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ENVIRONMENTAL CO-BENEFITS AND STACKING IN ENVIRONMENTAL MARKETS

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NOTE BY THE SECRETARIAT

This paper has been prepared by Jussi Lankoski, with valuable contributions by Markku Ollikainen, University of Helsinki, Finland (the theoretical framework for stacking in Chapter 3 and Annex A) and Marcel Aillery and Liz Marshall, Economic Research Service, United States Department of Agriculture, (EPIC model simulations and model region description in Chapter 4).

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EXECUTIVE SUMMARY

Establishing markets for ecosystem services has received increasing policy attention in recent years. Ecosystem markets have been used to leverage private investment in conservation practices, potentially reducing government outlays while achieving environmental objectives at lower costs. The environmental credit markets could provide incentives for additional environmental improvements through the supply of environmental co-benefits. Indeed, many agri-environmental practices have simultaneous effects on multiple environmental goods. These environmental co-benefits result from the jointness between a given environmental practice and multiple environmental outputs. It has been debated in this context whether a single agri-environmental practice should be able to earn credits from multiple environmental markets. Such credit stacking would allow a farmer to receive multiple payments for a single management practice on the same field parcel.

The stacking of environmental credits has both advantages and disadvantages. On the one hand, allowing stacking may increase farmers' participation in government conservation programmes and environmental markets, since multiple payment and credit revenues are more likely to cover farmers' opportunity costs of environmental practice adoption. Moreover, allowing stacking may provide incentives for a more optimal combination of various environmental outputs and encourages higher quality environmental practices that may not be profitable with a single payment or credit revenue stream. On the other hand, allowing stacking may also complicate interpretations of additionality in the context of multiple environmental markets. If the credit revenue from the primary environmental market already compensates adoption costs of the practice then a question arises whether the environmental co-benefits of the given practice can be considered environmentally additional in secondary markets since due to jointness they are already provided through incentives created by the primary ecosystem credit market.

The key policy issues addressed in this paper are: (i) implications of stacking for environmental additionality and integrity of environmental markets, (ii) impacts of stacking on market participation, offset supply and equilibrium offset prices in interlinked offset markets, and (iii) effect of transaction costs on market participation. Analysis focused on carbon and water quality offset and credit markets. The context and starting point for the theoretical and empirical analysis is the farmers' choice of alternative soil carbon sequestration practices, such as no-till and green fallowing, and farmers' incentives to participate voluntarily in carbon offset markets without the possibility to sell water quality credits. Environmental credit stacking is then incorporated in the analysis and farmers are permitted to sell both carbon and water quality credits.

Farmers' incentives to participate in environmental credit markets are not solely based on opportunity costs of practice adoption and revenue from environmental credit markets but also on transaction costs related to market participation. Thus, credit revenue from the markets needs to compensate both opportunity costs and transaction costs of their participation. A theoretical analysis shows that credit stacking increases farmers' participation in carbon offset markets, and through increased participation provides additionality in environmental service provision. Credit stacking also provides incentives for adoption of more environmentally effective practices due to credit revenue from water quality offsets, and

thus provides additional environmental services relative to those practices that are adopted through incentives provided by a carbon offset market alone. Theoretical analysis further shows that environmental markets are interlinked so that credit price changes in one market will shift credit supply in another market, thus affecting equilibrium prices.

Empirical application of the model is based on data estimates for the U.S. Corn Belt region. Three model areas, drawn from the eastern, central and western portions of the Corn Belt region, were selected for use in the empirical analysis. The study captures production heterogeneity through regional differences in climate and land resources. Model cropping systems were identified based on two representative crop rotations for the Corn Belt region — Continuous Corn and Corn-Soybean (alternate years) — and three tillage systems — conventional, reduced and no-till. Model regions are further differentiated by Highly Erodible Lands (HEL) and Non-Highly Erodible Lands (NonHEL). Crop yield, input use and environmental parameters are generated by the Environmental Productivity and Integrated Climate (EPIC) model. Environmental parameters generated by EPIC include: (i) soil erosion, (ii) nitrogen runoff, (iii) phosphorus runoff and (iv) changes in soil carbon. EPIC parameters developed for production alternatives are applied to model 36 different cropping systems.

In the empirical application, the baseline scenario without carbon and water quality offset markets is analysed first. Corn yields and farmers' profits vary significantly between the regions, based on parameters from the EPIC simulation results. CO_2-eq emissions from cultivation practices and fertilizer application are smaller than soil carbon sequestration in each Region and thus net GHG emissions are negative. Nitrogen runoff varies between regions and especially between HEL and NonHEL lands. Both erosion and phosphorus runoff are significantly higher for HEL lands relative to NonHEL lands. As regards the social value of environmental effects, agriculture's net climate impact is positive and, relative to other environmental effects, combined nutrient runoff damage from nitrogen and phosphorus runoff is significant in each Region. Agriculture's overall net environmental impact is negative in each Region, but the profitability of production makes ex-post social welfare (profits less net environmental damage) positive across Regions.

As regards carbon offset markets and agricultural supply of carbon offsets in the studied regions, the following conclusions can be drawn. First, provision of CO_2-eq offsets through reduction of nitrogen application is not profitable without water quality offsets due to the relatively small impact of applied nitrogen on N_2O emissions and thus a small amount of CO_2-eq offsets produced. Similarly due to significant profit foregone, the establishment of green set-asides is not profitable without water quality offsets. A conversion from conventional tillage and reduced tillage to no-till is profitable in some cases although current low carbon offset prices and transaction costs have a significant negative impact on the number of participating parcels. Overall, carbon offset markets with current offset prices do not necessarily incentivize farmers to participate in environmental markets without the possibility of stacking water quality offsets.

When farmers are allowed to stack water quality credits the profitability of carbon sequestration practices increases. Reduced nitrogen application levels becomes a profitable option and 21% of field parcels - representing 4.6 million acres- participate in the market with water quality offset prices at current levels of USD 3/lb for N and USD 4/lb for P. For an offset price range of USD 1-4/lb for N and USD 2-4/lb for P, the number of participating parcels varies between 9% and 38% of eligible acreage. Also the establishment of green set-aside and streamside buffer strips becomes profitable in the lower productivity and highly erodible lands with current offset prices. High participation rates among farmers may, however, result in an oversupply of nutrient credits and as a consequence equilibrium credit prices and farmers' credit revenue would decrease.

To conclude, both the theoretical and the empirical analysis show that allowing stacking of water quality credits provides additional environmental services through increased participation of farmers in carbon offset markets and through increased adoption of environmental practices that are effective in promoting both carbon offsets and nutrient credits. The agri-environmental practices analysed in this study are widely used measures in various OECD countries and although their opportunity costs and effectiveness in promoting supply of carbon offsets and nutrient credits varies over space, some general conclusions can be drawn. First, current CO_2-eq offset prices do not necessarily compensate profit foregone of adopting these practices, and thus allowing stacking of water quality credits or government incentive payments makes adoption more profitable. Stacking of government agri-environmental payments with environmental credits increases farmers' participation in both government conservation programmes and environmental markets and can provide additional income for farmers. Secondly, if environmental markets are local and small, with limited demand for credits, then stacking may lead to oversupply of credits, resulting in decreased equilibrium credit prices. As a consequence, environmental practices that used to be profitable to adopt with credits from one market may require sale of credits to several markets in order to cover the adoption costs of the practice.

1. Introduction

Ecosystems provide various services and goods to society, including *provisioning services* (e.g. related to the production of food, fiber, and fuels), *regulating services* (air, climate, water, pest, and disease regulation), *supporting services* (nutrient and water cycling) and *cultural services* (aesthetic values and recreation). Government incentive payments and payments for ecosystem services (PES) have been used by several countries to address the under-provision of various ecosystem services.

There has been increasing interest in establishing markets for environmental goods and services. The interest in environmental markets has been driven by a general shift from command-and-control policies toward market-based instruments in environmental protection, an increased capacity to value environmental goods and services, and an increased demand for environmental services and goods by governments, private firms, and consumers. Hence the creation of markets may harness additional resources and involve more stakeholders in environmental protection (Marshall and Selman, 2011).

In many environmental markets, such as carbon trading and water-quality trading, government regulation drives demand for offsets: firms in the regulated sector purchase allowances from other firms in the regulated sector or offsets from unregulated sectors to comply with the regulation and to minimise their compliance costs. In these cases, markets do not necessarily provide incentives for additional environmental improvements above what is required by regulation, but ensure that environmental objectives are met at lower costs.

Environmental offset and credit markets are increasingly used to address greenhouse-gas emissions and water quality in some OECD countries, including Australia, Canada, and the United States. For example, in the United States 51 water quality trading programmes — active or under development— has been identified and three major regional carbon trading programmes exist or have been proposed (Marshall and Weinberg, 2012).

The establishment of environmental credit markets could provide incentives for additional environmental improvements through the supply of environmental co-benefits. Many agri-environmental practices affect several environmental goods and services at the same time. For example, the adoption of no-till cultivation usually reduces soil erosion, nitrogen runoff and particulate phosphorus runoff (but may increase dissolved phosphorus runoff), and conversion of cropland to perennial grasses increases soil carbon sequestration while improving wildlife habitat and water quality.

These environmental co-benefits result from the jointness between a given environmental practice and multiple environmental outputs. It has been debated in this context whether a single agri-environmental practice should be allowed to earn credits from multiple environmental markets. This credit stacking occurs when a farmer receives multiple credit revenues for a single management practice on the same field parcel. Stacking is also said to occur when a farmer receives both a government incentive payment for an environmental practice adoption and co-benefit credits from environmental markets.

The stacking of environmental credits has both advantages and disadvantages. On the one hand, it increases farmers' participation in under-subscribed government conservation programmes and environmental markets, since multiple payment and credit revenues are more likely to cover farmers' opportunity costs of adopting a particular environmental practice. Moreover, allowing stacking may provide incentives for a more optimal combination of various environmental outputs and encourage higher quality multi-benefit environmental practices that may not be profitable with a single payment or credit revenue stream. On the other hand, allowing stacking may also complicate interpretations of additionality and baselines in the context of multiple environmental markets. If credit revenue from the primary environmental market already covers the adoption costs of the practice, the question arises whether

environmental co-benefits of the given practice can be considered additional in secondary markets since due to jointness they are already provided through incentives created by the primary environmental credit market.

The key policy issues addressed in this paper are: (i) implications of stacking on environmental additionality and the integrity of environmental markets, (ii) impacts of stacking on market participation, credit supply and equilibrium credit prices in interlinked environmental markets, and (iii) the effects of transaction costs on market participation.

The focus of the paper is on carbon and water-quality offset and credit markets. The context and starting point for the theoretical and empirical analysis is farmers' choice of alternative soil carbon sequestration practices and farmers' incentives to participate voluntarily in carbon-offset markets without the possibility to sell water-quality credits. Then the paper considers what happens when environmental credit stacking is allowed and farmers can also sell water quality credits. The aim is to provide a theory-based analysis of additionality and credit stacking in the context of multiple environmental markets, and to provide an empirical application of the theoretical framework based on US data estimates for the Corn Belt.

The paper is structured as follows. A review of literature is provided in Chapter 2. The theoretical model is developed in Chapter 3. In Chapter 4, the data and the parametric model are presented. Results are provided in Chapter 5 while Chapter 6 concludes.

2. Review of the literature

2.1. Key issues and definitions

Ecosystem services are benefits that arise from the regulating, supporting, and provisioning services supplied by ecosystems (MEA, 2005). Payments for ecosystem services (PES) have been proposed to address market failures related to ecosystem services by providing incentives to enhance their provision. According to Wunder (2007), a PES is a voluntary transaction where a well-defined environmental service is bought (sold) by at least one buyer (provider) and the provider secures service provision under certain conditions.

Environmental credit stacking, or briefly ***stacking***, occurs when a single agri-environmental practice produces several environmental outputs and earns credits from multiple environmental markets (Gillenwater, 2012c; Marshall and Selman, 2011; Cooley and Olander 2011).

Cooley and Olander (2011) distinguish between three forms of stacking: horizontal, vertical and temporal stacking. Horizontal stacking occurs when a project performs more than one distinct management practice on separate parcels and the project participant receives a single payment for each practice. As the definition suggests, this approach is more about grouping of offsets and is usually not considered a problem. Vertical stacking is the most relevant form and occurs when a project receives multiple payments for a single management activity on the same field or forest parcel. Vertical stacking entails establishing more than one credit type on the same parcel. Potential problems here involve double payments and additionality. Temporal stacking differs from vertical stacking only in the sense that payments are disbursed over time. For instance, the farmer receives carbon credits from a shift to no-till cultivation, and later, when water quality markets are developed, the farmer obtains water quality credits as well.

As regards potential advantages of stacking, the following have been raised. On a positive note stacking can spur participation in environmental markets and environmental programmes where a single market or programme may not pay farmers enough to make an environmental practice profitable (Cooley and Olander, 2011). Multiple programmes involving multiple payment streams are more likely to cover

farmers' opportunity costs of environmental practice adoption. Hence, without stacking some cost-effective environmental practices that contribute to several environmental outputs will not be implemented. Moreover, stacking may provide incentives for a more optimal combination of various environmental outputs that reflects complementarities and trade-offs between these outputs (Gillenwater, 2012c). Multiple payment streams also help to encourage higher quality multi-benefit environmental practices that may not be profitable with a single payment stream, such as restoring wetlands instead of planting vegetative buffer strips along watercourses (Cooley and Olander, 2011).

Box 1. Survey on credit and payment stacking

Fox et al. (2011) conducted a national survey on credit stacking in the US. Approximately 1 500 individuals involved in environmental markets were contacted, and replies were received from 309 individuals (20% response rate). Respondents represented the following stakeholder categories: credit sellers (117), researchers (89), policymakers (82), credit buyers (17), and credit exchanges (4). As regards the definition of stacking there was a strong consensus that stacking means establishing more than one credit type on spatially overlapping areas (83.5% of the respondents selected this definition). The selling of credits in different markets raised concerns regarding additionality and double counting. However interest in the concept of stacking was high as 73.6% of the respondents stated that they are either involved in credit stacking, or are interested in getting involved in the future. The survey showed that there is a fairly even interest across various environmental markets including species banking, wetland banking, water quality trading and carbon trading. Buyers of environmental credits are extremely interested in engaging agriculture as farmers could supply the markets with reasonably priced credits if these are jointly produced with conservation practices that were originally government-funded (so-called payment stacking).

Source: Fox et al. (2011).

However, allowing stacking also raises concerns related to additionality, that is whether offsets and credits are generated from practices that would have occurred without payment. Hence, questions have been raised as to whether the environmental improvements provided by the offsets and credits are enough to fully mitigate all impacts they allow and whether the offsets and credits are indeed truly additional.

A prerequisite for earning credits is to assess whether the environmental benefits provided by a given practice are ***additional*** relative to a ***baseline***. Gillenwater (2012a) defines additionality and a baseline as follows:

- "**Additionality** is the property of an activity being *additional*. A proposed activity is *additional* if the recognised policy interventions are deemed to be causing the activity to take place. The occurrence of additionality is determined by assessing whether a proposed activity is distinct from its baseline (see below).

- A **baseline** is a prediction of the quantified amount of an input to or output from an activity resulting from the expected future behaviour of the actors proposing, and affected by, the proposed activity in the absence of one or more policy interventions, holding other factors constant (*ceteris paribus*). The conditions of a baseline scenario are described in a baseline scenario."

Additionality is a key criterion in many environmental markets. Additionality ensures that an environmental practice compensates (offsets) the allowed impact and thus ensures environmental integrity of market mechanisms (Bennett, 2010; Cooley and Olander, 2011).

As regards additionality, a project is eligible for a payment only if the offsets generated come from practices that would not have occurred in the absence of the payment (Gillenwater, 2012c). Thus, offsets must go beyond the current or projected business-as-usual emissions. From this standpoint selling offsets is not additional if these offsets are provided from management practices that are privately optimal for the

farmer. Precise determination of baseline emissions and emissions reductions from adopted conservation practices at each individual farm and field parcel can be very costly due to the heterogeneous nature of agriculture. Thus, in practice less costly approaches to estimate baseline emissions, based on observed technologies and practices, such as a baseline technology or practice, are typically used for determining a baseline. The baseline technology or practice acts as a proxy for actual or expected emissions and may not accurately present baseline emissions on a particular field parcel (Marshall and Selman, 2011). Models and other tools, such as the NRCS/EPA Nitrogen Trading Tool (NTT), can be used to estimate baseline emissions on the basis of geographic, agronomic, and land-use information (Ribaudo et al., 2010).

