HOW SCALE AND SCOPE OF ECOSYSTEM MARKETS IMPACT PERMIT TRADING:
EVIDENCE FROM PARTIAL EQUILIBRIUM MODELING IN THE CHESAPEAKE BAY WATERSHED

by

Natalie Loduca

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STATEMENT OF COMMITTEE APPROVAL

Dr. Carson Reeling

Department of Agricultural Economics

Dr. Thomas Hertel

Department of Agricultural Economics

Dr. Jing Liu

Department of Agricultural Economics

Approved by:

Dr. Nicole Olynk Widmar

Graduate Chair of the Department of Agricultural Economics

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATIONS

BMPs	Best Management Practices
CAC	Command and Control
CES	Constant Elasticity of Substitution
CET	Constant Elasticity of Transformation
CREP	Conservation Reserve Enhancement Program
CBW	Chesapeake Bay Watershed
CWA	Clean Water Act
ECHO	Enforcement and Compliance History Online
EPA	Environmental Protection Agency
EPRI	Electric and Power Research Institute
GHG	Greenhouse Gas
LCA	Least-Cost Abatement
MAC	Marginal Abatement Cost
N	Nitrogen
NPS	Nonpoint Source
NPDES	National Pollutant Discharge Elimination System
Nr	Reactive Nitrogen
PS	Point Source
REAP	Regional Environmental and Agricultural Programming
SIMPLE-G	Simplified International Model of agricultural Prices, Land use and the Environment, on a Grid
TMDL	Total Maximum Daily Load
USDA	United States Department of Agriculture
WIPS	Watershed Implementation Plans
WQT	Water Quality Trading
WQTPs	Water Quality Trading Programs
WRP	Wetlands Reserve Program
WWTP	Wastewater Treatment Plant

ABSTRACT

This study uses the Simplified International Model of agricultural Prices, Land use and the Environment, on a Grid (SIMPLE-G), a partial equilibrium model of agricultural production, to explore how the scale and scope of environmental quality markets influence farm-level production decisions and market performance. I simulate how permit trading affects producers' input use decisions, and ultimately pollution emissions, by modifying the supply nest structure of the model to include water quality permits as an additional output from agricultural production. Conservation practices improving water quality may also result in ecosystem co-benefits (e.g., reduced greenhouse gas emissions and habitat provision). Hence, I extend SIMPLE-G to quantify these co-benefits and simulate the effects of allowing conservationist producers to "stack" permits (i.e., to supply multiple permit types for each co-benefit). I find that, overall, permit production increases with the scale and scope of the markets. At the smallest market size—which allows trading only within 8-digit hydrological unit code watersheds—unintended policy implications arise as the stacked markets cause one conservation practice to crowd out the other. Meanwhile, the largest market—which allows trading across the Chesapeake Bay Watershed—produces nitrogen permits more efficiently which may lead to less of the secondary permits in comparison to other market configurations. The results of this study support the Environmental Protection Agency's urging of the expansion of the scale and scope of ecosystem markets.

INTRODUCTION

Nonpoint source (NPS) pollution from farmland is diffused across land at heterogeneous rates depending on several factors, including soil type, weather conditions, and fertilizer application timing (Chen et al., 2019). NPS nutrient runoff from agricultural fields is a primary source of water pollution in the form of reactive nitrogen, which leads to excessive nutrient concentrations in nearby water systems, causing eutrophication. This can create large hypoxic zones and harmful algal blooms, resulting in biodiversity loss and habitat degradation (Galloway et al., 2003).

The US Clean Water Act, established in 1972, is a federal framework for regulating industrial point source (PS) polluters, and water pollution regulation is usually deferred to the states in terms of enforcement practices. In contrast, voluntary state programs are the main basis for NPS pollution management. Despite the significant role of states in implementing and enforcing water management policy, watersheds in the United States spread over multiple state boundaries, generating a common pool resource problem. This, combined with a lack of cohesive enforcement measures and the large geographical scale of many water systems, results in an over-production of nutrients that enter surface waters.

