parangahawewa 2011, p. 1031). However, other evidence indicated that there is no increase in pasture production associated with use of nitrification inhibitors (MacDonald et al. 2010), in which case their economics would be less favorable. Doole (2014) employed that more realistic assumption and found that nitrification inhibitors were not profitable for New Zealand dairy farmers. Focussing on mitigation of greenhouse gas emission from dairy farmers, he concluded that nitrification inhibitors would be included in the most efficient package of mitigation measures only if high abatement rates were required. A similar result was found by Doole and Romera (2014) for reductions in nitrogen leaching. As noted earlier, the New Zealand dairy industry subsequently adopted a policy of not using nitrification inhibitors due to potential consumer concerns about residues in milk.

Land Retirement

Retirement of land from agriculture to a non-agricultural land-use is another option for reducing nutrient and sediment pollution of waterways. Part of the cost of this option is the opportunity cost of converting the land – in other words, the profit foregone. Land retirement can have a high opportunity cost, but can still be worthwhile (i.e., environmental benefits exceed financial costs) if it is well targeted (Ribaudo et al. 1994). The Conservation Reserve Program in the United States (Hellerstein 2017), and the Set Aside approach formerly used in Europe are two examples of this approach (Fraser 2015). Given the choice, farmers will, of course, choose land with relatively low opportunity cost to enroll in such schemes. Even so, significant areas of retirement are expensive to achieve in most cases. This is especially the case if retirement is targeted to land with the high capacity for pollution abatement, rather than land with the lowest opportunity cost (e.g., Roberts et al. 2012).

Buffer Strips

We move now to pollution mitigation practices that are implemented outside the field, starting with buffer strips. This, and the following three practices generate public benefits through reducing water pollution, but from the farmer's perspective, they generate only costs (apart from the extent to which farmers place a value on pollution reductions).

As noted above, buffer strips can be constructed with different widths of vegetation. Their pollution-reduction benefits increase at a decreasing rate as their width increases, whereas their cost increase is approximately linear. If their benefits can be quantified in monetary terms, knowledge of these two relationships can be used to calculate the optimal buffer width. In many cases, the optimal balance occurs at a width that delivers less than 100 percent mitigation of pollution. Two studies that did this calculation have both concluded that narrow strips are most cost-effective: Sieber et al. (2010) in Germany and Mtibaa et al. (2018) in Tunisia. Although narrow buffer strips are less costly than wider options, they generate a large share of the potential benefits and so would be recommended to program managers.

Jeffrey et al. (2017) estimated the cost to farmers in Alberta, Canada from establishing 10-meter-wide buffer strips. From simulations for a representative Alberta cropping farm, they estimated that the mean cost to farmers from establishing buffer strips was CAD$73 per year per hectare of land removed from production. Of concern is their finding that farmer participation in government programs to reduce the risks faced by farmers (Growing Forward and Growing Forward 2) would increase the cost of buffer strip establishment to more than CAD$100 per year per hectare. This is because payments to farmers under the scheme increase the profitability of cropping, and therefore increase the opportunity cost of converting cropland to buffer strips.

Flood-plain restoration (two-stage ditches)

No published evidence expressed in $ per unit area.

Wetlands

No published evidence expressed in $ per unit area, but cost is likely to be moderate or higher.

Bioreactors

The National Resources Conservation Service (undated) in Illinois developed a software tool to estimate the required size, cost, and performance of a bioreactor installed in a field with a specified soil and within a specified county in Illinois.[4] To illustrate the results, for a 100-acre field, using default settings, the estimated cost of the recommended bioreactor is $18,800 in total, or $188 per acre treated, or $0.69 per lb of N removed. Alternatively, for a 25-acre field, with a system life of 5 years (instead of 10), the recommended bioreactor cost $6,800 in total, or $272 per acre treated, or $2.00 per lb of N removed.

Cost Effectiveness of Pollution Abatement

Results from some studies are expressed in terms of cost-effectiveness (the cost per unit of pollution abatement) rather than the cost per unit area. Comparing the economics of wetlands,

[4] See www.wq.illinois.edu/dg/Equations/Bioreactor.exe (accessed 17 September 2018). This address provides an exe program that may not operate on all operating systems.

cover crops and two-stage ditches, Roley et al. (2016) found: “Wetlands were the most cost-effective practice (in $ per kg N removed) over both time horizons. Over 50 years, the two-stage ditch ranked second in cost-effectiveness and cover crops were least cost-effective, while over 10 years, cover crops were second and two-stage ditches were least cost-effective [because of their high up-front costs].”

Christianson et al. (2013) calculated the cost-effectiveness of seven pollution abatement methods. As noted earlier, two of the methods had negative costs (i.e., positive benefits) of abatement: springtime nitrogen application (rather than fall) at −$14, and nitrogen application rate reduction at −$1.60 per kg N per year. The in-field vegetative practices of cover crop and crop rotation were the least cost-effective (means: $55 and $43 per kg N per year, respectively). With means of less than $3 per kg N per year, controlled drainage, wetlands, and bioreactors were fairly comparable with each other. They concluded that no individual technology or management approach will be capable of addressing drainage water quality concerns in entirety.

5. 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. Typically these have taken pollution reduction targets as given, rather than attempting to trade off benefits and costs to determine optimal targets.