Bennett (2010) discusses alternative ways to ensure additionality in PES schemes and ecosystem markets. In a *project-specific assessment*, additionality is evaluated on a project by project basis and the regulating entity is generally allowed discretion when granting credits. *Standardised additionality assessments* use general criteria in the evaluation of additionality.[1] They are likely to perform well when the difference between business-as-usual and the environmental practice is clear, and relative to a project-specific assessment they may have lower administrative costs (Bennett, 2010).

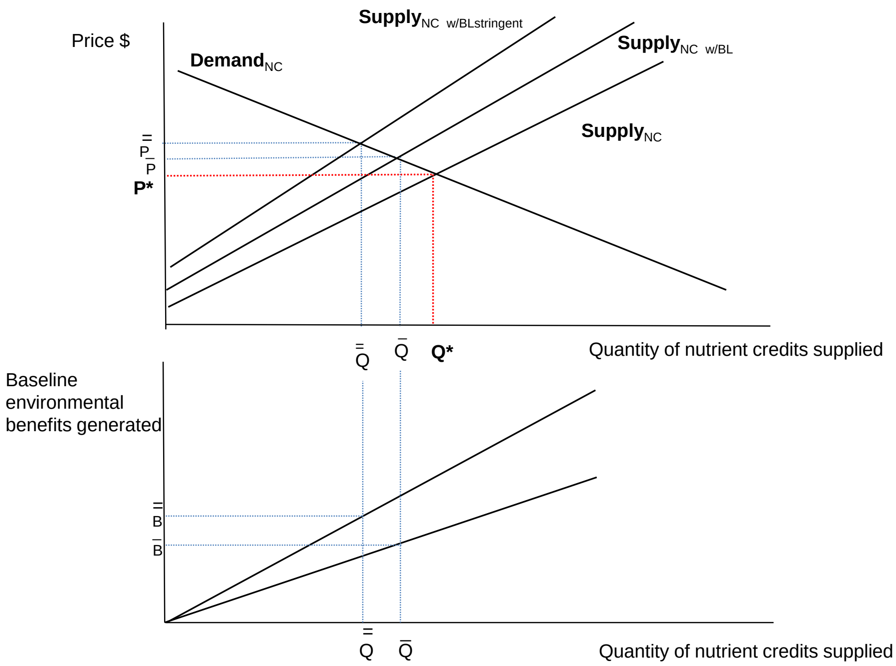
Due to the inherent uncertainty related to various environmental outputs of a given environmental practice *trading ratios* can be used to discount environmental benefits and thus the environmental credits provided to a farmer. Trading ratios enable the use of conservative estimates of environmental benefits, and thus help to secure the environmental performance of markets. On the other hand, discount rates that are too high may reduce farmers' willingness to participate in environmental markets and thus will have implications for the economic and environmental performance of the markets (Bennett, 2010).

Another approach for safeguarding the environmental integrity of offset markets is to require farmers to achieve a certain environmental baseline before participating in offset markets (Marshall and Selman, 2011). Figure 1 shows that an environmental performance requirements for entering the offset market shifts the supply curve of nutrient credits (NC) and rotates it up due to the baseline requirement (NC w/BL). More stringent baseline requirements (NC w/BL stringent) further shifts the nutrient credit supply curve and results in fewer credits supplied and increased equilibrium credit prices. The lower part of Figure 1 shows how the baseline requirements and stringent baseline requirements reduce the quantity of nutrient credits supplied while increasing additional environmental benefits (Marshall and Selman, 2011).

Gillenwater (2012b) develops a conceptual framework for a standardised assessment of additionality and baselines for environmental credits and offsets. The framework builds on explicitly recognised policy interventions, theories of behaviour, and objective models. In order to have more objective standardised approaches for the assessment of baselines and additionality it is necessary: (i) to define the type of policy intervention and specify treatment variables that represent it, (ii) to make assumptions regarding the theory of behaviour (pure rationality, bounded rationality, and altruism), and (iii) on the basis of (i) and (ii), to develop a decision model (causal model) for assessing baselines and additionality.

1 Standardised assessments include, for example, legal, time, financial and technology criteria. Thus, a proposed environmental practice fails the additionality test if: (i) it is required by law, (ii) it has already been adopted, (iii) it could be implemented without a payment stream from credits, or (iv) it employs common practice or business-as-usual technology (Bennett, 2010).

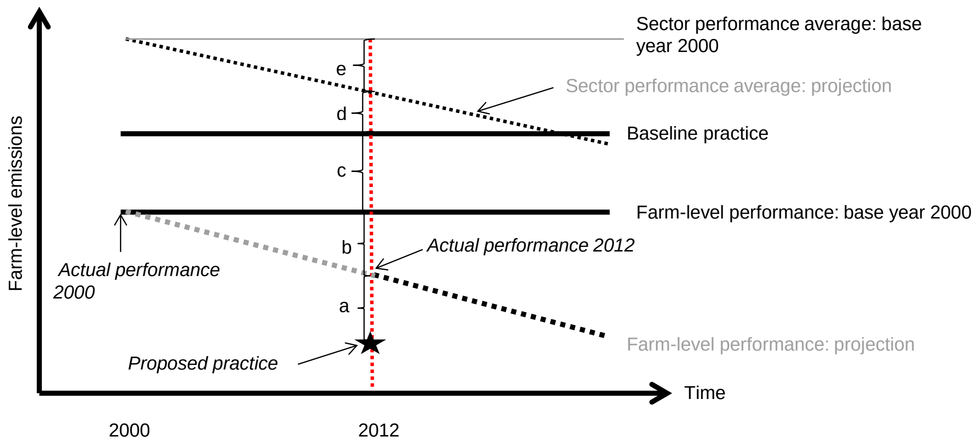
Figure 1. Supply and demand for nutrient credits with a baseline environmental performance requirement to participate in offset market



Source: Marshall and Selman (2011).

Marshall and Weinberg (2012) analyses potential tradeoffs between the precision of environmental baselines and associated costs with defining, measuring, and verifying of environmental baselines across heterogeneous landscapes. A tradeoff exists between the implementation costs of verifying additionality and the potential environmental performance loss if additionality is not properly verified. They analyse the following baseline standards: (i) a baseline technology or practice, (ii) a baseline year, (iii) a static or a dynamic performance baseline, and (iv) the farm level or the sector level baseline. In the case of a baseline technology or practice, environmental improvements beyond the environmental performance of the baseline technology or practice are considered creditable while in the case of a baseline year improvements relative to performance in a baseline year are considered creditable. A dynamic baseline incorporates projected changes in farming practices, technology development and adoption, and resulting environmental performance. Farm-level baselines compare a farm's environmental performance relative to a baseline determined on the basis of the farm's historical, current or projected production patterns while a sector baseline compares a farm's environmental performance to that of the sector average (Marshall and Weinberg, 2012).

Figure 2. Environmental credits awarded under different baseline scenarios in the case of a farmer who has already adopted environmental practices



Baseline emissions level selected	Amount of carbon credits awarded
Farm-level performance: base year 2000	a + b
Farm-level performance: projection	a
Baseline practice	a + b + c
Sector performance average: projection	a + b + c + d
Sector performance average: base year 2000	a + b + c + d + e

Source: Marshall and Weinberg (2012).

Figure 2 describes how the number of credits awarded to a farmer depends on the definition of the environmental baseline (Marshall and Weinberg, 2012). In this example, the farmer is an early adopter of good environmental practices and has environmental performance that exceeds the sector average. Depending on the definition of the baseline, the environmental credits awarded to the farmer could vary from (a) to (a+b+c+d+e), representing a move from the actual environmental performance in 2012 to a level associated with a new environmental practice adoption (red star). If sector-level average is used as a baseline, the environmental benefits of practice adoption would be overestimated and “non-additional” credits (b+c+d+e) would be awarded to the farmer. In an opposite case (not shown in Figure 2) where a farmer’s environmental performance is below the sector average, using the sector-level average would underestimate the environmental benefits of a given farmer’s practice adoption and thus the farmer would receive fewer credits than should be awarded from an additionality viewpoint.

Financial additionality requires that adoption of an environmental practice is not financially viable without a credit payment. In the context of multiple environmental markets farmers can sell credits into multiple markets, but they need to demonstrate that an environmental practice adoption is not viable without the combined credit revenue from the multiple markets (Marshall and Selman, 2011). For example, if a farmer establishes a green set-aside and a carbon payment fully compensates the farmer’s foregone profit, this practice adoption would fail the financial additionality test in water quality markets, and thus water quality co-benefits of the green set-aside would be considered non-additional.

Bundling refers to a case where the adoption of an environmental practice receives a single payment for the provision of multiple environmental outputs. Through bundling, multiple environmental goods can be traded as a single environmental credit that reflects environmental improvements in multiple environmental outputs. In contrast, unbundling means that multiple environmental outputs of an environmental practice are divisible and each of them could earn specific credits. Thus, unbundling is a necessary condition for stacking as there will be nothing to stack if multiple environmental outputs are bundled in a single environmental credit (Gillenwater, 2012c).

Environmental markets can be developed for bundled environmental goods or services. Bundling, however, requires the use of environmental indices to describe the effects of agri-environmental practices on various environmental outputs, or, at least a good proxy for these effects, such as an acre of wetland (Cooley and Olander, 2011; Marshall and Selman, 2011). One way to organise markets for bundled environmental goods is to establish an aggregator institution that buys bundled credits from farmers and then unbundles and sells them to different individual markets (Binning et al., 2002; Marshall and Selman, 2011).

Cooley and Olander (2011) analyse different forms of stacking and their impacts on net environmental outcome. Government incentive payments (or PES) stacked with other incentive payments (or PES) will not create negative net environmental outcomes, since these payments do not allow adverse impacts elsewhere. Allowing producers to sell environmental goods and services that have been partially paid for with government funds can distort the offset market, however, which can affect the ability of the market to actually achieve cost-effective reductions. Stacking of mitigation credits with other environmental credits, however, may create negative environmental outcomes due to double counting. For example, if a wetland establishment receives both bundled wetland credits (including water quality) and single water quality credits, then one mitigation action would allow two separate impacts on water quality of which only one would be offset, leading to a net loss of environmental services (Cooley and Olander, 2011).

Participants in environmental markets include sellers (e.g. farmers), buyers (e.g. wastewater treatment plants) and regulators. Regulatory agencies are responsible for making decisions about which practices are eligible for generating credits, establishing record keeping and reporting tools, determining and enforcing trading ratios and conducting verifications (Lal et al., 2009). Trading facilitators ease exchange of credits and include brokers, aggregators and central exchanges; their role is to bring credit sellers and buyers together under the rules set by regulators. State agencies, local conservation districts, NGOs, private firms and entrepreneurs can act as trading facilitator. Brokers help in drafting a trade agreement between the seller and the buyer while aggregators usually collect credits from several sellers and sell them in bulk to the buyers and thus they have trade contract with both the seller and the buyer. Similarly central exchanges purchase credits from multiple sellers and sell them to different buyers (Lal et al., 2009).

If a buyer of environmental goods and services were to enter into a contract to pay a farmer to use specified practices, the farmer’s compliance with the terms of the contract would have to be monitored. When the practices are easily observable, such as changes in land cover, monitoring can be done at relatively low cost using remote sensing technology (CAST, 2000). However, if environmental service credit contracts specify other changes in management, such as tillage practices and reduced use of fertilizer and pesticides, the cost of monitoring compliance may be substantially higher.

Box 2. Principles of double funding

IEEP (2012) analyses the discussions related to double funding in the context of CAP 2014-2020 policy proposals. Double funding refers to a case where the same compliance costs for the same environmental action are funded twice from public funds, and is not permitted under the rules for public expenditure in the EU. As regards CAP 2014-2020 policy proposals, the double funding debate relates to the interrelationship between greening measures in Pillar 1 and the agri-environment-climate measure (AECM) in Pillar 2. The Commission's original proposals were clear that AECM measures need to be additional to greening measures and thus, greening measures establish a new baseline for the AECM in Pillar 2. Indeed the Commission's CAP-REFORM Fiche No 17 clearly states that cross-compliance and the greening measures of Pillar 1 will form a new and higher baseline for the more targeted AECM measures in Pillar 2 and that the AECM payments shall not cover any compliance costs of Pillar 1 greening measures.

According to the Commission proposal, farmers that comply with requirements of organic farming legislation will be entitled to the greening payment under Pillar 1. This is not considered double funding, since the intervention logic or rationale is different for Pillar 1 (environmental improvement) and Pillar 2 (compensation for loss of income and extra costs incurred from switching to organic production). Thus, the adopted practice and environmental outcome may be the same, but if the payment rationale is different, then this is not considered to raise the double funding problem.

The Commission's proposals received criticism from both Member States and the European Parliament as well as from various stakeholders, and a number of counterproposals were put forward. For example, farmers' unions (such as Copa-Coega) proposed that farmers who comply with AECM in Pillar 2 should be *de facto* eligible for Pillar 1 green payments on top of their agri-environment payments. In the political agreement on a new direction for common agricultural policy a technical solution was adopted to address double funding: to apply a reduction rate to Pillar 2 payments for certain measures that are not deemed to be environmentally additional to the greening measures.

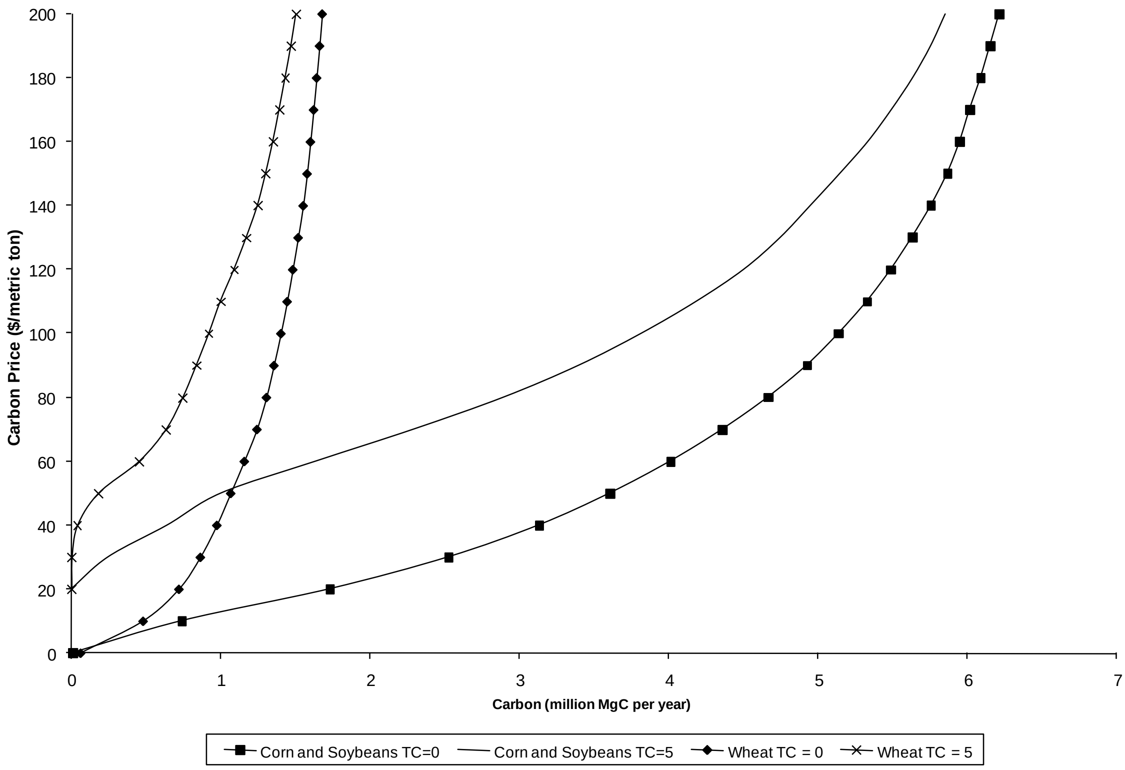
The European Commission adopted on March 11, 2014 the so-called "delegated acts" for the new CAP, clarifying how the CAP reform is to be implemented on farms across the EU. According to the delegated act on rural development Pillar 2 payments must cover only the additional costs and/or income foregone related to the commitments that go beyond the relevant obligatory practices set out in the Pillar 1 greening provisions. For those measures for which it is not possible to identify accurately which elements go beyond the greening provisions the agri-environment-climate payment must be reduced by a lump sum corresponding to the part of the Pillar 1 greening payment.