There is increasing interest in market-based policies for managing NPS nutrient pollution—especially water quality trading (WQT). Under this policy, a regulator determines a binding cap that defines the maximum allowable amount of emissions a producer can release. Permits are allocated to producers in accordance with this cap, and if producers release emissions above this amount, they must purchase permits from producers who have an excess of permits due to adoption of abatement measures. The US Environmental Protection Agency (EPA) Assistant Administrator for Water, David Ross, recently issued a memo to EPA regional administrators encouraging states to establish WQT markets to address water pollution, and 11 states have established trading programs so far (Ross, 2019).

Of the states with trading programs, only Pennsylvania, Virginia and Connecticut have seen significant trading (Davies, 2019). Most watersheds are unsuitable for WQT since unregulated NPSs tend to dominate pollution flows (Ribaudo and Nickerson, 2009). Further, market performance is often limited by restrictions on scale and scope. Here, "scale" refers to the physical area over which polluters can trade. "Scope" refers to the type of activities that can generate tradable permits. Indeed, trading can typically only take place among firms in the same state and watershed. Further, individual markets typically allow polluters to trade only a single type of permit (e.g., nitrogen permits), even though the activities that generate these permits can also affect other environmental outcomes. Polluters typically cannot collect permit revenues for multiple ecosystem benefits generated by the same activity.

Expanding the geographic scope of trades or allowing producers to collect payments for multiple environmental benefits produced by water quality improvements through so-called "credit-stacking" programs can improve market performance (Woodward, 2011). In addition to encouraging the establishment of WQT programs in general, the US EPA has also encouraged expanding both the scope and scale of WQT nationally (Ross, 2019). Though existing studies examine the design of WQT in general (Horan & Shortle, 2011; Lankoski et al., 2015; Reeling et al., 2018b; Shortle & Horan, 2013; Woodward & Kaiser, 2002), no study examines market design for large-scale trading programs. Hence, the goal of this research is to examine the effects of varying WQT market scale and scope on environmental outcomes at a regional scale.

I use the Simplified International Model of agricultural Prices, Land use and the Environment, on a Grid model (SIMPLE-G), a large-scale partial equilibrium model, to simulate the impact of a hypothetical WQT market in the Chesapeake Bay Watershed (CBW). The CBW comprises 64,000 square miles of complex water systems traversing New York, Pennsylvania, Maryland, Delaware, Virginia, West Virginia, and the District of Columbia. Each year since 1985, the CBW has been affected by an average hypoxic zone of 1.7 cubic miles (Miller and Duke, 2013). The Total Maximum Daily Load (TMDL) established by the EPA for the CBW is an aggregate of 92 smaller TMDLs, but there has been relatively little success in achieving this goal (Environmental Protection Agency, 2018). In order to ensure progress moving forward, The Chesapeake Bay Program Partnership, an inter-state initiative, created The Chesapeake Watershed Agreement to define Water Implementation Plans (WIPs) at the state level (Environmental Protection Agency, 2014).

Similar models analyze nutrient reduction in the agricultural sector to aid the EPA's Mississippi River Gulf of Mexico Watershed Nutrient Task Force and understand unintended policy consequences (Liu et al., 2018; Marshall et al., 2018), while smaller regional studies analyze the feasibility of WQT in specific regions (Corrales et al., 2014). The benefit of using SIMPLE-G is the ability to assess the effects of market design choices across large spatial scales using a coupled economic-biophysical model.

This study extends work by Liu et al. (2018), who use SIMPLE-G to solve for the cost-effective allocation of various best management practices (BMPs) (including reductions in nitrogen fertilizer and wetland installation) to reach a set reduction in nitrogen leaching in the Mississippi River Basin. I extend this work by explicitly modeling a WQT market in which producers optimally choose to supply nutrient permits to regulated PSs, where permits are generated from the adoption of two different BMPs: (i) a reduction in applied nitrogen fertilizer and (ii) wetland construction. The first practice is a nutrient management plan that prevents a portion of applied nitrogen fertilizer from leaching into the soil and transferring to surface water systems. The second BMP is considered an "edge-of-field" practice in that it is meant to capture water draining from the field and foster denitrification.