In the United States, the work of Cathy Kling and her collaborators stands out. 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 (particularly SWAT) (Kling et al. 2006; Kling et al. 2014; Panagopoulos et al. 2014). Theirs is the only work to analyze the entire Upper Mississippi River Basin with an approach that considers adoption of a set of conservation practices, links conservation practices to instream water quality, and does so at a reasonably fine spatial scale (119 sub-watershed for the whole Basin). They estimate the cost of each management practice considered in each sub-watershed 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). As noted earlier, their work highlights the great difficulty of achieving ambitious pollution reduction targets. 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 (N_2O) emissions. For a particular set of assumptions, they found that the privately optimal nitrogen rate for corn following soybeans was 165 kg per ha. There was considerable uncertainty about the magnitude of the social costs, so they investigated a range of costs, from $0.05 to $10.00 per kg of N. Because of the flatness of the payoff function, even small social costs led to relatively large reductions in the optimal nitrogen rate when the social cost was considered. For example, Figure 1 shows results for a social cost of $0.50 per kg of N applied (a "mid-range" estimate).

[Figure 1. *Private versus public net returns to nitrogen applied to Minnesota soybeans*]

Internalizing the $0.50 social cost reduces 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 very small, at around $6 per hectare (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 social cost of just under $1.00 per kg of N. The higher the social cost, the lower the optimal rate. For example, it falls to 66 kg N per ha for a social cost of $3.47 per kg N.

Beverly et al. (2016) developed an integrated model for one region (the Burnett Mary region) within the catchment area for the Great Barrier Reef in Australia. They modeled farms in five watersheds including a mix of farm types (livestock grazing and sugarcane) and a mix of farm sizes. They explored two different sets of targets: one matching existing policy ("Reef Plan

Targets"), and more ambitious target ("Ecologically Relevant Targets"). They modeled multiple pollutants: dissolved inorganic nitrogen (DIN), dissolved organic nitrogen (DON), particulate nitrogen (PN), dissolved organic phosphorus (DOP), dissolved inorganic phosphorus (DIP), particulate phosphorus (PP), total suspended solids (TSS) and herbicides (PSII). The model included a wide range of sustainable management practice from which the model could select. Their model identified the optimal practice changes on farmers of different types (grazing or sugar) accounting for a range of soil types, allocated spatially across the whole region. This research demonstrates how complex and demanding it is to do a study at this level of detail. The researchers found that pollution-reduction targets can be met at lower costs if agricultural practices are optimized over the whole region, rather than requiring each individual watershed to meet targets. Some watersheds can reduce pollution at relatively low cost. With appropriate targeting of effort, the relatively modest targets built into current policy could be achieved within the existing program budgets. However, a requirement that each individual watershed should meet the specified percentage reduction targets was inefficient and could not feasibly be met for all watersheds. The achievement of more ambitious targets, consistent with what the community probably expects, would be substantially more expensive than existing budgets allow.

A broadly similar study by Doole et al. (2013) used optimization to investigate the cost-effective management of multiple, nonpoint pollutants (P and sediment) affecting two river catchments in Victoria, Australia. This study also accounts for effects of actions on multiple pollutants. The region includes a mix of cropping and grazing, so the model represents seven different land uses. A biophysical model was used to estimate pollution reductions for 266 combinations of the seven alternative land uses. These results were embedded in a spatial economic model, which selects practices across the catchments. Results show that the cost of abatement depends non-linearly on targets: as the target becomes more ambitions, costs increase at an increasing rate, especially for phosphorus. For this context, the model indicated that the most cost-effective abatement is in the livestock zones, rather than cropping zones. Streambank protection and repair of erosion gullies are the most cost-effective mitigation options. This study highlights the need to be clear about which pollutants are most important to reduce, as the best strategies for mitigation are different for different pollutants. Like Beverly et al. (2016), it shows that there can be substantial cost savings from targeting abatement to selected locations rather than applying it uniformly across the landscape.

Another Australian study, by Roberts et al. (2012), looked at optimal management of phosphorus in a large watershed for the Gippsland Lakes, which have high values for tourism, fishing and the environment. Like the previous two studies, this is a watershed with multiple agricultural industries and various nutrient management actions for each industry. The analysis concluded that the top priority was to address nutrient losses from the irrigated dairy industry. The work highlighted the importance of considering realistic levels of landholder adoption of new practices and the feasibility of achieving target adoption levels. Costs, landholder adoption of new practices and socio-political risks appear neglected in the formulation of many water quality programs. Similar to the previous study, they found that marginal costs of pollution abatement increase dramatically at higher abatement target levels.

Comparable analyses have been conducted for dairy farms close to Lake Karapiro in New Zealand by Doole and Pannell (2012) and Doole (2015). These studies add to the weight of evidence behind some of the conclusions already mentioned: the benefits of targeting effort, the increasing marginal cost of higher abatement targets, and the high overall cost of achieving ambitious targets. The costs amounted to "… 4 or 14 per cent reductions in profit to achieve 30 or 50 per cent N reductions with differentiated emissions standards. A differentiated policy instrument results in cost savings relative to a uniform standard, but in this case study, the difference in costs is not large" (Doole and Pannell 2012, p. 99).

Van Grinsven et al. (2013) conducted a benefit-cost analysis of the use of nitrogen fertilizer in Europe as a whole. They estimated that "… the economic benefit of N in primary agricultural production ranges between €20-80 billion/yr and is lower than the annual cost of pollution by agricultural N which is in the range of €35-230 billion/yr. Internalizing these environmental costs would lower the optimum annual N-fertilization rate in Northwestern Europe by about 50 kg/ha." (Van Grinsven et al. 2013, p. 3571).