Sources: IEEP (2012) and AgraEurope (2014).

For example, if carbon offset contracts are carried out through private-market trading programs, various transaction costs will have to be borne by project participants (McCarl, 2002; van Kooten, et al, 2002; Mooney et al., 2004a, b). Transaction costs include normal financial transaction expenses (legal and broker fees, etc.), as well as costs associated with verifying contract compliance. Very few reliable data are available to estimate transaction costs for implementing carbon and water quality offsets and credits. Based on experience with two pilot programs, McCarl (2002) estimated that market transaction costs (costs of organizing and participating in the market) for carbon credits could be around USD 0.83 per acre. However, it is difficult to generalize this estimate to other cases. Moreover, the costs of implementing programs are likely to decrease with experience, new measurement and monitoring technologies and with competition among service providers.

Mooney et al. (2004a, b) estimated the measurement and monitoring costs that are likely to be required to verify compliance with carbon credits produced by agricultural soil C sequestration. In a case study of their prototype measurement scheme, the upper estimate of measurement costs is 3% of the value of a C credit. While these estimates should be interpreted with caution— because they are not based on an actual contract implementation—they suggest that measurement costs would not necessarily prevent farmers from participating in a market for carbon offset credits.

Figure 3. C supply curves for adoption of conservation tillage in major central U.S. crop systems, with zero and USD 5/acre transaction costs



Source: Antle et al. (2007).

Antle et al. (2007) incorporated transaction costs in C supply curves, as illustrated in Figure 3. Transaction costs have the effect of creating a threshold price for the C supply curve. When C sequestration rates are relatively low, as is the case with the wheat system, transaction costs create a relatively high price threshold (equal to about USD 30 per metric tonne C). For the relatively higher C sequestration rates associated with the corn and soy beans, the threshold is about USD 20 per tonne. Figure 3 also shows that the effect of the transaction costs diminishes at higher C prices. Thus, it can be concluded that transaction costs are likely to be particularly important when C prices are low and in regions where C storage rates are low.

As regards water quality trading it has been estimated that finding trading partners and conducting verification plans amounts to USD 0.05 per pound of N credits if a clearinghouse is used and USD 0.10 if the trade is bilateral (Pennvest, 2012). Farmers' credit estimation and verification costs have been estimated to be 5% of farmers' total offset revenue (Newburn and Woodward, 2011).

The environmental effectiveness of management practices may vary greatly over space due to spatial heterogeneity and stochastic weather conditions. The environmental integrity and credibility of environmental markets, however, requires reasonably accurate estimates of how various practices perform under heterogeneous conditions. Accurate performance estimates help to reduce uncertainty about the quality and quantity of credits supplied from agriculture and increases demand for environmental services from agriculture as potential purchasers of credits find them more reliable. While monitoring environmental practice performance through edge-of-field or ambient quality measurement would be the most precise method for verifying environmental credits, it would be extremely costly and impractical due

to the non-point nature of many environmental services provided by agriculture (Ribaudo et al., 2010). Model-based estimates can be developed for many services from agriculture that can be seen as reasonably accurate by market participants (Kroeger and Casey, 2007). Moreover, government can support research on the effectiveness of environmental practices in the provision of environmental services. For example, the United States Department of Agriculture (USDA) supports the development of tools and methods for quantifying environmental services provided by different environmental practices. These tools help farmers to estimate environmental credits produced from a given practice while increasing confidence among other market participants regarding creditable practices (Ribaudo et al., 2010).

2.2. Environmental credit markets and stacking

Woodward (2011) provides a formal analysis of credit stacking in environmental markets. In his analysis the key question is whether there are economic reasons to allow or forbid stacking. Stacking is possible when two or more environmental effects are complements in the firm's abatement cost function. At the firm level, complementarity means that the abatement cost is lower than the sum of the costs of abating each pollutant separately. Woodward shows that credit stacking can be welfare increasing if the emission caps for all pollutants are set at their optimal levels, taking into account that complementarity lowers abatement costs and thus increases the socially optimal level of abatement and thus environmental quality. Hence, if the optimal level of pollution abatement is sought and emissions caps in separate markets are established with complementarity in mind then a policy allowing stacking leads to the least-cost allocation of abatement and thus maximizes social net benefits. However, single-market restrictions may work better if the multiple markets are not well coordinated or set optimally. The slopes of the marginal benefit and marginal cost curves affect the efficiency of stacking. When the marginal benefit curve is relatively steep or the marginal cost curve is relatively flat then stacking is efficient and improves social welfare relative to a situation where stacking is not allowed.

Horan et al. (2004) analyses the stacking of government agri-environmental payments and water quality trading credits. Joint implementation of these two policy mechanisms provides insights as regards efficiency and distributional implications of stacking. In the analysis the government incentive payments are provided to farmers to reduce the use of polluting inputs. Incentive payments are either targeted (farm-specific level that takes into account marginal damage from each farm) or non-targeted (payment level is averaged across all farms). The analysis examines both the coordinated and non-coordinated use of agri-environmental payments and water quality trading. In the uncoordinated case agri-environmental payments already exists and the trading authority takes this into account when designing the trading programme. If stacking is allowed, then these two programmes cross-subsidise each other and if it is not allowed, then the agri-environmental programme reduces the ability of the water quality trading programme to achieve environmental gains at least cost (since additional units of abatement from farms are more costly to purchase than the initial units). The degree of improved policy performance among these policies is found to depend on whether programmes are coordinated or not, whether stacking is allowed, and whether agri-environmental payments are targeted. Under policy coordination stacking provides efficiency gains through both programmes' joint influence on farmers' marginal decisions. As regards the non-coordinated policies stacking increases (decreases) welfare if agri-environmental payments are targeted (non-targeted). The analysis also shows that stacking may not solely benefit farmers, but may transfer part of the value of agri-environmental payments to point source polluters.

Lentz et al. (2013) analyses water quality trading and credit stacking in a case when both buyers and sellers of credits can only reduce pollution with large, discrete investments that yield discontinuous supply and demand of credits. Wastewater treatment plants (WWTPs) pay farmers to reduce nutrient runoff through establishment of wetlands and farmers may or may not be allowed to earn multiple environmental service credits from one wetland (nitrogen, phosphorus, and wildlife credits). Simulation results from the U.S. Corn Belt show that establishing wetlands is a more cost-effective way to reduce nitrogen runoff than

abatement by WWTPs. Stacking of environmental credits may improve social welfare and change market outcomes by altering the set of wetlands established if the demand for the primary market (nitrogen) is strong enough to cover most of the establishment costs of the wetlands and the supply of nitrogen credits is not exhausted. However, simulations further show that stacking may not have an effect on market outcomes under several circumstances and that stacked credits may not be truly additional. For example, stacking does not change the market outcome if either demand is so limited that a single large wetland would suffice to fulfill demand for credits or demand is so strong that supply of primary nitrogen credits is exhausted and thus market factors would not change the composition of wetlands that are established. The timing of credit sales is also important; if co-benefit credits (phosphorus and wildlife) are sold only after the wetlands are established to sell credits to the primary market (nitrogen), stacking does not affect real market outcomes.

Marshall and Selman (2011) discuss extensively the key design features of environmental markets when an explicit objective is to ensure environmental improvement and the integrity of environmental credits in addition to minimizing compliance costs of the regulated sectors. These key design features include: (i) baseline practice or performance requirements for credit and offset suppliers, (ii) retirement ratios altering the proportion of credits demanded to those supplied, and (iii) the role of ancillary benefits (co-benefits). Furthermore, they analyse the potential implications of environmental co-benefits on compliance costs and environmental objectives in the context of multiple environmental markets. Environmental co-benefits and stacking may complicate interpretations of additionality and baselines in the context of multiple environmental markets. For example, if a carbon offset market drives farmers to adopt no-till and if joint water quality benefits are created, a question arises whether those benefits can be considered additional in water quality markets since due to jointness they are already provided through incentives created by the carbon offset market (Marshall and Selman, 2011).

In their discussion Marshall and Selman (2011) distinguish between the primary market (the one that triggers the adoption of an environmental practice) and the secondary market (potential environmental credits from co-benefits). Additionality for the primary market is straightforward as it represents the difference between the current practice and the new adopted environmental practice.

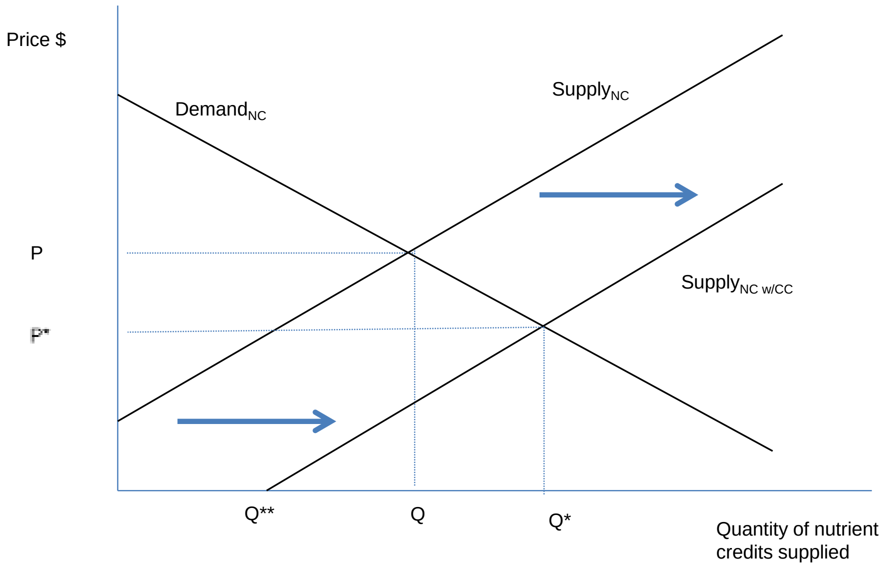
But what is the correct baseline for the secondary market? Is it the current practice or the newly adopted environmental practice incentivised by the primary market? The former represents a case where multiple environmental markets are independent and work as a single market for a given environmental good, while the latter indicates that the existence of the primary market has already incentivised the adoption of the environmental practice with co-benefits, without the existence of the secondary market, and thus these co-benefits cannot be considered additional from an inter-market additionality viewpoint (Marshall and Selman, 2011).

If environmental markets are interlinked and revenue from the primary market already covers adoption costs of the environmental practice, then allowing farmers to receive co-benefit credits from the secondary markets represents a windfall gain for farmers and a decrease in the environmental effectiveness of the combined regulation and offset market since non-additional credits are sold as offsets for increased emissions elsewhere (Marshall and Selman, 2011). Thus, a financial additionality test can be employed in the context of multiple environmental markets: if the adoption of an environmental practice is already profitable with credit sales to the primary market, secondary markets fail the financial additionality test (Marshall and Selman, 2011).

Marshall and Selman (2011) present a graphical analysis of environmental credit supply in interlinked environmental markets. In the example, farmers supply nutrient credits to the water quality market by adopting environmental practices that jointly produce carbon credits, such as no-till or green fallowing. If the secondary market for carbon exists and farmers earn revenue for carbon credits, the supply curve for

nutrient credits will shift as revenue from carbon credits reduce the costs of nutrient credits supply (Figure 4). As a result, due to returns from the carbon credit market, farmers are willing to supply more nutrient credits (Q*) at a lower price (P*). Revenue from the carbon credit market fully covers the cost of supplying Q** of nutrient credits and thus from a water quality market viewpoint, these credits are non-additional and the total expenditure (P*Q**) for these credits represents wasted conservation resources.

Figure 4. Supply of nutrient credits (NC) with a carbon credit (CC) market



Source: Marshall and Selman (2011).

As shown in Figure 4, stacking can change the value of environmental credits by increasing the aggregate supply and reducing the equilibrium credit price in the market. Thus, allowing stacking provides farmers multiple payment streams, but simultaneously reduces credit prices and therefore revenue from each individual market (Cooley and Olander, 2011). Hence, if environmental credit prices decrease due to stacking, environmental practices that used to be profitable to adopt with credits from one market may require sale of credits to several markets in order to cover adoption costs of the practice. This may have implications for financial additionality in the secondary markets as well. Environmental practices that were originally financially non-additional in the secondary markets may become additional if reduced credit revenue from the primary market no longer covers adoption costs (Cooley and Olander, 2011).

Marshall and Selman (2011) discuss alternative regulatory design options for addressing additionality in interlinked environmental markets. The first option is to allow the sale of credits only to one (primary) market and treat environmental co-benefits from the adoption of a practice as ancillary benefits rather than creditable additional benefits. The shortcoming of this option is that some low-cost multi-benefit practices

may not be adopted if the revenue stream from one market alone does not fully cover the adoption costs. The second option is to allow credit sales to several markets as long as the additionality criterion for each individual market is fulfilled; in this case there is no requirement for inter-market additionality — that is, for each market the baseline is the current practice. The disadvantage of this option stems from the supply of non-additional credits from an inter-market viewpoint. The third option is to employ financial additionality criteria to decide which practices are eligible for sale of credits to multiple markets. The fourth option for addressing inter-market additionality is to create a single aggregator institution that buys bundled environmental services from farmers and then unbundles them and sells credits within the individual markets (Marshall and Selman, 2011).

To conclude discussion in this section, in many cases the driver of offset demand is flexibility in government regulation which allows the regulated sector to purchase offsets from unregulated sectors to comply with the regulation. This flexibility enables reduction of total abatement costs if non-regulated sectors can reduce emissions at a lower cost. In this case the offset market does not provide incentives for additional environmental improvements above what is required by regulation, but ensures that environmental objectives are met at lower social costs. The environmental integrity of the offset market requires that all offsets are additional, because offset purchasers are allowed to increase their emissions above permitted levels. Hence, if the fundamental purpose of the offset market is to reduce the compliance costs of the regulation, while achieving the emission-reduction target, then non-additional offsets or credits should not be allowed. Because credit stacking complicates interpretations of additionality and baselines, it may produce some non-additionality in offset and credit markets, which in turn compromise environmental targets of the regulation.

3. Theoretical framework of agricultural supply of carbon and water quality offsets

Non-technical description of the theoretical framework of agricultural supply of carbon and water quality offsets is provided in this chapter. For interested readers a formal theoretical framework and analysis is presented in Annex A.

3.1. Baseline cultivation: no offset markets

Consider cultivation under heterogeneous land productivity illustrated for simplicity by two productivity classes, high and low. Land productivity within both classes is assumed homogenous. Each farmer owns one field parcel so that the total number of land equals total number of farmers. Both land productivities can be cultivated using two alternative tillage methods: conventional tillage based on moldboard plowing or conservation tillage, say no-till, based on direct drilling of seeds and fertilizer in the soil. The crop is produced using fertilizer as variable input and a set of other inputs (seeds, labor, and capital) that can conveniently be regarded constant per hectare. Crop yield as a function of fertilizer input varies depending on the tillage method.

In the baseline with no offset markets, farmers choose optimal fertilizer application, based on relative prices and allocate each parcel of land to the tillage method which produces highest profits. It is assumed that no-till is more suitable for high productivity land and conventional tillage for low productivity land.[2] Under these assumptions and based on the optimal use of inputs, high land productivities are allocated to no-till and low productivities to conventional tillage.

2 Ogle et al. (2012) conducted a literature review to compile results from studies evaluating changes in yield following the adoption of no-till cultivation. Their results of corn (maize) yields are influenced by nitrogen application levels so that yield losses are relatively lower (higher) when nitrogen application level is high (low).

This baseline is the point of departure for the analysis of agricultural production in the presence of carbon offset markets. Note that the baseline within each productivity class of land entails that the farmers are homogenous, that is, they face identical relative costs. This feature will change when a voluntary carbon offset market is introduced to the analysis as farmer-specific transaction costs start to matter for farmers' willingness to participate in carbon and water quality offset markets.