These BMPs have ancillary benefits, also known as co-benefits, which result in additional improvements to environmental quality. These co-benefits include greenhouse gas emissions abatement (since a percentage of applied nitrogen fertilizer volatizes and enters the atmosphere as nitrous oxide, a potent greenhouse gas with nearly 300 times the global warming potential of carbon dioxide; Environmental Protection Agency, 2017) and wildlife habitat provision (since wetland construction can be beneficial for migratory birds; De Steven & Lowrance, 2011). I assume producers who adopt these BMPs can also generate and collect revenue from greenhouse gas permits and wildlife habitat permits. By including these co-benefits as additional revenue streams within SIMPLE-G, I explore the effects of changing the scope of markets via "credit-stacking."

I find that aggregate nitrogen abatement increases with trading as both the scope and scale of the WQT market increase. When the scale of markets is relatively small, as is the case when polluters can only trade with others within individual 8-digit Hydrological Unit Code (HUC) markets, I find tradeoffs between nitrogen abatement from wetland and cropland that demonstrate a crowding-out effect. Specifically, the extra revenues from wetland-related permits crowd out potential abatement from cropland activity, ultimately undermining the final environmental outcomes. In the HUC market case, it is more profitable to focus on wetland construction and expand crop production as opposed to reducing fertilizer application on cropland.

Under larger-scale markets—i.e., when polluters can trade with others within states or even across the entire CBW—an increase scope leads to an increase in permit generation across all permit types. Spatial patterns occur that highlight the differences in the leaching potential of rainfed and irrigated land. As the scale of the market expands, there is a shift in nitrogen fertilizer application from rainfed cropland to irrigated cropland. This demonstrates the comparative advantage that crop producers on rainfed cropland benefit from due to the higher return from reducing nitrogen fertilizer on rainfed cropland versus irrigated cropland. Within the CBW market, crop producers can better target reductions of nitrogen fertilizer on regions with higher returns due to higher leaching potential. Landowners are able to construct wetland area on grid cells with higher nitrogen removal rate potential. These two factors lead to the CBW market case having higher production of the primary nitrogen permits, but slightly lower secondary permit types, greenhouse gas and habitat permits, than the State market case. These results are in agreement with US EPA's recommendations to increase the scale and scope of WQT.

LITERATURE REVIEW

Nutrient Information

To understand the importance of water quality policies and the potential gains from nitrogen credit trading programs, it is essential to start with the origin of the problem. The lifecycle of nitrogen is generally referred to as the "nitrogen cascade" because of its effects and transformations throughout multiple environmental media. Nitrogen (N) is one of the essential elements for living organisms, and of these vital elements, it is the most abundant on Earth. However, N is biologically unavailable to most organisms when in its natural, molecular form of N_2. Therefore, the triple bond between the two molecules must be broken to form reactive N (Nr). Prior to human existence, this process occurred through lightning and biological nitrogen fixation, which was offset through denitrification and nitrogen fixation processes. Once reactive nitrogen is created, the molecules can enter the atmospheric or terrestrial environment, and eventually make their way to the aquatic environment, transforming amidst these systems throughout nitrogen's life cycle (Galloway et al., 2003). With technological advancement and the growing human population, N fixation has increased exponentially since the 1960s thus increasing its concentration throughout the environment.

The Green Revolution in the United States gave rise to increased agricultural production due to increased mechanization, new cultivation techniques, research developments in crop genetics, and enhanced inputs (Pingali, 2012). These enhanced inputs included irrigation, pesticides, and the development of chemical fertilizers through nitrogen fixation. Fritz Haber won the 1918 Nobel Prize in Chemistry for his discovery of the synthesis of ammonia, and Carl Bosch won the 1931 Nobel Prize in Chemistry when he became the pioneer of the industrial production of N fixation (Willem Erisman et al., 2008). Although the Haber-Bosch process for the synthesis of ammonia was originally developed to provide reactive nitrogen for explosives, it has also the enabled the production of chemical fertilizers.