Hyytiäinen et al. (2015) presented a spatially explicit bioeconomic model to study transboundary nutrient pollution of the Baltic Sea. This represented nine countries that surround the Baltic Sea and included the benefits of pollution reduction determined from community surveys in each of the countries. For each country, they estimated benefits and costs of five discrete policy goals. Costs and benefits varied significantly between countries. The optimal target for nutrient reductions is less ambitious than envisaged by the Baltic Sea Action Plan. Achieving a version of the optimal target that leaves not country worse off would generate overall benefits of €170 million per year.

A factor that is not emphasized in some of these studies is the time lags between management changes and environmental improvements. In certain cases, these time lags can be very long, particularly if pollutants are delivered to water bodies via groundwater. For example, Delaware's Inland Bays suffer from nitrates in groundwater. It is estimated that it takes around 50 years for groundwater to travel from agricultural land in the watershed to the Bays (Meals et al., 2010).

6. Strategies Used by Government to Promote Water-Quality-Improving Practices

Billions of dollars are spent around the world in public programs to improve water quality. Policies have often fallen short of their targets to reduce the negative impacts of agriculture on water quality (OECD 2012).

One factor contributing to the challenges of delivering effective policy is that agricultural water pollution comes from diffuse sources – sources that are widely dispersed, hard to identify, and numerous. Designing policies to control diffuse-source agricultural pollution is more difficult than policies to address point sources of pollution, because they are: usually invisible due to low concentrations taking diffuse, indirect and often complex pathways into water systems; commonly difficult and costly to measure; generally cumulative in their impact on water systems due to the effects of runoff and leaching from large areas; and highly variable in space and time because of influences outside of the farmers' control, such as the weather and different soils.

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 policy makers.

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).
- Farmer advice and education (information provision), sometimes combined with measures to build farmer capacity or social capital.
- Economic instruments, such as water-quality trading schemes.
- 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 3).

[Table 3. *Policies used to address water quality impacts from agriculture in the United States*]

In the United States, regulatory constraints on water pollution are mainly used for point-source polluters (using discharge permits) because in these cases, discharge points can readily be identified and metered (Shortle and Uetake 2015). The same system is used to regulate major agricultural point sources, such as large Concentrated Animal Feeding Operations, but is generally not used for agricultural non-point-source polluters.

As shown in Table 3, there are a number of federal programs that provide farmers with financial incentive payments to encourage adoption of practices that reduce water pollution. The main programs are the Environmental Quality Incentives Program (EQIP), the Conservation Reserve Program (CRP) and the Conservation Stewardship Program (CSP), but there are also a number of smaller or regionally targeted schemes. Despite the considerable sums of money spent in these programs, “Water quality is … an area where current agri-environmental policies are not achieving established goals, leaving agriculture as a leading cause of water quality problems” (Shortle and Uetake 2015, p.4).

Water Quality Trading schemes typically involve industrial and municipal polluters meeting their requirements under the NPDES permit system through the purchase of pollution reduction credits from agriculture. However, many such schemes have seen very few trades (Olmstead 2010). This may be due to high transaction costs and adverse exchange rates (trading ratios) due to uncertainty. These both arise from the non-point-source nature of most nutrient and sediment pollution from agriculture. Estimating farm-level emissions is costly, due to the presence of multiple small polluters, the difficulty of know what practices are used by each farmer, and the dependence of outcomes on complex hydrological and weather processes (Suter et al. 2008).

Shortle and Uetake (2015, p.26) note that “The dominant approach to state management of agricultural nonpoint pollution in practice is not regulatory, but is instead the application of voluntary compliance mechanisms that encourage the adoption of agricultural pollution control practices (referred to as BMPs) through education, technical assistance, and financial support through cost-sharing subsidies for the adoption of [Beneficial Management Practices] BMPs”. For example, Ferguson (2015) outlines education efforts to reduce the environmental impact of irrigated crop production in the Platte River Valley in Nebraska. Given that the on-farm practices being promoted are costly to some degree, the success of these information-based approaches at achieving voluntary uptake has been limited.

It is notable that in many of the above policies, the focus is on reducing proxies for pollution, rather than directly reducing the impacts of pollution. Most commonly the policy aims to reduce inputs (such as fertilizers, pesticides) or movement of nutrients and sediment into water bodies (e.g., by constructed wetlands), but in a few cases, the aim is to regulate ambient concentrations of water pollutants (Shortle and Horan 2001). All of these may be poorly correlated with actual damages from pollution.

7. Potential Opportunities for PepsiCo to Contribute to Reducing Water Pollution

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

Many studies show that a targeted approach to abatement of water pollution 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.

As a supplement to this report, the research team has begun compiling the data required to conduct a new meso-scale spatial analysis based on a set of criteria agreed with scientists, potentially including variables related to soil type, rainfall amounts, timing and intensity, slope, proximity to water bodies, ground cover, crop type, participation in government agri-environmental programs, current land management practices, and so on. The aim is to explore the feasibility of targeting hot-spots for intervention and demonstrating an approach to doing so.

Informing and Persuading

Traditionally farm advice programs were primarily provided by government, but there has been a growth in water supply utilities and the agro-food chain providing farm advisory services. For example, Walmart (2018, p.37) has set environmental standards for the food it purchases, and Land O'Lakes SUSTAIN supports the provision of sustainable farm business management advice to its farmer clients.[5]

The aim of informing and persuading farmers is to get as far as possible with existing available sustainable practices without having to provide additional financial support or other forms of incentive. PepsiCo actively participates in some of these broad-based initiatives, but could also complement (or even substitute) these efforts with a more targeted approach to creating, collating and analyzing (site- and time-specific) research-backed information of relevance to the particular farmers, or farming communities, from whom they source their primary agricultural inputs. This may involve PepsiCo developing new university-farmer-input supply relationships to enable the development and delivery of actionable information to their farmer partners.