3.2. Cultivation under voluntary participation in carbon markets

It is assumed that the farmer primarily participates in a voluntary carbon market and produces carbon offsets. Carbon offsets provide potential additional revenue. If a farmer wishes to produce offsets and participate in carbon markets, three measures are in principle available to produce carbon offsets from crop cultivation.

First, the farmer may reduce applied nitrogen fertilizer, which stimulates microbial conversion of soil nitrogen to nitrous oxide (N_2O) emissions. Reducing nitrogen application below the baseline optimum for a given tillage method yields carbon offsets from reduced N_2O emissions, based on the relative heat-trapping potential of CO_2 and N_2O. Second, if the farmer applies conventional tillage, then offsets can be produced by switching to no-till, which increases carbon storage in cropland soils. Third, the farmers may allocate some field parcels to long-term green set-aside which enhances carbon sequestration. Participation in carbon offset markets involves transaction costs, which may represent an important constraint on market entry. Transaction costs are comprised of costs of participating in the offset market as well as additional, farmer-specific transaction costs relating to gathering and producing information on offset markets and implementation of measures needed to produce offsets. Thus, while farmers within both land productivity classes face similar yields and relative costs, transaction costs may differ.[3] Hence, farmers in each productivity class can be arranged by the size of transaction costs from lowest to the highest cost. For farmers facing high transaction costs, it may not be optimal to participate in offset markets.

Under the carbon offset market optimal fertilizer intensity is reduced relative to the baseline. Once optimal fertilizer application is known, farmers in each land productivity class choose the tillage method. *In the high productivity class*, no-till was the baseline cultivation technology and the possibility of selling carbon credits reinforces its superiority relative to conventional tillage. *In the low productivity class*, carbon offset revenues improves returns to no-till cultivation relative to conventional tillage, yet both benefit from the carbon market. Depending on the revenue of the generated offsets under no-till relative to those generated under conventional tillage, either no-till is introduced or farmers continue employing conventional tillage.

In the high productivity class, farmer participates in the carbon offset market provided that profits from cultivation plus offset revenue less transaction costs is higher than baseline profits from cultivation. Farmer is indifferent between participation and nonparticipation when transaction costs are equal with the difference of profits under carbon market participation and baseline. For transaction costs below this threshold value, farmers participate in offset markets and for costs above this value they do not participate but continue with the baseline cultivation technology.

In the low productivity class, conventional tillage is the baseline technology but farmers may adopt no-till if it provides higher revenue and participation in the carbon offset market is profitable. Based on the two alternative possibilities for tillage method choice, the participation decision can be expressed as

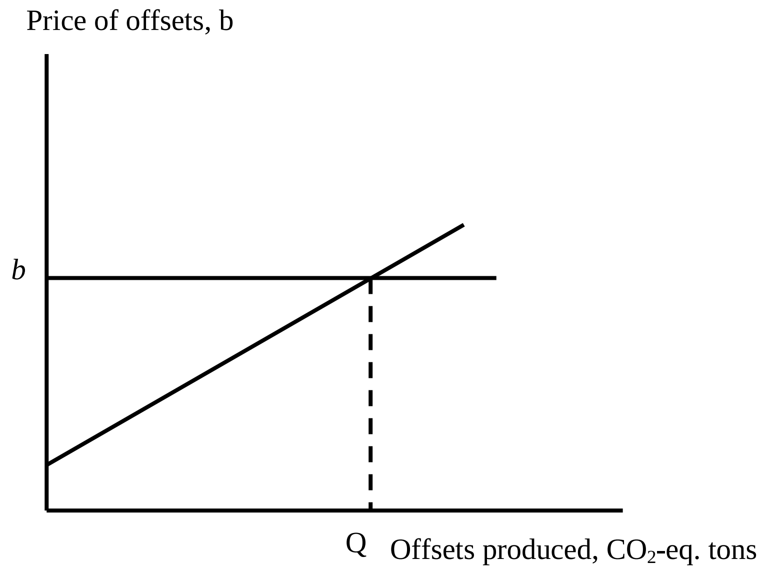
3 Mettepenningen et al. (2009) have analysed the variation of farmers' private transaction costs in the context of European agri-environmental policy. They found that "search costs", including the costs of looking for information on specific agri-environmental measures and comparing these with alternatives, show wide variation. Mean search costs were EUR 11.1/ha with a standard deviation of 54.2.

follows. Suppose first that no-till is the superior tillage method for farmers who participate in the carbon market. Then the farmers with transactions costs below critical value participate in carbon markets using no-till but farmers with transaction costs above critical value will not participate and continue to employ conventional tillage. If instead conventional tillage is more profitable under carbon markets, both participating and non-participating farmers continue with conventional tillage, and participating farmers only reduce their fertilizer application for generating carbon offsets.

The supply of offsets to carbon markets from crop production can be defined as the sum of offsets over participating farmers in high and low productivity classes. The offset supply is generated by reduced fertilizer use and possibly by a discrete shift from conventional tillage to no-till in participating lands from low productivity class. Furthermore, on low productivity land green set-aside establishment may become profitable and thus affect offset supply.

The impact of an increase in carbon offset price on supply of carbon offsets is positive, as number of participating farmers increases in offset price and fertilizer intensity decreases in it. Hence, the total supply of carbon offsets from crop production is an increasing function of carbon price. The supply function of offsets from crop production is illustrated in Figure 5. For an exogenous offset price *b* the optimal amount of supplied carbon offsets is *Q*.

Figure 5. The supply function of carbon offsets from crop production



3.3. Water quality offsets as function of carbon offset supply

Farmers' production choices regarding the supply of carbon offsets may have multiple effects on water quality. Reduced fertilizer use decreases nutrient runoff, while green set-aside planted to perennial grasses (without applied fertilizer) may significantly reduce all forms of runoff. No-till adoption generally decreases erosion, nitrogen runoff and particulate phosphorus runoff (but may increase dissolved reactive phosphorus runoff).[4]

4 Water quality credits from no-till adoption depend on whether nitrogen or phosphorus or both are traded and the trading ratio needs to reflect the fact that no-till may increase dissolved phosphorus runoff.

It is assumed in the analysis that a farmer can either participate in one market (the carbon market) or multiple markets (both carbon and water quality markets).[5] Additionality is an important policy concern in the establishment of environmental markets. Two alternative interpretations of additionality in environmental services are employed in the analysis below. The first interpretation is based on financial additionality: an increase in the number of participating farmers in the carbon market due to the possibility of stacking water-quality credits. Providing more revenue from stacked water-quality credits makes participation in the carbon market profitable for those farmers who would not have participated without the possibility to stack water-quality credits. The second interpretation for additionality used in this analysis is inter-market additionality, which requires that farmers who are willing to sell water-quality offsets need to employ abatement practices that go beyond those required by participation in the carbon-offset market. One possibility to increase the use of abating inputs is to establish vegetated field strips (buffer strips) between waterways and the field parcel. These can very effectively reduce both soil erosion and nutrient runoff.

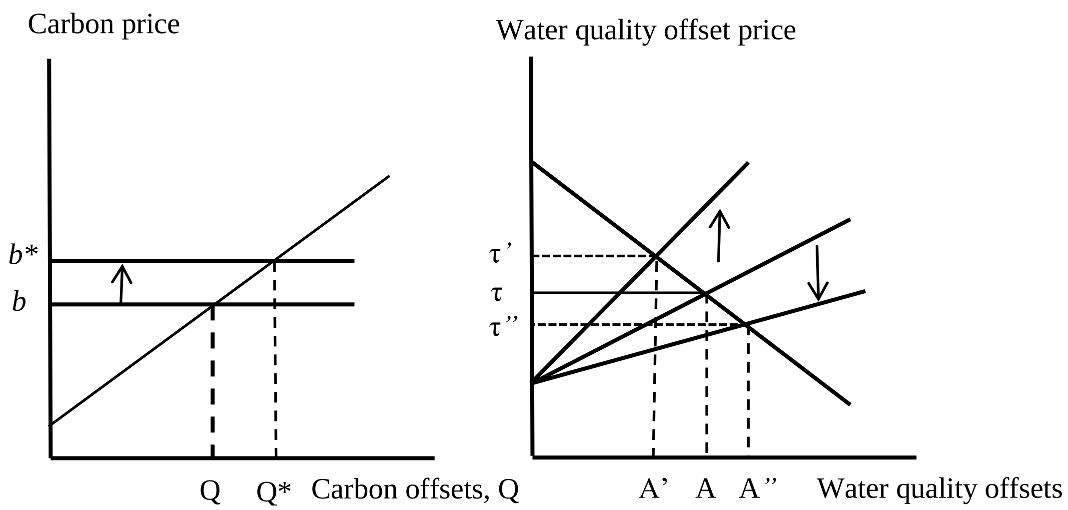
A. Financial additionality

Let the price of water-quality offsets be τ. It was discussed above that in both high productivity land and low productivity land farmers with transaction costs higher than threshold value do not participate in the carbon markets. When stacking of water-quality offsets is allowed and farmers get additional revenue from water-quality offsets, the new threshold level of transaction costs increases by the value of water-quality offsets. The supply of water-quality offsets, β, from increased participation in carbon markets depends on its own price and the carbon price. The impact of an increase in the price of water-quality offsets increases participation and the supply of water-quality offsets. The impact of a higher carbon price is generally ambiguous due to two opposing mechanisms. A higher carbon-offset price invites more farmers to participate in the carbon market. This reduces the number of participating farmers in water-quality offset markets. This impact is counter-affected by the further decrease in fertilizer use intensity, which increases water quality offsets for a given water offset price. If the former dominates, the supply of water-quality offsets decreases, but if the latter one dominates then it increases. In the special case the two opposing effects may offset each other and the supply of carbon offsets does not change. Whether the increased revenue from water-quality offsets dominates depends largely on the distribution of transaction costs. Water-quality offset supply decreases if differences between transaction costs increase sharply at the tail of distribution. Thus, in the general case an increase in the carbon price may increase, decrease or maintain the supply of water quality offsets.

Figure 6 illustrates these possibilities. In the left-hand side panel, carbon price increases from b to b^* (for simplicity carbon price is treated as exogenous). Supply of carbon offsets adjusts along the supply curve and the amount of supplied carbon offsets shifts from Q to Q^*. In the right-hand side panel, the original equilibrium in the water quality offset market is given by price τ and amount of offsets A. Increasing carbon price may lead to three alternative equilibria: equilibrium remains constant or the supply function shifts upwards or downwards.

5 Note that by assumption a carbon market is the primary market in this analysis and thus the possibility of participating just in the water quality market is not analysed here.

Figure 6. The impact of carbon price on the supply of water quality offset and their price



Analysis becomes more complicated when the possibility of project quality differentiation is allowed so that combination of practices generates higher level of offsets and not only participation rate but also conservation and input use intensities change.

B. Inter-market additionality

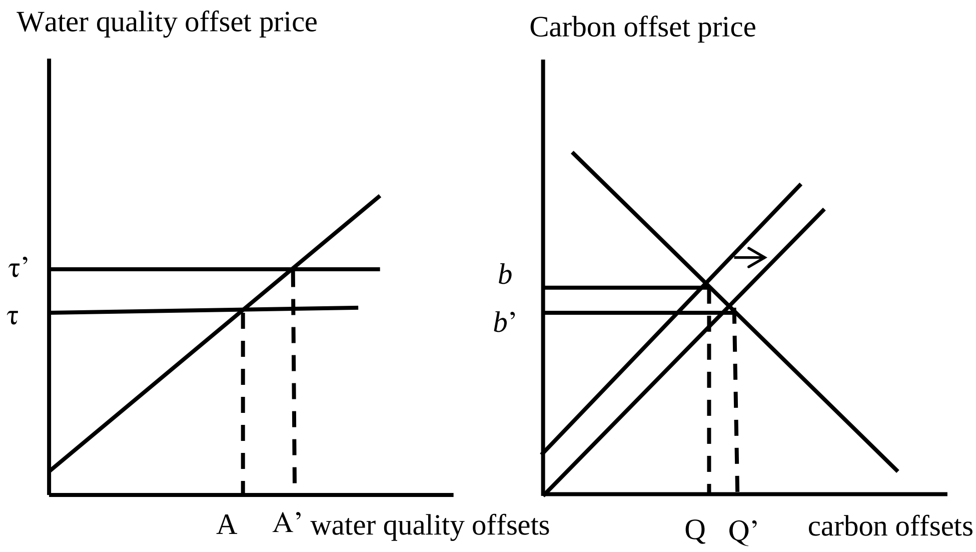
A participating farmer may optimize fertilizer intensity and tillage method for both carbon and water quality offset markets in order to achieve a greater revenue stream from offsets. To this end the farmer is allowed to establish a vegetated field strip to support the production of both carbon and water quality benefits on working cropland. The baseline for the carbon markets is the same as above but the baseline for the water quality offset markets changes. This baseline must account for the fact that farmers participating in carbon markets produce additional benefits above the level determined by optimal input use for participation in carbon markets in the absence of water quality offsets. Thus, the new baseline in water quality offset markets is defined by fertilizer intensity under participation in carbon markets in the absence of water quality offset markets.

The farmer chooses both fertilizer intensity and filter strip width so as to maximize profits. The optimal fertilizer intensity is now smaller than in the case where fertilizer use was optimized only in terms of carbon benefits. The vegetated field strip width depends on the establishment cost of the practice, the foregone income lost from cultivation and on the revenue from carbon offsets and water quality offsets. A field strip is only established provided the return to its establishment is high enough. If a field strip is established, only part of the parcel is allocated to it, because the marginal increase of produced water quality offsets decreases with the width of the field strip. Both choices provided indicate increased use of abating inputs and higher quality of the project in terms of environmental benefits.

As before, a farmer compares profits under nonparticipation and participation and is indifferent between them at the threshold value of transaction costs. Given that farmers are allowed to optimize both fertilizer application and field strip establishment under a positive water quality offset price, the number of participating farmers increase and the practice adoption supplies more carbon offsets and water quality offsets than in the previous cases. Supply of carbon offsets expands with carbon price as before. Furthermore, carbon offset supply expands also with the higher water quality offset price. In both cases,

the number of farmers participating in carbon markets increases and farmers increase use of abating inputs. Figure 7 illustrates the impact of water quality offset price on carbon offset supply.

Figure 7. The impact of water quality offset price on carbon offset supply and carbon price



From Figure 7, an increase in the price of water quality offsets shifts the carbon supply curve outwards so that the equilibrium carbon price decreases and the amount supplied to the market increases. Water quality offset supply increases in offset price via two mechanisms: increased participation and increased use of abating inputs.

An increase in the carbon offset price has an impact through multiple channels. Higher carbon price increases the number of those farmers who would participate even without water quality offsets, which decreases water quality offset supply. But it also increases the number of those farmers for whom participation in the carbon market is profitable only with the water quality offset. Depending on which impact dominates, this effect is either positive or negative. Higher carbon price increases the use of both abating inputs, which tends to increase supply of water quality offsets. Again, all possibilities are present: supply of water quality offsets may increase, decrease or remain constant. Relative to the previous case, however, an increasing supply impact is more likely.

4. Model data

Empirical application of the model is based on data estimates for the U.S. Corn Belt region. The Corn Belt was selected as a case-study area for the following reasons:

- The Corn Belt represents a major agricultural region of the U.S., accounting for a significant share of national corn and soybean production.
- The region provides a mix of no-till, reduced-till and conventional tillage acreage in corn and soybean production.
- A significant amount of environmental set-aside acreage (i.e. riparian buffers, filter strips, grass cover crops) is enrolled in the region through the Conservation Reserve Program (CRP) and other USDA conservation programs.

- Heterogeneity in soil, slope and climate conditions exists across areas of the Corn Belt.
- There is increasing policy interest in environmental markets to address regional water quality and other environmental concerns.