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 "lose" option has clear limits: evidence shows some adoption (in the right socio-economic circumstances) of sustainable practices that impose low net costs on farmers, but that adoption tends to fall away rapidly as the net costs on farmers increase. Farmers' willingness to adopt is also influenced by their beliefs about how beneficial the practices will be in terms of public environmental benefits. Environmentally concerned farmers are prepared to make more of a private sacrifice if they are convinced that the public benefits are larger. The planned 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.

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[5] See www.landolakessustain.com. There are other programs seeking to inform and thereby affect the environmental outcomes of farming, such as Field to Market (https://fieldtomarket.org/), Soil Health Partnership (www.soilhealthpartnership.org), and the Midwest Row Crop Collaborative (https://midwestrowcrop.org/).

The other 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 third-party advice or requests for behavioral change. For example, in an Indian example, telling high users of energy how their consumption compared with that of their neighbors prompted them to use less (Sudarshan 2017). Sometimes the solution calls for low-cost practical support to convince people to change. Research in Britain into why people did not take up financial incentives to reduce energy consumption by insulating their homes found one possibility was the hassle of clearing out the attic. A "nudge' was designed whereby insulation firms would offer to clear the loft, dispose of unwanted items and return the rest after insulating it. This led to a threefold increase in take-up of an insulation grant.

Pepsi 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.

Empowering

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

The spatial information from the *Targeting* component (above) and insights about adoption of new practices from Paper 1 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.

The advantage of this strategy is that, if successful, it would increase the willingness of farmers to voluntarily adopt sustainable practices. For example, in a situation where there are strong potential public environmental benefits from pollution abatement but there is a lack of pollution abating practices that are attractive to farmers, providing a new practice with positive private net benefits for farmers would improve pollution outcomes by improving the propensity for farmers to voluntarily adopt sustainable practices.

Coordinating

This is about PepsiCo using 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. It involves PepsiCo adding value to the other 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 public policy, rather than relatively untargeted approaches which are often seen 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 for the crops and production locales of most concern to the company's supply chain, or the R&D to develop new technologies.

Incentivising

We understand that Pepsi is unlikely to choose to provide direct financial support to farmers to incentivize their decisions about adoption of sustainable practices. However, Pepsi may be able to contribute to incentivization in other ways, such as: 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.

An existing knowledge gap is about the extent to which, and circumstances in which, incentive payments tend to crowd out voluntary adoption of sustainable practices by farmers. Pepsi could support socio-economic research in this area, to improve the overall effectiveness of incentive payments where they are used.

Another potentially high-payoff possibility is investment in research to develop and test novel mechanisms and instruments, to provide maximum benefit for the available public resources. For example, reverse auctions are used to identify those farmers who are willing to deliver environmental benefits at least cost in terms of incentive payments (e.g., in the Conservation Reserve Program and in various Australian programs). Research could attempt to develop novel mechanisms for revealing this type of information without necessarily having to invest large amounts in incentive payments (to be dispersed through auctions).

8. Conclusion

Agriculture is now the main source of water pollution in many countries, particularly countries where farmers apply high levels of chemical fertilizers to crops. There are various pollutants, but the main ones are nitrogen, phosphorus, and sediment. These reach water bodies as an unintended side effect (or "externality") from standard agricultural practices, delivered in surface waters, in groundwater or via drainage.

There is a range of practices that can be adopted by farmers to reduce water pollution, including practices within the field (e.g., changes in fertilizer rates, timing or type) and practices outside the field that intercept pollutants (e.g., constructed wetlands or bioreactors). The effectiveness of these practices and their attractiveness (or otherwise) to farmers is case-specific. The practices also vary in their relevance to different pollutants.

Modeling results for various case studies indicate that adoption of these practices by farmers has the potential to significantly reduce water pollution. However, it also highlights the difficulty of fully mitigating water pollution problems from agriculture. Moderate reductions in pollution are possible at low to moderate cost, and for some farmers, it may even be possible to reduce pollution with a net benefit for farmers, but ambitious targets to mitigate most pollution are likely to be very costly to meet.

An implication is that initiatives to reduce water pollution from agriculture are likely to benefit from a targeted approach. Analyses to select particular locations, farming types and pollution mitigation practices can potentially improve the impacts substantially.

A second implication is 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?

Based on are consideration of all these issues, it appears that the best opportunities to reduce nutrient losses at low cost relate to fertilizer management. Both rates and timing of application of fertilizers may be worth addressing.

There are various ways in which private agribusiness firms could contribute to the delivery of public goods by reducing water pollution. We have discussed five broad categories of approaches that could be further explored, under the headings Targeting, Informing and Persuading, Empowering, Coordinating and Incentivizing.

References

Aase, J.K. and Schaefer, G.M. (1996). Economics of tillage practices and spring wheat and barley crop sequence in the northern great plains. *Journal of Soil and Water Conservation* 51 (2), 167-170.

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.

Aronsson, H., Hansen, E.M., Thomsen, I.K., Liu, J., Øgaard, A.F., Känkänen, H. and Ulén, B. (2016). The ability of cover crops to reduce nitrogen and phosphorus losses from arable land in southern Scandinavia and Finland. *Journal of Soil and Water Conservation* 71(1), 41-55.

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.

Beketov, M.A., Kefford, B.J., Schäfer, R.B. and Liess, M. (2013). Pesticides reduce regional biodiversity of stream invertebrates. *Proceedings of the National Academy of Sciences* 110(27), 11039-11043.

Beverly, C., Roberts, A., Doole, G., Park, G. and Bennett, F.R. (2016). Assessing the net benefits of achieving water quality targets using a bio-economic model. *Environmental Modelling and Software* 85, 229-245.

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.

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 18 December 2018).

Christianson, L., Tyndall, J. and Helmers, M. (2013). Financial comparison of seven nitrate reduction strategies for Midwestern agricultural drainage. *Water Resources and Economics* 2-3, 30-56.

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 18 December 2018).

Cooper, R.J., Hama-Aziz, Z., Hiscock, K.M., Lovett, A.A., Dugdale, S.J., Sünnenberg, G., Noble, L., Beamish, J. and Hovesen, P. (2017). Assessing the farm-scale impacts of cover crops and non-inversion tillage regimes on nutrient losses from an arable catchment. *Agriculture, Ecosystems and Environment* 237, 181-193.

Dabney, S.M., Delgado, J.A. and Reeves, D.W. 2001. Using winter cover crops to improve soil and water quality. *Communications in Soil Science and Plant Analysis* 32(7-8):1221-1250.

Deasy, C., Quinton, J.N., Silgram, M., Bailey, A.P., Jackson, B. and Stevens, C.J. (2009). Mitigation options for sediment and phosphorus loss from winter-sown arable crops. *Journal of Environmental Quality* 38, 2121–2130.

de Klein C, et al.; Intergovernmental Panel on Climate Change (2006). IPCC Guidelines for National Greenhouse Gas Inventories. Agriculture, Forestry and Other Land Use, eds Eggleston HS, Buendia L, Miwa K, Ngara T, Tanabe K (Institute for Global Environmental Strategies, Kanagawa, Japan), Vol 4, pp 11.11–11.54.

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.

Dinnes, D.L., Karlen, D.L., Jaynes, D.B., Kaspar, T.C., Hatfield, J.L., Colvin, T.S. and Cambardella, C.C. (2002). Nitrogen management strategies to reduce nitrate leaching in tile-drained Midwestern soils. *Agronomy Journal* 94, 153-171.

Doering, O.C., Diaz-Hermelo, F., Howard, C., Heimlich, R., Hitzhusen, F. and Kazmierczak, R. (1999). Evaluation of the economic costs and benefits of methods for reducing nutrient loads to the Gulf of Mexico: Topic 6 report for the integrated assessment on hypoxia in the Gulf of Mexico, NOAA Coastal Ocean Program Decision Analysis Series No. 20. NOAA Coastal Ocean Program, Silver Spring, MD.

Doole, G. and Pannell, D.J. (2012). Empirical evaluation of nonpoint pollution policies under agent heterogeneity: regulating intensive dairy production in the Waikato region of New Zealand. *Australian Journal of Agricultural and Resource Economics* 56(1), 82-101.

Doole, G., Vigiak, O., Roberts, A.M. and Pannell, D.J. (2013). Cost-effective strategies to mitigate multiple pollutants in an agricultural catchment in North-Central Victoria, Australia. *Australian Journal of Agricultural and Resource Economics* 57(3), 441-460.

Doole, G.J. (2014), 'Least cost greenhouse gas mitigation on New Zealand dairy farms.' *Nutrient Cycling in Agroecosystems* 98, 235–251.

Doole, G.J. (2015). Efficient mitigation of nitrogen leaching in pasture-based dairy systems. *Nutrient Cycling in Agroecosystems* 101, 193–209.

Doole, G.J. and Parangahawewa, U. (2011). Profitability of nitrification inhibitors for abatement of nitrate leaching on a representative dairy farm in the Waikato region of New Zealand. *Water* 3, 1031–1049.

Doole, G.J. and Romera, A.J. (2014). Cost-effective regulation of nonpoint emissions from pastoral agriculture: a stochastic analysis. *Australian Journal of Agricultural and Resource Economics* 58, 471–494.

Fan, A.M. and Steinberg, V.E. (1996). Health implications of nitrate and nitrite in drinking water: An update on methemoglobinemia occurrence and reproductive and developmental toxicity. *Regulatory Toxicology and Pharmacology* 23(1), 35-43.

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.

Fennessy, M.S. and Cronk, J.K. (1997). The effectiveness and restoration potential of riparian ecotones for the management of nonpoint source pollution, particularly nitrate. *Critical Reviews in Environmental Science and Technology*, 27(4), 285-317.

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

Fraser, R.W. (2015). *Applications of Principal-Agent Theory to Agricultural Land Use Policy: Lessons from the European Union*, World Scientific, Imperial College Press, U.K.

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

Hansen, E.M., Munkholm, L.J., Melander, B. and Olesen, J.E. (2010). Can non-inversion tillage and straw retainment reduce N leaching in cereal-based crop rotations? *Soil and Tillage Research* 109, 1-8.

Hansen, L., Delgado, J.A., Ribaudo, M.O. and Crumpton, W.G. (2012). Minimizing costs of reducing agricultural nitrogen loadings: choosing between on- and off-field conservation practices. *Environmental Economics* 3, 98-113.

Hellerstein, D.M. (2017). The US Conservation Reserve Program: The evolution of an enrollment mechanism. *Land Use Policy* 63, 601-610.

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

Hyytiäinen, K., Ahlvik, L., Ahtiainen, H., Artell, J., Huhtala, A. and Dahlbo, K. (2015). Policy goals for improvedwater quality in the Baltic Sea: when do the benefits outweigh the costs? *Environmental and Resource Economics* 61, 217–241.

Janosky, J.S., Young, D.L. and Schillinger, W.F. (2002). Economics of conservation tillage in a wheat–fallow rotation. *Agronomy Journal* 94(3), 527-531.