Three model areas, drawn from the eastern, central and western portions of the Corn Belt region, were selected for use in the empirical analysis (Figure 8). The model areas were drawn from the Regional Environment and Agriculture Programming Model (REAP), a US agricultural sector model maintained by the Economic Research Service of the U.S. Department of Agriculture (ERS-USDA).[6] The REAP model defines regions based on the intersection of USDA Farm Production Regions, USDA Land Resource Regions (LRRs) and sub-watershed basins defined by the US Geological Survey (USGS).[7] The study regions identified for this analysis include portions of the Ohio, Upper Mississippi and Lower Missouri river basins located within the five Corn Belt states. The three study regions (Regions 1-3), with REAP model identifier, region location and sub-watershed, are as follows:

Region 1: CBM0503 – Eastern Corn Belt – Ohio River (Muskingum-Skotot-Miami sub-basins)

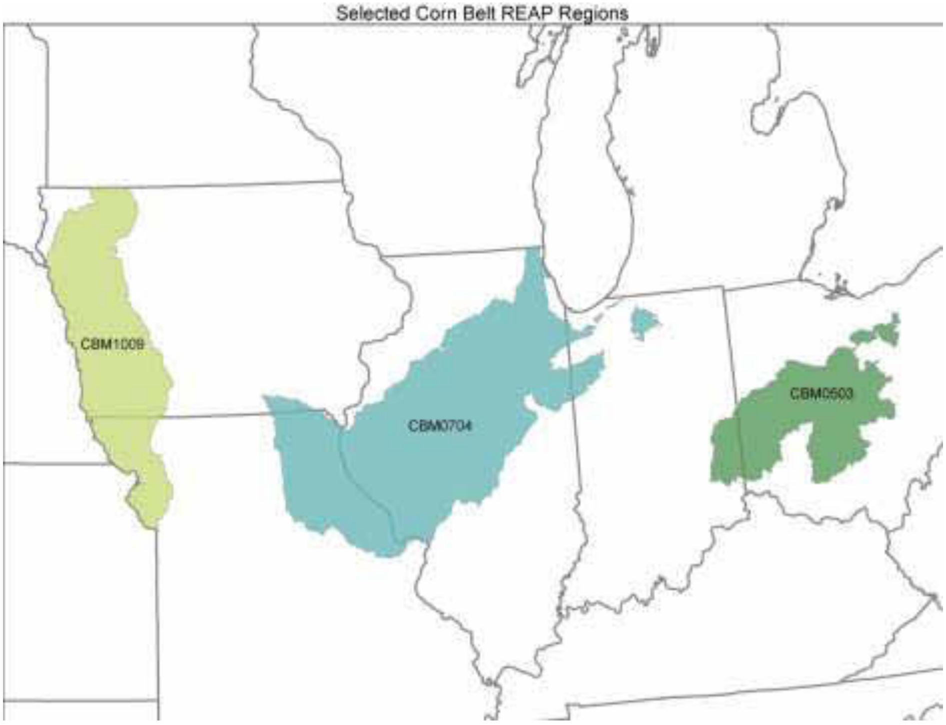
Region 2: CBM0704 – Central Corn Belt – Upper Mississippi (Salt-Sny-Illinois sub-basins)

Region 3: CBM1009 – Western Corn Belt – Lower Missouri.

6 REAP is a static, partial equilibrium optimization model of the US agricultural sector that quantifies agricultural production, crop management and resource use, farm returns, and associated environmental indicators at a national and regional scale. The model estimates welfare-maximizing levels of fieldcrop and livestock production and selected processed commodities under changing policy, technology, market and resource settings. The REAP model has been applied to address a wide range of agri-environmental issues including soil conservation and environmental policy design, climate change adaptation and mitigation policy, and regional effects of trade agreements. For a discussion of the REAP model, see Malcolm et al., 2012, Appendix A.

7 USDA Farm Production Regions (FPRs) follow multi-state political boundaries; the Corn Belt region comprises Iowa, Illinois, Indiana, Ohio and Missouri. USDA Land Resource Regions represent geographic concentrations of commodity production, with comparable soils and climate conditions; model regions selected for the empirical analysis fall within LRR 'M': Central Feed Grains and Livestock'. REAP model regions are further disaggregated to reflect sub-watershed boundaries following USGS 4-digit hydrologic unit codes.

Figure 8. Model regions in the study



Two representative crop rotations for the Corn Belt region are considered in this study: Continuous Corn and Corn-Soybean (alternate years). The two rotations represent approximately 77% of field crop acreage within the Corn Belt region.

The study captures production heterogeneity through regional differences in climate and land resources. Average annual precipitation (weighted across farmland acreage) ranges from about 102 cm (40 inches) in the eastern Corn Belt to about 81 cm (32 inches) in the drier western reaches. Model regions are further differentiated by Highly Erodible Lands (HEL) and Non-Highly Erodible Lands (NonHEL), based on field data from the 2007 Natural Resources Inventory (NRI). Soil assumptions by crop rotation and HEL and NonHEL categories reflect predominant soils by land type and region from the NRI. Regional heterogeneity in soil moisture and land quality may have varying impacts on crop yield, production costs and environmental outcomes, with implications for private and societal incentives for conservation practice adoption.

Crop yield, input use and environmental parameters are generated by the Environmental Productivity and Integrated Climate (EPIC) model.[8] EPIC is a crop biophysical simulation model designed to estimate the effect of management practices on crop yields, soil and water dynamics, nutrient cycling and pesticide fate at the field level. The model uses a daily time step to simulate seasonal crop growth and soil and water processes for alternative tillage, crop rotation, and soil and nutrient management regimes, under a distribution of region-specific weather scenarios to estimate long-run equilibrium outcomes. Environmental parameters generated by EPIC include, for example: (i) soil erosion, (ii) different types of

8 EPIC was developed jointly by the USDA Agricultural Research Service (USDA-ARS), the USDA Natural Resources Conservation Service (USDA-NRCS) and Texas A&M University.

nitrogen runoff and leaching, (iii) different types of phosphorus runoff and (iv) changes in soil carbon.[9] EPIC parameters developed for production alternatives in the ERS REAP model are applied to model 36 cropping systems defined in the US case-study analysis (Annex B: Table B.1).[10]

Production cost data for the study were obtained from the REAP model, which draws on cost-of-production estimates from USDA's Agricultural Resource Management Survey (ARMS) and production data developed at the World Resources Institute (WRI). Primary cost items in the study include: nitrogen and phosphate fertilizer, seed, pesticide, energy, labor, variable and fixed costs of machinery, and land rent. Most of the cost items vary by rotation and tillage practice (no-till, reduced-till and conventional tillage), with some adjustments for HEL/NonHEL. Farmers' profits are defined by total revenue from crop production less variable and fixed costs of production. Cost and price data are from years 2010-2013.

Quadratic crop nitrogen response functions ($y=a+bN-cN^2$) estimated with US data were calibrated for 36 region-crop-rotation-tillage-erodibility combinations with known nitrogen application level and known yield level. The original value of parameters a and c from published research (Boyer et al., 2013) were retained in the nitrogen response function and parameter b was solved to correspond to known nitrogen (N) application level and yield level for each combination. Thus 36 nitrogen response functions were derived (one per each combination).

This empirical application of the theoretical framework focuses on three environmental outcomes: surface water quality, GHG emissions and soil carbon sequestration. Using the EPIC data estimates, functional expressions were estimated for nitrogen runoff, phosphorus runoff, and sediment runoff. These exponential runoff functions provide the core of the environmental component of the model. A share of field parcel may be retained as a field strip (also called buffer strip or filter strip) in order to reduce surface runoff of nitrogen, phosphorus and sediment. The effectiveness of field strips for nutrient and sediment runoff control is based on FABRI (2007).

Establishment of green set-asides (grass cover) reduces nutrient and sediment runoff relative to crop production while sequestering carbon and providing wildlife habitat (ICF, 2013). Establishment cost of green set-asides and estimates of soil carbon sequestration with cropland conversion to grass are based on ICF (2013). Estimates of nutrient and sediment runoff reduction capacity of green set-asides are based on NRCS (2012). For each field parcel, the opportunity cost of green set-aside establishment is based on the highest profit use (either conventional, reduced-till or no-till systems, depending on their relative profits).

Soil carbon sequestration rate for each region/rotation/tillage/erodibility combination are obtained from EPIC. Life-cycle-analysis estimates of GHG emissions are defined for conventional and conserving tillage production chains—including tillage, planting, fertilizer and pesticide application, grain drying, etc. (Adler et al., 2007). Soil N_2O emissions from nitrogen application are also taken into account (Grandy et al., 2006).

Although this study focuses on how farmers respond to offsets prices and thus farmers' abatement costs, the social valuation estimates for environmental effects are employed in order to compare relative value and significance of different environmental effects and to see how much offset prices deviate from the social valuation of given environmental effects. The social cost of damage from nitrogen runoff,

9 Nitrogen runoff and leaching includes nitrogen losses in solution, in sediment, drainage tiles, leaching, and additional subsurface flows. Phosphorus runoff includes phosphorus loss in solution and in sediment.

10 The analysis focuses on rainfed (non-irrigated) production, which is predominant for corn and soybean production in the Corn Belt. Separate EPIC runs were generated, with and without drainage tiles, and yield, input and environmental coefficients were acreage-weighted to provide a composite measure.

phosphorus runoff, and sediment runoff are assumed to be proportional to their aggregate runoff. The marginal cost of runoff damage is assumed constant per pound of nitrogen, phosphorus, and sediment.[11] Marginal damage from sediment runoff and erosion is fixed at USD 2.77/ton of erosion in the Corn Belt region (Hansen and Ribaudo, 2008). Marginal water quality damage from nitrogen and phosphorus runoff is set at USD 6.38/pound of N-equivalent runoff (Birch et al., 2011). For GHG emissions, a constant marginal social damage estimate of USD 24/ton of CO2-eq is employed (Tol, 2005).

Demand functions for environmental service markets are estimated for both water-quality and carbon credits. Nutrient reduction credit prices are based on N and P credit prices reported for a water-quality trading clearinghouse established in Pennsylvania (O'Hara et al., 2012).[12] Prices for carbon credits are based on auction prices reported by the Regional Greenhouse Gas Initiative, operated by the Northeast and Mid-Atlantic states (RGGI, 2014).

5. Results

This chapter provides results of the empirical application of the model. First, the baseline results without offset markets are presented.[13] Next, carbon sequestration practices and offset markets are analysed without water quality markets. Finally, water quality offset markets are introduced and stacking of credits is allowed. Key issues to be addressed include: (i) implications of stacking on market participation incentives and acreage response, (ii) offset supply and equilibrium offset prices, and (iii) impact of transaction costs on market participation.

5.1. Baseline without carbon offset markets

Table 1 provides a summary of baseline results. The results are presented for three model Regions (located within the eastern, central and western Corn Belt) and within each Region for NonHEL (Non-highly erodible) and HEL (Highly erodible) lands separately. Results are given as an average per acre and weighted by land use shares of alternative tillage methods and crop rotations within each Region and erodibility class.

11 This assumption of constant marginal damage from nutrient and sediment runoff is a simplification. However, as the changes in nutrient and sediment runoff are not very large, constant marginal damage provides a reasonable approximation.

12 A multi-state water-quality trading market in the Ohio River basin, encompassing much of the eastern Corn Belt region, is to become operational in 2014. The Ohio River Basin Trading Project has been promoted as a cost-effective means of addressing local water-quality concerns as well as hypoxia in the Gulf of Mexico.

13 These Baseline results are representative estimates of costs, returns and environmental parameters. It should be noted that these parameters vary spatially and over time, based on soil conditions, weather events, commodity market conditions, and many other factors.

Table 1. Average per acre results for HEL and NonHEL lands in Regions 1, 2, and 3[1]

		Region 1		Region 2		Region 3	
Production and profits		**NonHEL**	**HEL**	**NonHEL**	**HEL**	**NonHEL**	**HEL**
	Nitrogen application, lbs/acre	112	109	111	115	111	115
	Corn production, lbs/acre	9542	9323	8808	8218	6683	6722
	Profits, USD/acre	755	762	686	586	453	433
Environmental effects	Carbon sequestration, lbs/acre	741	532	1073	536	1127	456
	CO_2-eq emissions from fertilizer and cultivation, lbs/acre	394	387	391	404	391	404
	Nitrogen runoff, lbs/acre	38	39	41	45	14	38
	Phosphorus runoff, lbs/acre	2.0	3.4	1.0	3.7	1.5	3.1
	Erosion, tons/acre	1.4	5.6	0.6	9.8	0.8	6.5
Social value of environmental effects and ex-post SW, USD/acre	CO2-eq, net climate damage	-4	-2	-8	-2	-9	-1
	Nutrient runoff damage	116	123	121	142	44	118
	Erosion damage	4	15	2	27	2	18
	Net environmental damage	116	136	114	168	37	135
	Ex-post social welfare	639	626	572	418	416	298

1. Region 1: CBM0503 – Eastern Corn Belt – within Ohio River basin (Muskingum-Skotot-Miami sub-basins);
   Region 2: CBM0704 – Central Corn Belt – within Upper Mississippi basin (Salt-Sny-Illinois sub-basins);
   Region 3: CBM1009 – Western Corn Belt – within Lower Missouri basin.

Corn yields and profits vary significantly between the regions. Based on the model simulation results for representative field conditions used in the case-study analysis, Region 1 (Eastern Corn Belt) represents the relatively highest productivity region and Region 3 (Western Corn Belt) the lowest productivity region. Profitability generally follows measures of yield productivity. Nitrogen application levels do not vary significantly between Regions and erodibility classes due to weighted averages. However, nitrogen applications do vary by crop rotation, tillage system and HEL and NonHEL land quality, ranging from 77 lbs/acre to 158 lbs/acre. Due to nitrogen fixation, soybean-corn rotations have significantly lower nitrogen application levels relative to continuous corn cultivation.

As shown by Table 1 soil carbon sequestration levels are higher for NonHEL lands in each region and technical sequestration capacity varies between model Regions. Erosion rates show a large difference between HEL and NonHEL lands, with highest erosion rates under conventional tillage and lower rates

under reduced tillage and no-till. CO_2-eq emissions from cultivation practices and fertilizer application are smaller than soil carbon sequestration in each Region and thus net GHG emissions are negative. Nitrogen runoff varies between regions and especially between HEL and NonHEL lands. Phosphorus runoff is significantly higher in HEL lands relative to NonHEL lands due to higher erosion rates and resulting sediment-bound phosphorus runoff (particulate phosphorus).

Environmental indicators vary considerably by tillage system. In general, sediment, nitrogen and phosphorus runoff are lower and carbon sequestration rates higher under conserving tillage systems—no-till and reduced till. Conserving tillage systems are generally more profitable on HEL soils, particularly in the drier western Corn Belt region where soil-moisture retention is a more significant concern and the probability of excessive early-season field wetness is lower.

As regards the social value of various environmental effects considered in the analysis, one can infer that agriculture's net climate impact is positive as soil carbon sequestration more than offsets CO2-eq emissions from cultivation practices and fertilizer application. Relative to other environmental effects, however, the combined nutrient damage from nitrogen and phosphorus runoff is significant for each model Region. Erosion damage is significantly higher for HEL lands than for NonHEL lands due to large differences in erosion rates. In social value terms, agriculture's overall net environmental impact is negative for each Region, but profitability of production makes ex-post social welfare (profits less net environmental damage) clearly positive and relatively high across Regions.

Table 2 provides aggregate results for HEL and NonHEL lands in Regions 1, 2, and 3.[14] Share of HEL (68%) is largest in Region 3, which is also the lowest productivity Region (based on model parameter estimates). In Regions 1 and 2 the share of HEL is 11% and 17%, respectively. All tillage methods (conventional, reduced, and no-till) are represented in each Region and across HEL and NonHEL lands, although the share of conventional tillage on HEL lands is small. The share of conservation tillage (reduced tillage and no-till combined) is higher in HEL lands than NonHEL lands. It is worth noting that the share of conservation tillage (reduced tillage and no-till combined) is already very high in the baseline in every Region, exceeding 82%.

Aggregate corn production and farmers' profits are significantly higher on NonHEL lands, especially in Regions 1 and 2. As already indicated by per acre results, soil carbon sequestration exceeds CO2-eq emissions from fertilizer and cultivation practices, and nitrogen runoff dominates negative environmental effects based on damage costs assumptions in the analysis. However, ex-post social welfare estimates are driven largely by profitability of production. With the largest land area and mean productivity, Region 2 has the largest aggregate profits and net-environmental damage, but also the highest ex-post social welfare effects.