Jeffrey, S.R., Trautman, D.E. and Unterschultz, J.R. (2017). Canadian agricultural business risk management programs: implications for farmwealth and environmental stewardship. *Canadian Journal of Agricultural Economics* 65, 543–565.

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.

Justes, E. (ed.) (2017). *Cover Crops for Sustainable Farming*. Versailles; Springer.

Kaye, J.P. and Quemada, M. (2017). Using cover crops to mitigate and adapt to climate change. A review. *Agronomy for Sustainable Development* 37(1), Article number 4.

Keeney, D.R. and Hatfield, J.L. (2008). The nitrogen cycle, historical perspective, and current and potential future concerns, Chapter 1 in J.L. Hatfield and R.F. Follett, eds. *Nitrogen in the Environment: Sources, Problems, and Management*. Amsterdam: Elsevier Inc.

Kladivko, E.J., Kaspar, T.C., Jaynes, D.B., Malone, R.W., Singer, J., Morin, X.K. and Searchinger, T. (2014). Cover crops in the upper midwestern United States: Potential adoption and reduction of nitrate leaching in the Mississippi River Basin. Journal of Soil and Water Conservation 69(4), 279-291.

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.

Lambert, D. and Lowenberg-DeBoer, J. (2000). *Precision agriculture profitability review.* Site-specific Management Centre. West Lafayette: Purdue University. https://agriculture.purdue.edu/ssmc/frames/newsoilsx.pdf (accessed 18 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.

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.

MacDonald, K.A., Williams, Y. and Dobson-Hill, B. (2010). Effectiveness of a nitrification inhibitor (DCn) on a coastal Taranaki dairy farm. *Proceedings of the New Zealand Grassland Association* 72, 147-152.

McCasland, M., Trautmann, N.M., Porter, K.S. and Wagenet, R.J. (2012). Nitrate: Health Effects in Drinking Water, Cornell University Cooperative Extension, Pesticide Safety Education Program, http://psep.cce.cornell.edu/facts-slides-self/facts/nit-heef-grw85.aspx (accessed 19 December 2018).

McCully, P. (1996). *Silenced Rivers: The Ecology and Politics of Large Dams*, Zed Books, London. Cited extract from https://www.internationalrivers.org/sedimentation-problems-with-dams (accessed 18 December 2018)

Meals, D.W., Dressing, S.A. and Davenport, T.E. (2010). Lag time in water quality response to best management practices: a review. *Journal of Environmental Quality* 39, 85-96.

Meisinger, J. J., Hargrove, W.L., Mikkelsen, R.L., Williams, J.R. and Benson, V.W. (1991). Effects of cover crops on groundwater quality. pp. 57– 68. In: W.L. Hargrove (ed.) *Cover Crops for Clean Water*. SWCS. Ankeny, IA.

Minnesota Pollution Control Agency (2008). Turbidity: Description, Impact on Water Quality, Sources, Measures - A General Overview, Water Quality/Impaired Waters #3.21, https://www.pca.state.mn.us/sites/default/files/wq-iw3-21.pdf (accessed 18 December 2018).

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.

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.

Oliver, D.P. and Kookana, R.S. (2012). Minimising agricultural pollution to enhance water quality in the Philippines and Australia. *Agricultural Water Management* 106, 1-2.

Olmstead, S.M. (2010). The economics of water quality. *Review of Environmental Economics and Policy* 4(1), 44–62.

Olson, R.V. and Swallow, C.W. (1984). Fate of labeled nitrogen fertilizer applied to winter wheat for five years *Soil Science Society of America Journal* 48, 583-586.

Ongley, E.D. (1996). *Control of Water Pollution from Agriculture*, FAO Irrigation and Drainage Paper 55, FAO, Rome.

Panagopoulos, Y., Gassman, P.W., Arritt, R., Herzmann, D.E., Campbell, T., Jha, M.K., Kling, C., Srinivasan, R., White, M. and Arnold, J.G. (2014). Surface water quality and cropping systems sustainability under a changing climate in the upper Mississippi river basin. *Journal of Soil Water Conservation* 69 (6), 483–494.

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. (2017). Economic perspectives on nitrogen in farming systems: managing trade-offs between production, risk and the environment. *Soil Research* 55, 473-478.

Pannell, D.J., Llewellyn, R.S. and Corbeels, M. (2014). The farm-level economics of conservation agriculture for resource-poor farmers. *Agriculture, Ecosystems and Environment* 187(1), 52-64.

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.

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.

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.

Raun, W.R. and Johnson, G.V. (1999). Improving nitrogen use efficiency for cereal production. *Agronomy Journal* 91: 357-363.

Ribaudo, M.O., Heimlich, R., Claassen, R. and Peters, M. (2001). Least-cost management of nonpoint source pollution: Source reduction versus interception strategies for controlling nitrogen loss in the Mississippi Basin. *Ecological Economics* 37(2), 183-197.

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.

Roberts, A.M. Pannell, D.J. Doole, G. and Vigiak, O. (2012). Agricultural land management strategies to reduce phosphorus loads in the Gippsland Lakes, Australia. *Agricultural Systems* 106(1), 11-22.

Roesch-McNally, G.E., Basche, A.D., Arbuckle, J.G., Tyndall, J.C., Miguez, F.E., Bowman, T. and Clay, R. (2017). The trouble with cover crops: Farmers' experiences with overcoming barriers to adoption. *Renewable Agriculture and Food Systems* 33, 322-333.

Rogowski, A.S. (1990). Estimation of the groundwater pollution potential on an agricultural watershed. *Agricultural Water Management* 18, 209-230.