14 One short ton (abbreviated throughout the paper as simply "ton") is equal to 0.9072 metric tonnes."

Table 2. Aggregate results for HEL and NonHEL lands in Regions 1, 2, and 3

	Region 1		Region 2		Region 3	
	NonHEL	HEL	NonHEL	HEL	NonHEL	HEL
Acreage, million acres	2.7	0.3	10.3	1.7	4.1	2.8
Share of CNV/RED/NLL, %	15/58/27	2/42/56	20/56/24	8/41/51	15/59/27	1/43/56
Corn production, billion lbs	26.3	2.8	95.1	15.2	29.9	21.8
Profits, billion USD	2.7	0.3	8.9	1.4	2.7	2.1
Carbon sequestration, million tons	1.0	0.1	5.5	0.5	2.2	0.5
Nitrogen runoff, million lbs	69	9	242	52	41	50
Phosphorus runoff, million lbs	4	1	9	4	5	4
Erosion, million tons	0.7	0.5	3.0	5.7	0.9	2.6
CO2-eq emissions from fertilizer and cultivation practices, million tons	0.44	0.04	1.77	0.27	0.67	0.44
Nutrient runoff damage, million USD	211	28	727	160	134	156
Erosion damage, million USD	1.9	1.5	8.2	15.9	2.4	7.3
Net climate damage, million USD	-14.1	-1.1	-89.2	-4.7	-37.2	-0.7
Ex-post social welfare, billion USD	2.5	0.3	8.2	1.3	2.6	1.9

5.2. Carbon offset markets

As discussed in the theoretical framework it is assumed here that if a farmer wishes to participate in voluntary carbon markets, three measures are available to generate carbon offsets from crop cultivation. First, the farmer may reduce fertilizer intensity below the baseline optimum under each tillage method (conventional, reduced, and no-till) in order to reduce N_2O emissions. Second, the farmer can switch from conventional and reduced tillage to no-till. Third, a farmer can allocate some field parcels to long-term green set-aside in order to sequester soil carbon.

The analysis further assumes that participation in carbon offset markets involves transaction costs (TCs) for farmers. Three levels of costs are assumed: TCs = 0, TCs = USD 1/acre, and TCs = USD 2/acre, reflecting a range of costs that might conceivably occur in the marketplace. In addition, farmer-specific randomised transaction costs are included in order to represent variation in farmers' cost of gathering and producing information on offset markets and implementing conservation measures needed to produce offsets.

The first option for producing CO_2-eq offsets involves reduced nitrogen application levels and related reductions in applied phosphorus. It was assumed here that farmers reduce 30% of their nitrogen

application in order to decrease N_2O emissions.[15] Although a 30% reduction in applied nitrogen is significant from a production and profits standpoint, it has only a small impact on N_2O emissions. CO_2-eq offsets provided by this option vary from 65 to 132 lbs/acre. In none of the cases does offset revenue compensate farmers' foregone profit and transaction costs of market participation, even with zero transaction costs and offset price as high as USD 100/ton of CO_2-eq.

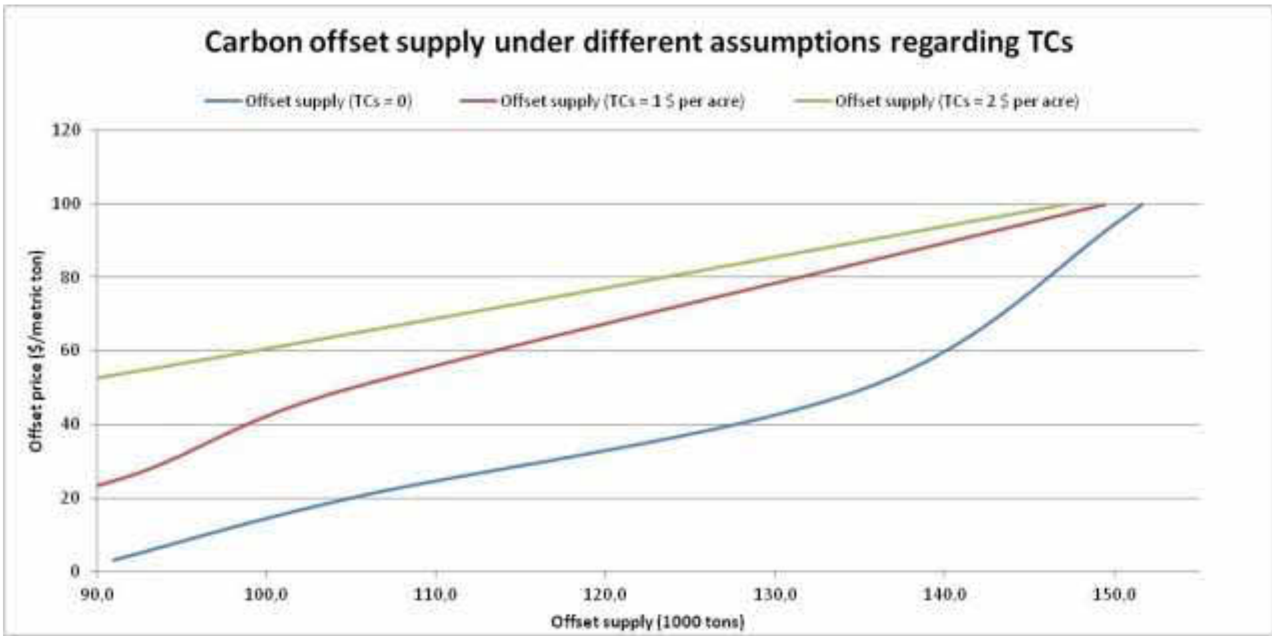
Similarly, establishment of green set-asides to produce CO_2-eq offsets alone is not profitable for farmers in any of the cases. Farmers' profit foregone plus establishment cost of green set-aside are not compensated by offset revenue, even with TCs of zero and a carbon offset price of USD 100/ton of CO_2-eq.[16]

Farmers' incentives for generating offsets through tillage system conversion vary across Regions. Without accounting transaction costs, a switch to no-till is profitable for conventional tillage and reduced tillage farmers only in Region 3 NonHEL lands with carbon offset prices starting from USD 3/ton of CO_2-eq. With zero TCs, carbon offsets will be supplied from a total of 2.4 million acres, or 68% of NonHEL acreage in the region. With positive TCs, profitability of offset supply in the region decreases, and depending on the level of TCs and offset price, the share of participating parcels ranges between 8% and 13% (representing 1.7 – 2.7 million acres).

Figure 9 shows the total carbon offset supply as a function of offset price (between USD 3 and USD 100 per ton of CO_2-eq) under differing cost assumptions regarding TCs.

Figure 9 clearly illustrates the impact of transaction costs on carbon offset supply.

Figure 9. Carbon offset supply under different Transaction Cost (TC) assumptions



15 Analysis assumes a 30% reduction of nitrogen application for each case considered. Another option would be to optimize the amount of reduced nitrogen application for each case with alternative levels of carbon offset price and transaction costs.

16 Note that the current offset price is USD 3/ton of CO_2-eq.

As regards carbon offset markets and agricultural supply of carbon offsets in the case study the following observations can be drawn. First, due to the relatively small impact of applied nitrogen on N_2O emissions and thus the small amount of offsets produced, reduced nitrogen application is not a profitable option for farmers without compensation for water quality offsets. Similarly, the establishment of green set-aside is not profitable without water quality offsets due to significant profit foregone. In contrast, a switch from conventional tillage and reduced tillage to no-till is profitable in some cases, although prevailing carbon offset prices and transaction costs have significant impact on the number of participating parcels. One reason why current carbon offset prices do not incentivize farmers to participate carbon offset markets without stacking is the fact that there is already widespread adoption of no-till in the baseline and consequently, marginally, switching additional acreage to no-till is much more costly. Overall one can argue that carbon offset markets with current offset prices do not adequately incentivize farmers to participate environmental markets without the possibility of stacking water quality offsets.[17]

5.3. Water quality offset markets

5.3.1 *Financial additionality*

As shown in the theoretical analysis, farmers' production choices regarding the supply of carbon offsets have implications for water quality. Reduced fertilizer use decreases nutrient runoff while no-till conversion and green set-aside reduce both soil erosion and nutrient runoff. With the added potential revenue from water quality offsets, farmers' participation in carbon offset market becomes more profitable. Thus, the possibility of selling co-benefit credits should enhance the profitability of carbon offset options (analysed in the preceding section) and potentially increases farmer participation in offset markets.

As regards water quality offsets, the following assumptions are made on the basis of the recent literature (O'Hara et al., 2012 and Pennvest, 2012). Base prices for nutrient credits are fixed at USD 3/lbs of N and USD 4/lbs of P. Credit estimation and verification costs for farmer are assumed to be 5% of offset revenue and other transaction costs are set at USD 0.1/lbs of N and P.

Recall that the first option to provide carbon offsets involves reduced nitrogen application level. Note that the baseline here is farmers' applied nitrogen without carbon offset sales since there was no carbon market participation in this option. The potential to sell water quality offsets increases the profitability of adopting reduced nitrogen application. This option is especially profitable for continuous corn rotations on NonHEL lands but also on HEL lands. Overall this option is profitable for 21% of field parcels, representing 4.6 million acres, with offset prices of USD 3/lbs of N and USD 4/lbs of P. Reduction in offset prices decreases the number of participating parcels while an increase in prices expands participation. For offset prices from USD 1 to USD 4/lbs N and from USD 2 to USD 4/lbs P the number of participating parcels ranges between 9% and 38% across all regions.

With the possibility of selling water quality offsets, the establishment of green set-aside becomes profitable for highly erodible lands within the lowest productivity Region 3. Total number of participating parcels is 48 000 acres representing 0.2% of the total acreage in the study with offset prices of USD 3/lbs of N and USD 4/lbs of P. These offset prices may be viewed as minimum prices for farmer participation, since with lower prices the participation rate in the carbon market falls to zero. With higher offset prices of USD 4/lbs of N and USD 5/lbs of P, acreage in participating parcels increases up to 1.34 million acres

[17] Current carbon offset price is USD 3/ton of CO_2-eq and social damage estimate is USD 24/ton of CO_2-eq. So, there is a large difference between social value and market value of carbon offsets. Moreover, as shown in the analysis above even if carbon offsets prices would coincide with social value of CO2-eq emissions they would not be high enough to make adoption of the practices profitable without stacking of water quality credits.

representing 6.1% of total acreage. Hence, in the case of green set aside, allowing sale of water quality offsets increases farmer participation in carbon offset markets and as a result total carbon offset supply increases.

5.3.2 *Inter-market additionality*

Establishing vegetative buffer strips between field parcels and watercourses represents inter-market additionality, since new abatement practices and abating inputs are employed that exceed the requirements of carbon-market participation. The analysis assumes that a participating farmer establishes a buffer strip to supply both carbon and water quality offsets. Note that streamside buffer strips were not an option in the case of carbon offset markets without water quality offsets, and thus the baseline for buffer strips is the same for both carbon and water quality offset markets. It is assumed here that buffer strips cover 1.5% of the field parcel area (equivalent to 1.5 meter buffer in a field 100m*100m in size). It is further assumed that 10% of field parcels in the modelled area are bordering surface water courses, which provides a rationale for field strip establishment from a water quality viewpoint. With prevailing offset prices of USD 3/lbs of N and USD 4/lbs of P, field strip establishment would be profitable in 29% of field parcels in the modelled area. However, taking into account the criterion of field parcel proximity to surface water reduces the total acreage affected to 3% of field parcels. Due to effectiveness of vegetated field strips in nutrient runoff reduction and the relatively small profit foregone, incentives for establishment are highest on HEL lands and they are established even with the lowest nutrient credit prices (USD 1/lbs of N and USD 2/lbs of P). Total carbon offset supply increases by 4% with buffer strip establishment (their technical sequestration potential being similar to green set-aside per unit land area). Figures 10 and 11 show the total supply of nitrogen and phosphorus credits under various credit prices, aggregated across the three model regions.

Demand functions for water-quality credits were derived based on reported N and P credit prices in a water-quality trading clearinghouse established in Pennsylvania (O’Hara et al., 2012). Because of the size of the modelled regions, nutrient credit supply easily leads to oversupply of credits and a resulting reduction in equilibrium credit prices. For example, the nitrogen credit supply from reduced fertilizer use alone would reduce equilibrium market price from USD 3.7/lbs N to USD 2.1/lbs N. The phosphorus credit market is even thinner; P credit supply from reduced fertilizer use would decrease equilibrium market price from USD 4.3/lbs P to USD 1.2/lbs P.[18]

[18] Note that CO_2 offset markets are typically large and thus it seems unlikely that the price of CO_2 offset will change significantly if stacking is allowed in some particular water catchment.

Figure 10. Nitrogen credit supply (tons) under different credit prices (USD/lbs), aggregated across the three model regions

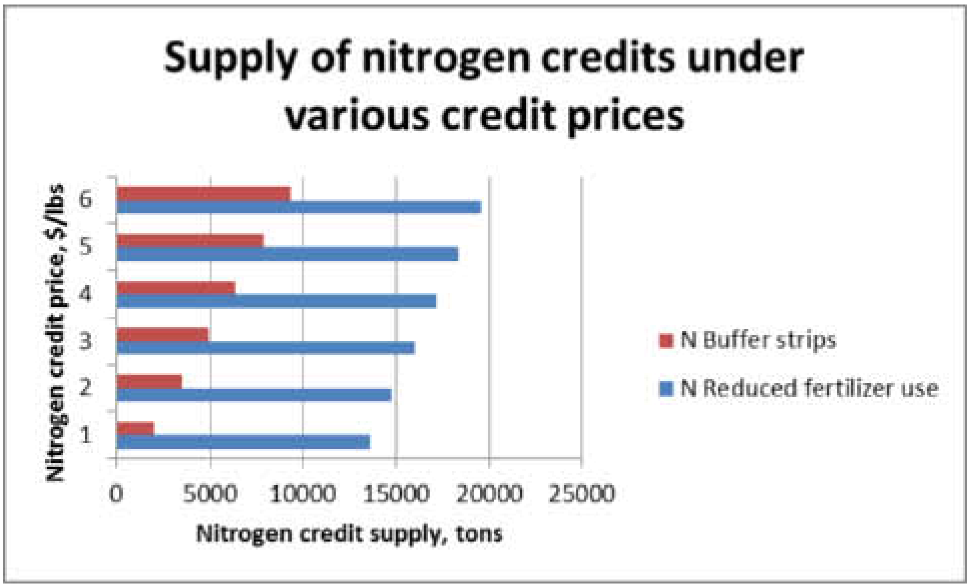
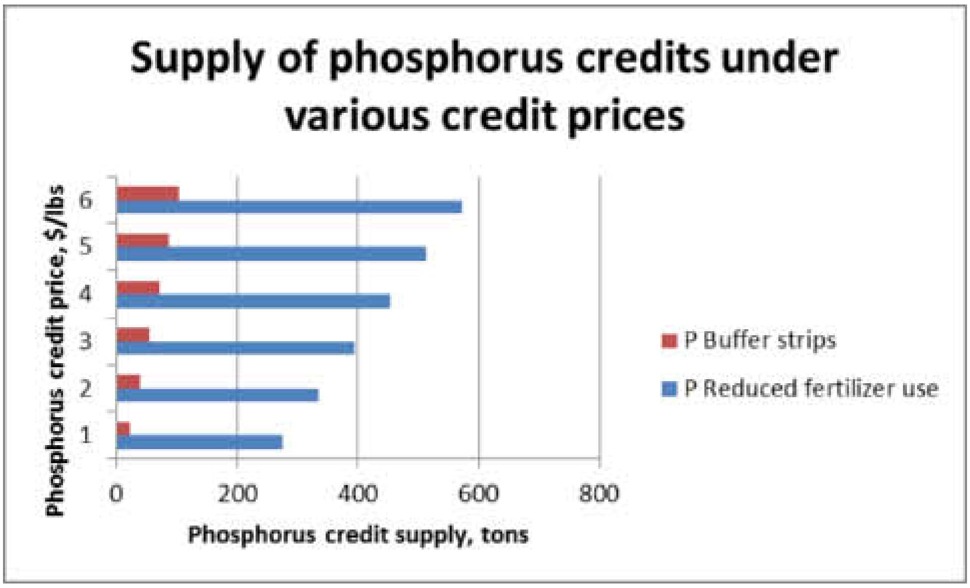


Figure 11. Phosphorus credit supply (tons) under different credit prices (USD/lbs), aggregated across the three model regions



46. Figure 12 shows farmer's profit increase (USD/acre) from the adoption of no-till as a function of CO_2-eq offset price with and without possibility of stacking water quality credits from no-till adoption. Transaction costs of carbon offsets are assumed to be USD 1/acre and water quality credit prices of USD 3/lbs of N and USD 4/lbs of P.