Roley, S.S., Tank, J.L., Stephen, M.L., Johnson, L.T., Beaulieu, J.J. and Witter, J.D. (2012). Floodplain restoration enhances denitrification and reach-scale nitrogen removal in an agricultural stream. *Ecological Applications* 22 (1), pp. 281-297.

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.

Ruser, R. and Schultz, R. (2015). The effect of nitrification inhibitors on the nitrous oxide (N2O) release from agricultural soils—a review. *Journal of Plant Nutrition and Soil Science* 178, 171-188.

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 18 December 2018).

Sharpley, A.N. and Smith, S.J. (1994). Wheat tillage and water quality in the Southern Plains. *Soil and Tillage Research* 30, 33-48

Shcherbak, I., Millar, N. and Robertson, G.P. (2014). Global metaanalysis of the nonlinear response of soil nitrous oxide (N2O) emissions to fertilizer nitrogen. *Proceedings of the National Academy of Sciences* 111(25), 9199-9204.

Shortle, J.S. and Horan, R.D. (2001). The economics of nonpoint pollution control. *Journal of Economic Surveys* 15(3), 255-289.

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.

Soane, B.D., Ball, B.C., Arvidsson, J., Basch, G., Moreno, F. and Roger-Estrade, J. (2011). No-till in northern, western and south-western Europe: A review of problems and opportunities for crop production and the environment. *Soil and Tillage Research* 118, 66-87.

Stoneham, G., Chaudhri, V., Ha, A. and Strappazzon, L. (2003). Auctions for conservation contracts: an empirical examination of Victoria's bushtender trial. *Australian Journal of Agricultural and Resource Economics* 47, 477-500.

Stonehouse, D.P. (1997). Socio-economics of alternative tillage systems. *Soil and Tillage Research* 43(1-2), 109-130.

Sudarshan, A. (2017). Nudges in the marketplace: The response of household electricity consumption to information and monetary incentives. *Journal of Economic Behavior and Organization* 134, 320-335.

Suter, J.F., Vossler, C.A., Poe, G.L. and Segerson, K. (2008). Experiments on damage-based ambient taxes for nonpoint source polluters. *American Journal of Agricultural Economics* 90(1), 86-102.

Swinton, S.M. and Lowenberg-DeBoer, J. (1998). Evaluating the profitability of site-specific farming. *Journal of Production Agriculture* 11(4), 439-446.

Thomas, B.W., Hao, X., Larney, F.J., Goyer, C., Chantigny, M.H. and Charles, A. (2017). Non-legume cover crops can increase non-growing season nitrous oxide emissions. *Soil Science Society of America Journal* 81(1), 189-199.

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.

Tonitto, C., David, M.B. and Drinkwater, L.E. (2006). Replacing bare fallows with cover crops in fertilizer-intensive cropping systems: A meta-analysis of crop yield and N dynamics. *Agriculture, Ecosystems and Environment* 112, 58–72.

USDA-NRC (Natural Resources Conservation Service). *National Soil Survey Handbook,* title 430-VI. https://www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/ref/?cid=nrcs142p2_054242 (accessed 18 December 2018).

Van Grinsven, H.J.M., Holland, M., Jacobsen, B.H., Klimont, Z., Sutton, M.A. and Jaap Willems, W. (2013). Costs and benefits of nitrogen for Europe and implications for mitigation. *Environmental Science and Technology* 47(8), 3571-3579.

Wade, T., Claassen, R. and Wallander, S. (2015). *Conservation-Practice Adoption Rates Vary Widely by Crop and Region*, Economic Research Service, Economic Information Bulletin Number 147. Washington, D.C.: United States Department of Agriculture.

Walmart (2018) 2018 Global responsibility report summary. https://corporate.walmart.com/2018grr/ (accessed 23 November 2018).

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.

Woolfolk, C.W., Raun, W.R., Johnson, G.V., Thomason, W.E., Mullen, R.W., Wynn, K.J. and Freeman, K.W. (2002). Influence of late-season foliar nitrogen applications on yield and grain nitrogen in winter wheat. *Agronomy Journal* 94, 429-434.

Wortmann, C., Al-Kaisi, M., Helmers, M., Sawyer, J., Devlin, D., Barden, C., Scharf, P., Ferguson, R., Kranz, W., Shapiro, C., Spalding, R., Tarkalson, D., Holz, J., Francis, D. and Schepers, J.

(2013). Agricultural Nitrogen Management for Water Quality Protection in the Midwest. Heartland, Regional Water Coordination Initiative, University of Nebraska, Lincoln, NE.

Zhang, T.Q., Tan, C.S., Zheng, Z.M., Welacky, T. and Wang, Y.T. (2017). Drainage water management combined with cover crop enhances reduction of soil phosphorus loss. *Science of the Total Environment* 586, 362-371.