Figure 12. Farmer's profit increase (USD/acre) from no-till adoption as a function of carbon offset price (USD/ton of CO2-eq) with and without stacking of water quality credits

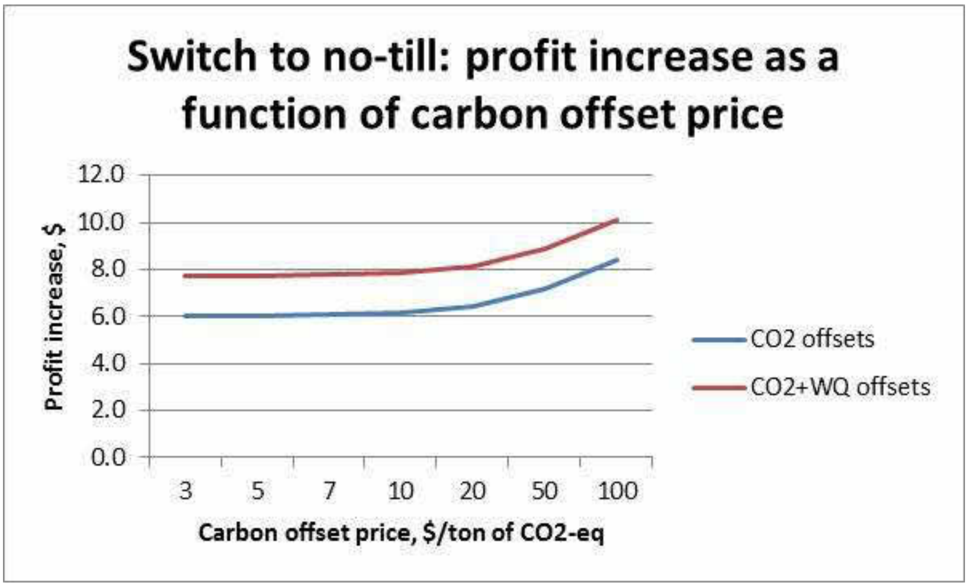


Table 3 provides a summary of aggregate results for acreage response, carbon offset and nutrient credit supply, and farm profits under various offset scenarios and environmental practices. Allowing stacking of water quality credits with carbon offsets has significant impact especially in the case of reduced fertilizer application. Recall that carbon offset revenue alone was not sufficient to compensate farmers' foregone profit and transaction costs of participating in carbon offset markets, even with zero transaction costs and a carbon offset price as high as USD 100/ton of CO_2-eq. Allowing stacking of water quality credits makes it a profitable option, with adoption on 21% of total modelled acreage. Due to the large acreage affected, this option results in the highest carbon offset and nutrient credit supply as well as the greatest increase in aggregated farm profit.

Table 3. Acreage response, carbon offsets, nutrient credits, and farm profit increase under various offset scenarios and environmental practices (carbon offset price USD 3/ton of CO2-eq, nutrient credit prices for N USD 3/lb and P USD 4/lb)

Offset scenario and environmental practice	Acreage response, million acres	Carbon offsets, 1000 tons	Nitrogen credits, tons	Phosphorus credits, tons	Farm profit increase, USD million
Carbon offsets: switch to no-till	0.8	44.4	-	-	18.4
Carbon offsets + water quality offsets: switch to no-till	1.0	49.4	23	83	20.4
Carbon offsets + water quality offsets: Green set-aside	0.05	20.7	1251	102	1.6
Carbon offsets + water quality offsets: Vegetated field strips	0.6	6.5	5140	58	12.4
Carbon offsets + water quality offsets: Reduced fertilizer use	4.6	141.2	16 408	418	74.6

Note: Acreage response for vegetated field strips represents acreage with vegetated field strips. Effective vegetated field strip area is 0.009 million acres.

Similarly, while carbon offsets alone do not provide sufficient incentives for adoption of vegetated field strips and green set-asides, the stacking of water quality credits make them profitable option. Despite the fact that only small acreages are affected, these measures contribute relatively significant levels of both carbon offsets and nutrient credits.

5.3.3 *Potential non-additional carbon offsets due to stacking*

Inter-market additionality analysis in this paper has been based on the assumption that a farmer can either participate in one market (the carbon market) or multiple markets (both carbon and water quality markets). If two markets exist, however, a farmer has usually the option to participate in one (e.g. carbon) or the other market (e.g. water-quality), and if stacking is allowed and a farmer participates in both markets then a question of inter-market additionality in offsets and credits produced arises. If a farmer can participate in just the water-quality market then part of the carbon-offsets supply would come from farmers participating in just the water-quality market and if the adoption of a particular environmental practice is profitable for them solely on the basis of credit revenue from the water-quality market then these carbon offsets are not additional to the carbon market and the stacked payment. Allowing stacking in this situation would result in non-additional credits from an inter-market additionality perspective. In fact, in most of the cases analysed in this section (reduced nitrogen application, the establishment of green set-asides and vegetated field strips) participation in the water-quality market alone supplies most of the carbon offsets and thus from an inter-market additionality viewpoint these offsets cannot be considered additional to the carbon market and the stacked payment.

6. Conclusions and policy implications

The key policy question addressed in this paper is whether a single agri-environmental practice should be allowed to earn credits from multiple environmental markets when the economic and environmental performance of these markets is considered. The case of interlinked carbon offset and water quality credit market is considered. The choice of alternative soil carbon sequestration practices and incentives to participate voluntarily in carbon offset markets is examined with and without a possibility of selling water quality credits. Farmers' incentives to participate in environmental credit markets are affected by opportunity costs of practice adoption, revenue from environmental credits, and transaction costs.

A theoretical analysis shows that credit stacking increases farmer participation in carbon offset markets and through increased participation provides additional environmental services. Credit stacking also provides incentives for adoption of more environmentally effective practices due to credit revenue from water quality offsets. It was further shown that environmental markets are interlinked so that credit price changes in one market impact credit supply and the equilibrium credit price in another market.

The empirical application of the model was based on data estimates for the U.S. Corn Belt region. In the empirical application, the Baseline scenario without carbon and water quality offset markets was analysed first. Both corn yields and farmers' profits exhibit significant heterogeneity between the regions. Net GHG emissions are negative in each model region, since CO_2-eq emissions from cultivation practices and fertilizer application are smaller than soil carbon sequestration. All runoff types (sediment, nitrogen, and phosphorus) vary between regions and especially between highly and non-highly erodible lands. An ex-post social welfare calculation shows that agriculture's aggregate net environmental impact is negative but the profitability of production makes ex-post social welfare clearly positive.

Farmers had several options to sequester carbon and reduce GHG emissions in order to supply carbon offsets: reduced fertilizer application, a switch from conventional and reduced tillage to no-till, establishment of green set-asides and the establishment of streamside buffer strips. The following conclusions arise from analysis related to carbon offset markets without the possibility of stacking water

quality credits. Provision of CO_2-eq offsets through a reduction in applied nitrogen is not profitable without water quality offsets, since the impact of nitrogen application on N_2O emissions is relatively small and as a result offset supply and offset revenue are small relative to profit foregone even with highest offset prices. Similarly due to significant profit foregone the establishment of green set-asides is not profitable without water quality offsets. Only a switch from conventional tillage and reduced tillage to no-till is profitable in some cases. However, carbon offset prices and transaction costs have a significant impact on the number of participating parcels and current CO_2-eq offset prices do not provide strong incentives for farmers to participate in carbon offset markets without stacking of water quality offsets.

When farmers are allowed to stack water quality credits the profitability of carbon sequestration practices increases. Reduced nitrogen application levels becomes the profitable option and 21% of field parcels participate in the market when water quality offset prices are at current levels of USD 3/lb for N and USD 4/lb for P. Also the establishment of green set-asides and streamside buffer strips becomes profitable in lowest productivity and highly erodible lands with current offset prices. Farmers’ high participation rate may, however, result in an oversupply of nutrient credits and significantly affect equilibrium prices of nutrient credits. For example, N credit supply from the reduced fertilizer use alone would reduce equilibrium market price for N from USD 3.7/lb down to USD 2.1/lb. The phosphorus credit market is even smaller and the P credit supply from reduced fertilizer use would decrease equilibrium market price from USD 4.3/lb down to USD 1.2/lb.

To conclude, both the theoretical and the empirical analysis show that allowing stacking of water quality credits provides additional environmental services through increased participation of farmers in carbon offset markets and through increased adoption of environmental practices that are effective in promoting both carbon offsets and nutrient credits. The agri-environmental practices considered in this study (reduced fertilizer use, vegetated field strips, green set-asides and no-till) are widely used across OECD countries to achieve environmental goals for water quality and greenhouse gas mitigation. While their opportunity costs and effectiveness in promoting supply of carbon offsets and nutrient credits vary over space, some general conclusions and policy implications can be drawn. First, current CO_2-eq offset prices do not necessarily compensate the foregone profit resulting from practice adoption. Allowing stacking of water quality credits and potentially government incentive payments makes adoption more profitable. Stacking of government agri-environmental payments with environmental credits would increase farmer participation in both government conservation programmes and environmental markets while providing additional income for farmers. Secondly, if environmental markets are small, with limited local demand for credits, then stacking may lead to an oversupply of credits and resulting declines in equilibrium credit prices. As a consequence, environmental practices that may have been profitable to adopt with credits from one market may require sale of credits to several markets in order to cover adoption costs of the practice.

ANNEX A. THEORETICAL FRAMEWORK OF AGRICULTURAL SUPPLY OF CARBON AND WATER-QUALITY OFFSETS

Baseline cultivation: no offset markets

Consider cultivation under heterogeneous land productivity illustrated for simplicity by two productivity classes, high and low. Land productivity within both classes is homogenous. Each farmer owns one field parcel so that the total number of land equals total number of farmers. Both land productivities can be cultivated using two alternative tillage methods: conventional tillage or no-till. The crop is produced using fertilizer, *l*, as variable productive input and a set of other inputs (seeds, labor, and capital) that can conveniently be regarded constant per hectare (denoted by M_i). Crop yield as a function of fertilizer input varies depending on the cultivation method.

Tillage method is denoted by *i*, $i = 1{,}2$ (conventional and no-till cultivation, respectively). Let *l* denote fertilizer input, then the crop yield, *y*, as a function of land productivity, *q*, is defined by $y^i = f^i(l,q)$. Let *p* denote crop prices and *c* price of fertilizers. In the baseline with no offset markets, farmers choose optimal fertilizer application, $l_i^0(p,c;q)$, based on relative prices and allocate each land productivity to the tillage method which produces highest profits. It is assumed that no-till is more suitable for high productivity land and conventional tillage for low productivity land.

Under these assumptions and based on the optimal use of inputs, the choice of tillage method and land productivities are related as follows:

land productivities *q* = high: $\pi^1(l_1^0(p,c);q) \le \pi^2(l_2^0(p,c);q)$ allocated to no-till cultivation,

land productivities *q* = low: $\pi^1(l_1^0(p,c);q) > \pi^2(l_2^0(p,c);q)$ allocated to conventional tillage.

This baseline is the point of departure for the analysis of agricultural production in the presence of carbon offset markets.

Cultivation under voluntary participation in carbon markets

It is assumed that the farmer participates in voluntary carbon markets in order to receive additional revenue. Let the unit price of offsets be *b*. If a farmer wishes to produce offsets and participate in carbon markets, three measures are in principle available to produce carbon offsets from crop cultivation.

First, the farmer may reduce fertilizer intensity below the baseline optimum under both tillage methods. This yields carbon offsets from reduced N_2O emissions. If ε denotes N_2O emissions from one unit of fertilizer use, then the amount of offset created is given by $\varepsilon(l_i^0 - l_i)$.

Second, if the farmer applies conventional tillage, then offsets can be produced by switching to no-till. Let *k* denote the amount of offsets generated by this switch and define $k = (E + \hat{k})$, where *E* is the reduction in carbon emissions from cultivation (no-till entails lower fossil fuel use) and $\hat{k}$ is the average annual carbon sequestration over a 50-year period under no-till.

Third, the farmers may allocate some field parcels to long-term green set-aside in order to sequester carbon. Let *H* denote the average annual carbon sequestration under green set-aside per hectare over a 50-year period and *W* its establishment cost. For this land the establishment cost per hectare is *W* and annual revenue from offsets sold *bH*, and the profits are denoted by $\hat{\pi} = bH - W$.

Participation in carbon offset markets involves transaction costs. Transaction costs are comprised of costs of participating in the offset market and additional, farmer-specific transaction costs, representing the farmer's cost of gathering and producing information on offset markets and measures needed to produce offsets. Farmer's *j* total transaction costs are denoted by θ_j. Thus, while farmers in both land productivity classes face the same yields and relative costs, farmers differ in their transaction costs. We arrange farmers in each productivity class by the size of transaction costs from lowest to the highest ones, that is, $\theta_1 < \theta_2 < ... < \theta_s$. Therefore, it will not be optimal for all farmers to participate in offset markets.

The comprehensive economic problem of the farmer *j* in productivity class *q* can be expressed for conventional and no-till, respectively, as follows:

$$\pi^i = pf^i(l_i;q) - cl_i + b\varepsilon(l_0^i - l_i) - M_i + \sigma_i bk_i - \theta_j \,,$$

where sigma is a 0 - 1 dummy variable with $\sigma_i = 0$ for *i* =1, and $\sigma_i = 1$ for *i* = 2 to address the assumption that under conventional tillage no carbon sequestration takes place.

The optimal choice of inputs is given by $l_i^* = l_i^*(p,c,b)$ and it is independent of the transaction cost. Given the fact that offset price is positive and fertilizer intensity is a continuous choice variable, it holds that $l_i^*(p,c,b) < l_i^0(p,c)$. Once optimal fertilizer intensity is known, farmer *j* in land productivity class *q* chooses the tillage method. *In the high-productivity class*, no-till was the baseline cultivation technology and possibility of selling carbon credits reinforces its superiority relative to conventional tillage. *In the low-productivity class*, carbon offset improve returns to no-till cultivation relative to conventional tillage, yet both benefit from carbon market. Depending on the revenue of the generated offsets under no-till relative to those generated under conventional tillage, either no-till cultivation is introduced or farmers continue employing conventional tillage. (Technically, the choice of technology under carbon markets is made by condition $\pi^1(l_1^*(p,c,b)) \geq (<) \pi^2(l_2^*(p,c,b))$ and it is independent of the transaction cost, which is the same for both cultivation technologies.

Participation in the carbon offset market

In the high productivity class, farmer *j* participates in the carbon offset market provided that $\pi^2(l_2^0(p,c);q) < \pi^2(l_2^*(p,c,b);q) - \theta_j$. Recall, the transaction costs were indexed from the lowest to the highest one. Therefore, one can infer the indifference between participation and nonparticipation as follows: $\hat{\theta}_j = \pi^2(l_2^*(p,c,b);q) - \pi^2(l_2^0(p,c);q)$. For transaction costs below this critical value, farmers participate in offset markets and for costs above this value they do not participate but continue with the baseline cultivation.

In the low productivity class, conventional tillage was the baseline technology but farmers may adopt no-till if it provides higher revenue and participation is profitable. Based on the two alternative possibilities for the choice of tillage method, the participation decision can be expressed as follows. Farmer *h* participates in the carbon market if $\pi^1(l_1^0(p,c);q) < \pi^i(l_i^*(p,c,b);q) - \theta_h$. This condition leads to the same indifference condition as was found in the high productivity class but this time for both tillage

methods. Suppose first that no-till is the superior tillage method for farmers who participate in the carbon market. Then the farmers with lower transactions costs than $\hat{\theta}_h$ participate in carbon markets using no-till but farmers with higher transaction costs than $\hat{\theta}_h$ will not participate and employ conventional tillage. If instead conventional tillage is superior also under carbon markets, both participating and non-participating farmers stick to conventional tillage and only reduce their fertilizer use intensity.

The supply of offsets, *S*, to the carbon markets from crop production can be defined as the sum over participating farmers in high and low productivity classes. Let the number of farmers participating in carbon market be *m* = 1,…j and that in the low productivity class be *n* =1,…*h*. Furthermore, to avoid unnecessary notation, assume that all low productivity farmers adopt either conventional tillage or no-till (that is, green set-aside is not adopted). As production conditions are identical, we can express the total supply of offsets as follows depending on which cultivation technology is superior in low productivity class:

$$S = m\left[\varepsilon(l_2^0 - l_2^*) + k\right] + n\left[\varepsilon(l_i^0 - l_i^*) + \sigma_i k\right],$$

where $\sigma_i = 0$ if conventional cultivation is optimal under carbon markets, and $\sigma_i = 1$ if no-till becomes optimal under carbon markets. Thus, the offset supply is generated by reduced fertilizer use and possibly by a discrete shift from conventional tillage to no-till in participating lands from low productivity class and the number of participating farms.

The impact of an increase in carbon offset price *b* on supply of carbon offsets can be expressed as follows:

$$\frac{\partial S}{\partial b} = \frac{\partial m}{\partial b}\left[\varepsilon(l_2^0 - l_2^*) + k\right] + \frac{\partial n}{\partial b}\left[\varepsilon(l_i^0 - l_i^*) + \sigma_i k\right] - \left[m\frac{\partial l_2}{\partial b} + n\frac{\partial l_i}{\partial b}\right] > 0\,.$$

The total impact is positive, as number of participating farmers increases in *b* (determined by the above indifference relation) and fertilizer intensity decreases in *b* (can be shown by resorting comparative statics of the model). Hence, the total supply of carbon offsets from crop production is an increasing function of carbon price.

The focus has thus far been on crop production. On low productivity land the role of green set-aside must be accounted for. If green set-aside is profitable enough, that is, a farmer has $\pi^i(l_i^0(p,c);q) < \hat{\pi} - \theta_h > \pi^i(l_i^*(p,c,b);q) - \theta_h$, then green set-aside is established.

Water quality offsets as function of carbon offset supply

Farmers’ production choices regarding the supply of carbon offsets have effects also on water quality. Reduced fertilizer use decreases nutrient runoff, green set-aside planted to perennial grasses with zero fertilizer use reduces considerably all forms of runoff, and no-till decreases erosion, nitrogen runoff and particulate phosphorus runoff (but may increase dissolved reactive phosphorus runoff).

It is assumed in the analysis that a farmer can either participate in one market (the carbon market) or multiple markets (both carbon and water-quality markets) and that the carbon market is primary market in this analysis and thus participating in just the water-quality market is not possible. Two alternative interpretations of additionality in environmental services are employed. *Financial additionality* refers to an increase in the number of participating farmers in the carbon markets due to possibility of stacking water

quality credits. *Inter-market additionality* requires that farmers who are willing to sell water-quality offsets need to employ abatement practices that go beyond those required by participation in the carbon offset market. One possibility to increase the use of abating inputs is to establish vegetated field strips (buffer strips) between waterways and the field parcel in order to reduce both soil erosion and nutrient runoff.

To examine both interpretations of additionality, let Z denote the (weighted) sum of nutrient runoff, defined in nitrogen equivalents as follows: $Z = z^N + \alpha z^P$, where α translates phosphorus runoff to nitrogen equivalents.

Financial additionality

Let the price of water quality offsets be τ. In this case, the baseline in water quality offset markets is the same as the baseline in the carbon offset market, $z^i(l_i^0;q)$, because a farmer would not participate in carbon offset markets under the prevailing offset price. Previously it was demonstrated that in high-productivity land farmers with transaction costs higher than $\hat{\theta}_j$ do not participate in the carbon markets and in low-productivity land the respective critical transaction costs were θ_h. Consider a farmer *x* in the high-productivity class and farmer *r* in the low-productivity class with transaction costs higher than the critical transaction costs ($\theta_x > \hat{\theta}_j$ and $\theta_r > \hat{\theta}_h$). To highlight the role of water quality offsets, define now a new profit function as a product of profits from cultivation and participation in the carbon market and the revenue obtainable from this solution when water-quality co-benefit credits can be sold:

$\Pi^2(b,\tau) = \pi^2(p,c.b) - \theta_x + \tau(z^2(l_2^0) - z^2(l_2^*))$ (high-productivity land)

$\Pi^i(b,\tau) = \pi^i(p,c.b) - \theta_r + \tau(z^1(l_1^0) - z^i(l_i^*))$ (low-productivity land)

New indifference relations for participation are now obtained. For high-productivity land, the critical value can be expressed as $\hat{\theta}_x = \pi^2(p,c.b) + \tau(z^2(l_2^0) - z^2(l_2^*)) - \pi^2(p,c)$. A comparison with previous indifference relation shows that the new critical level of transaction cost has increased by exactly the amount of water quality benefits, $\tau(z^2(l_2^0) - z^2(l_2^*))$. For low quality land the respective indifference is defined by $\hat{\theta}_r = \pi^i(p,c.b) + \tau(z^1(l_1^0) - z^i(l_i^*)) - \pi^1(p,c)$. Let $\tilde{m}$ and $\tilde{n}$ denote the number of farmers participating in carbon markets from high and low-productivity lands, respectively, when stacking is allowed. Then the supply of water quality offsets, *β*, from increased participation in carbon markets is defined by

$$\beta = (\tilde{m} - m)\left[z^2(l_2^0) - z^2(l_2^*)\right] + (\tilde{n} - n)\left[z^1(l_1^0) - z^i(l_i^*)\right]$$

Supply function of water quality offsets depends on its own price and carbon price. The impact of an increase in the price of water quality offsets and water quality offset supply is

$$\frac{\partial \beta}{\partial \tau} = \frac{\partial \tilde{m}}{\partial \tau}\left[z^2(l_2^0) - z^2(l_2^*)\right] + \frac{\partial \tilde{n}}{\partial \tau}\left[z^1(l_1^0) - z^i(l_i^*)\right] > 0$$

Hence, supply of water quality offsets is increasing in offset price thanks to increased participation. This is an intuitive outcome. For the impact of a change in the carbon offset price on water quality supply we have,

$$\frac{\partial \beta}{\partial b} = -\frac{\partial m}{\partial b}\left[z^2(l_2^0) - z^2(l_2^*)\right] - \frac{\partial n}{\partial b}\left[z^1(l_1^0) - z^i(l_i^*)\right] - (\tilde{m} - m)\frac{\partial z^2(l_2)}{\partial l_2}\frac{\partial l_2}{\partial b} - (\tilde{n} - n)\frac{\partial z^i(l_i)}{\partial l_i}\frac{\partial l_i}{\partial b} = ?$$

The impact of higher carbon price is generally ambiguous due to two opposing mechanisms. A higher carbon offset price invites more farmers to participate in the carbon market. This reduces the number of participating farmers in water-quality offset markets, as the difference $(\tilde{m} - m)$ and $(\tilde{n} - n)$ become smaller. This impact is counter-affected by the further decrease in fertilizer use intensity, which increases water quality offsets for a given water offset price. If the former dominates, water offset supply decreases, but if the latter one dominates then it increases. In the special case they may outweigh each other and supply of carbon offsets does not change. Whether the increased revenue from water-quality offsets dominates depends much on the distribution of transaction costs. If at the tail of distribution the differences between transaction costs increase sharply, water quality offset supply decreases. Thus, in the general case an increase in carbon price may increase, decrease or maintain the supply of water-quality offsets.

Intermarket additionality

A participating farmer may optimize fertilizer intensity and tillage method for both carbon offset markets and water quality offset market in order to achieve greater revenue stream from offsets. To this end the farmer is allowed to establish a vegetated field strip to reinforce the production of both carbon and water quality benefits. The baseline for the carbon markets is the same as defined in the above interpretation of additionality ($\varepsilon(l_i^0 - l_i)$), but the baseline for the water quality offset markets changes. This baseline must account for the fact that farmers participating in carbon markets produce additional benefits above the level determined by optimal input use for participation in carbon markets in the absence of water quality offsets. The new baseline in water quality offset markets is therefore $z^i(l_i^*;q) - z^i(l_i(p,c,b);q)$, where l_i^* represents fertilizer intensity under participation in carbon markets and in the absence of water quality offset markets.

In order to define revenue to farmers from participation, the following assumptions are specified for vegetated field strips. First, carbon sequestered in vegetated strips is similar to green set-aside per unit-area, but only applies across a smaller land area. Let v_i denote the share of parcel allocated to field strips. Then the carbon sequestered is v_iK. The same field strip generates water quality offsets by fixing nutrients already released from the fields, so that runoff from fields can be expressed as $z^i(l_i, v_i)$. Cost of establishing the field strip is v_iW, where W was defined above. Drawing on this notation, participating farmers with high productivity land solve the problem

$$\pi^i(\tau) = (1 - v_i)\left[pf^i(l_i;q) - cl_i + b\varepsilon(l_0^i - l_i)\right] - M_i + bv_iK - v_iW + \sigma_i bk_i - \theta_n + \tau(z^i(l_i^*) - z^i(\hat{l}_i, v_i))$$

where $\hat{l}_i = (1 - v_i)l_i$ denotes the cultivated part of the field parcel.

The farmer chooses both fertilizer intensity and filter strip width so as to maximize profits. The optimal fertilizer intensity is now smaller than in the case where fertilizer use was optimized only in terms of carbon benefits and is denoted by double star: $l_i^{**}(p,c,b,\tau) < l_i^*(p,c,b)$. The vegetated field strip width depends on the establishment cost of the practice, the foregone income lost from cultivation and on the revenue from carbon offsets and water quality offsets. A field strip is only established provided the return to its establishment is high enough. If a field strip is established, only part of the parcel is allocated to it, because the number of produced water quality offsets decreases with the width of the field strip. Therefore,

the size of the field strip is $0 \le v_i^*(b,\tau) < 1$. Both choices provided indicate increased use of abating inputs and higher quality of the project.

As before, an indifference relation can be developed by comparing profits under nonparticipation and participation. Given that farmers are allowed to optimize both fertilizer application and field strip establishment under a positive water quality offset price, the number of participating farmers increase. The indifference relation between participation and nonparticipation can be expressed for high productivity land as follows: $\hat{\theta}_s = \pi^2(p,c,b) + \tau(z^2(l_2^0) - z^2(l_2^*)) - \pi^2(p,c.b) + \Omega^2$. In this relation $\Omega^2 > 0$ and comprises net revenue from field strip adoption minus the net loss of profits from crop production due to further lowered fertilization rates. For low productivity land the same indifference condition reads $\theta_r = \pi^i(p,c.b) + \tau(z^1(l_1^0) - z^i(l_i^*)) - X^i$, where $X^i > 0$ and reflects the same net revenue items as in the high productivity land. Thus, for high and low productivity land, $\hat{\theta}_s > \hat{\theta}_x$ and $\hat{\theta}_\eta > \hat{\theta}_r$, respectively, so that the project supplies more carbon offsets and water quality offsets than the previous cases.

Denoting the amount of participating farmers by bar above *m* and *n*, the carbon offset supply can be defined as follows:

$$\hat{S} = \overline{m}\left[\varepsilon(l_2^0 - (1-v_2^*)l_2^{**}) + k + v_2^* K\right] + \overline{n}\left[\varepsilon(l_i^0 - (1-v_i^*)l_i^*) + \sigma_i k + v_i^* K\right]$$

Omitting the details, supply of carbon offsets increases with carbon price, $\partial\hat{S}/\partial b > 0$, just as before. Furthermore, carbon offset supply increases also with the higher water-quality offset price $\partial\hat{S}/\partial\tau > 0$. In both cases, the number of farmers participating in carbon markets increases and farmers increase use of abating inputs.

Based on farmers' choices, the supply of water-quality offsets is as follows

$$\hat{\beta} = (\overline{m} - m)\left[z^2(l_2^*) - z^2(\hat{l}_2^{**}, v_2^*)\right] + (\overline{n} - n)\left[z^1(l_1^*) - z^i(\hat{l}_i^{**}, v_i^*)\right].$$

Water-quality offset supply increases in offset price via two mechanisms: the number of participating farmers increases and farmers increase their use of abating inputs:

$$\frac{\partial\hat{\beta}}{\partial\tau} = \frac{\partial\overline{m}}{\partial\tau}\left[z^2(l_2^*) - z^2(\hat{l}_2^{**}, v_2^*)\right] + \frac{\partial\overline{n}}{\partial\tau}\left[z^1(l_1^*) - z^i(\hat{l}_i^{**}, v_i^*)\right] - (\overline{m} - m)\frac{dz^2(\hat{l}_2^{**}, v_2^*)}{dl_2} - (\overline{n} - n)\frac{dz^2(\hat{l}_{1i}^{**}, v_i^*)}{dv_i} > 0$$

An increase in the carbon offset price has an impact through multiple channels:

$$\frac{\partial\hat{\beta}}{\partial b} = \left(\frac{\partial\overline{m}}{\partial b} - \frac{\partial m}{\partial b}\right)\left[z^2(l_2^*) - z^2(\hat{l}_2^{**}, v_2^*)\right] + \left(\frac{\partial\overline{n}}{\partial b} - \frac{\partial n}{\partial b}\right)\left[z^1(l_1^*) - z^i(\hat{l}_i^{**}, v_i^*)\right]$$
$$- (\overline{m} - m)\frac{dz^2(\hat{l}_2^{**}, v_2^*)}{db} - (\overline{n} - n)\frac{dz^i(\hat{l}_i^{**}, v_i^*)}{db} = ?$$

A higher carbon price increases the number of farmers who would participate even without water-quality offsets, which decreases water quality offset supply. But it also increases the number of those farmers for whom participation in the carbon market is profitable only with the water quality offset. Depending which impact dominates, this term is either positive or negative. Higher carbon price increases

the use of both abating inputs, which tends to increase supply of water-quality offsets. Again, all possibilities are present: supply of water-quality offsets may increase, decrease or remain constant. Relative to the previous case, however, an increasing supply impact is more likely.

ANNEX B.

36 CROPPING SYSTEMS DEFINED IN THE US CASE-STUDY ANALYSIS AND THEIR DESCRIPTIVE ABBREVIATION FOR REGION/CROP/TILLAGE/ERODIBILITY COMBINATIONS

Table B.1. Descriptive abbreviation of regions, crops, cultivation method, and soil erodibility combinations

Rotation: (i) RCB (Corn-Soybean) (ii) RCCC (Continuous Corn)	**Tillage**: (i) CNV (Conventional tillage) (ii) RED (Reduced tillage) (iii) NLL (No-till)	**REAP region** and erodibility: (i) Eastern Corn Belt – CBM0503 (ii) Central Corn Belt – CBM0704 (iii) Western Corn Belt – CBM1009 (i) H (Highly erodible) (ii) N (Non-highly erodible)
RCB	CNV	CBM0503H
RCB	RED	CBM0503H
RCB	NLL	CBM0503H
RCCC	CNV	CBM0503H
RCCC	RED	CBM0503H
RCCC	NLL	CBM0503H
RCB	CNV	CBM0503N
RCB	RED	CBM0503N
RCB	NLL	CBM0503N
RCCC	CNV	CBM0503N
RCCC	RED	CBM0503N
RCCC	NLL	CBM0503N
RCB	CNV	CBM0704H
RCB	RED	CBM0704H
RCB	NLL	CBM0704H
RCCC	CNV	CBM0704H
RCCC	RED	CBM0704H
RCCC	NLL	CBM0704H
RCB	CNV	CBM0704N
RCB	RED	CBM0704N
RCB	NLL	CBM0704N
RCCC	CNV	CBM0704N
RCCC	RED	CBM0704N
RCCC	NLL	CBM0704N
RCB	CNV	CBM1009H
RCB	RED	CBM1009H
RCB	NLL	CBM1009H
RCCC	CNV	CBM1009H
RCCC	RED	CBM1009H
RCCC	NLL	CBM1009H
RCB	CNV	CBM1009N
RCB	RED	CBM1009N
RCB	NLL	CBM1009N
RCCC	CNV	CBM1009N
RCCC	RED	CBM1009N
RCCC	NLL	CBM1009N

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