Table 1. Summary of key practices for reducing water pollution from agriculture

Practice	Pollutants addressed	Surface water or groundwater	Within field or off-site	Key references	Interactions with greenhouse-gas emissions or sequestration
Fertilizer practices (fertilizer type, rate, placement (depth), timing)	Nitrogen, phosphorus	Both	Within field	Leslie et al. (2017) (fertilizer rates) Pannell (2006) (flat payoff functions)	At normal rates, around one percent of N fertilizer applied is converted to N_2O by soil microbes (de Klein et al. 2006). N_2O is extremely potent as a greenhouse gas – 300 times more potent than CO_2. At higher N rates, emissions are likely to increase exponentially (Shcherbak et al. 2014).
Variable rate technology (VRT) for N or P fertilizer	Nutrients	Both	Within field	Watkins et al. (1998) Babcock and Pautsch (1998)	To the extent that VRT results in overall reductions in fertilizer application, it would result in a reduction in N_2O emissions. The likely reduction is modest, at best.
Zero-till and conservation agriculture	Mainly phosphorus	Surface water	Within field	Cooper et al. (2017)	As well as reducing soil erosion, zero-till can result in higher carbon sequestration in the soil.
Cover crops	Sediment, nutrients, pesticides in runoff Nutrients (N) in drainage water	Surface water Groundwater	Within field	Aronsson et al. (2016) (N loss) Cooper et al. (2017) (N & P loss) Zhang et al. (2017) (P loss)	Legume cover crops can sequester carbon and reduce N fertilizer use (Kaye and Quemada, 2017). Non-legume cover crops can increase nitrous oxide (N_2O) emissions (Thomas et al., 2017). N_2O is a potent greenhouse gas.

Table 1. Summary of key practices for reducing water pollution from agriculture (continued).

Practice	Pollutants addressed	Surface water or groundwater	Within field or off-site	Key references	Interactions with greenhouse-gas emissions or sequestration
Nitrification inhibitors	Nitrogen	Groundwater	Within field	Ruser and Schultz (2015)	Nitrification inhibitors reduce emissions of N_2O.
Land retirement	Nitrogen, phosphorus and sediment	Both	Within field		Carbon sequestration in the vegetation that grows on the retired land.
Buffer strips	Nitrogen	Surface water	Off-site	Fennessy and Cronk (1997)	A modest amount of carbon sequestration in the buffer-strip vegetation.
Flood-plain restoration (two-stage ditches)	Nitrogen (minor effect)	Surface water Drainage water	Off-site	Roley et al. (2016)	Minimal
Wetlands	Nitrogen Very small effect on phosphorus	Surface water Drainage water	Off-site	Kovacic et al. (2000)	Minimal
Bioreactors	Nitrogen	Drainage water	Off site	Christianson (2016)	Minimal

Source: Compiled by authors.

Table 2. Assessment of key adoption factors for each practice for reducing water pollution from agriculture

Practice	Ease of trialing	Up-front costs	Profitability	Time lags	Risk and uncertainty	Complexity
Fertilizer practices	7	6	3	7	3	6
Variable rate technology	3	2	2	7	4	3
Zero-till and conservation agriculture	2	3	4-6	2	6	2
Cover crops	4	4	2	4	4	3
Nitrification inhibitors	2	2	1	4	4	3
Land retirement	3	4	1	4	4	4
Buffer strips	2	2	1	4	4	4
Flood-plain restoration	2	1	1	4	4	3
Wetlands	2	1	1	4	4	3
Bioreactors	3	1	1	4	4	3

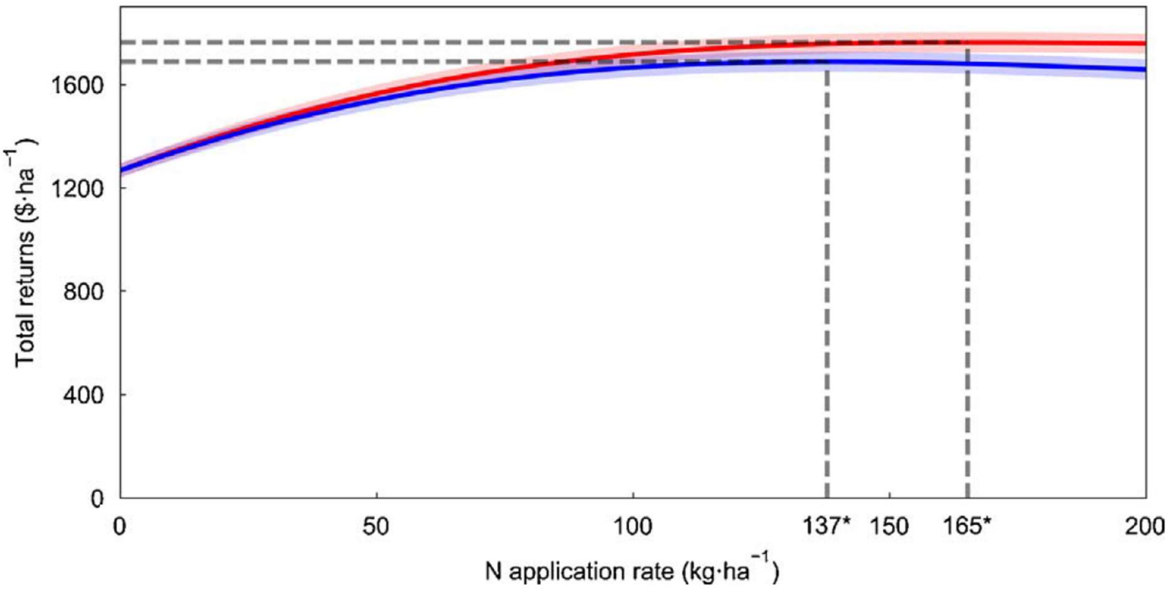
Notes: Ratings based on author judgment. 1 = Highly unfavourable for adoption, 4 = neutral, 7 = Highly favourable for adoption.

Table 3. Policies used to address water quality impacts from agriculture in the United States

Instrument type	US 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: Based on Shortle and Uetake (2015).

Figure 1. Private versus public net returns to nitrogen applied to Minnesota soybeans



Source: Gourevitch et al. (2018)

Notes: Private net returns (red) and public net returns (blue – with the social cost of nitrogen deducted) for nitrogen fertilizer application to corn after soybeans in Minnesota, assuming a social cost of $0.50 per kg N.