In terms of agricultural production, Nr synthesis has increased due to the cultivation of crops which promote the creation of Nr through biological nitrogen fixation as well as the Haber-Bosch process (Galloway et al., 2003). Row-crop farmers are heavily dependent upon nitrogen fertilizer application for improved crop yield. The United Nations predicts the global population to be between 9 billion and 10 billion by 2050 (United Nations, 2019) and with a growing population comes growing demand for crop commodities. Intensification of agricultural production has enabled farmers to meet this growing demand.

Nutrient runoff from agricultural fields in the form of reactive nitrogen is a primary source of water pollution, leading to an over-concentration in neighboring water sources. The agricultural sector does not internalize the corresponding negative externalities from excessive nutrient concentration. These externalities include eutrophication, large hypoxic zones and harmful algal blooms, resulting in loss of biodiversity and habitat degradation due to oxygen deprivation (Galloway et al., 2003). NPS pollution in the form of runoff is diffused across land at heterogeneous rates depending on several factors, including soil type, weather conditions, and fertilizer application timing. The increase in agricultural intensification, along with increased variability in precipitation, exacerbates this problem. A study conducted by the World Resources Institute in 2008 found 415 areas globally that suffer from eutrophication; 169 were hypoxic (Selman et al., 2008). The World Resources Institute's most recent dataset includes 762 coastal areas affected by eutrophication, of which 479 sites have been identified as hypoxic and 228 sites suffer from other signs of eutrophication, comprising algae blooms, loss of biodiversity, and negative effects to coral reefs (World Resources Institute, 2019). The application of nitrogen fertilizer also "stimulates microbial conversion of soil nitrogen to nitrous oxide (N_2O) emissions" (Lankoski et al., 2015). Nitrous oxide is a greenhouse gas (GHG) that traps heat thus damaging the stratospheric ozone layer and advancing the progression of climate change, with an historical increase of nitrous oxide in the atmosphere that coincides with the Green Revolution (Sanders, 2012).

Along with the agricultural sector, the industrial sector is also responsible for nitrogen emissions into the atmosphere and waterways. PS polluters, such as wastewater treatment plants (WWTPs), release pollutants from a discrete source, allowing the concentration of pollutants to be more easily quantified than those contained in agricultural runoff. Because the pollution is readily available for measurement and definitively tied to a single source, the federal government regulates PS polluters under the US Clean Water Act (CWA). Historical regulation has led to the marginal costs of municipal and industrial nutrient removal technologies greatly exceeding those of the agricultural sector (Stephenson & Shabman, 2017).

Regulation and Policy Framework

The Environmental Protection Agency (EPA) expanded the Federal Water Pollution Control Act of 1948 by establishing The US CWA in 1972 as a federal framework for regulating industrial PS polluters to enforce surface water quality standards (The Federal Water Pollution Control Act, 1948). Under the CWA, the EPA developed the Total Maximum Daily

Load (TMDL) and the National Pollutant Discharge Elimination System (NPDES). The CWA calls for states to prioritize bodies of water based on the severity of pollution and risk factors such as damage to human health, aquatic life and public usage. This allows the states to establish TMDLs, or the maximum amount of pollution that allows for the maintenance of quality standards, for the most threatened water bodies. These defined standards provide the framework for forming waste load allocations for PS, load allocations for NPS, and a margin of safety to account for uncertainty, which translate into NPDES permits to ensure compliance. The state submits TMDLs to the EPA, along with supporting documentation for approval. The EPA can step in and create the TMDL themselves if they disapprove of the state developed TMDL or if the state does not take the initiative to develop a TMDL. If the TMDL is approved and NPDES permits are developed, the state is authorized by the EPA to oversee the permit system (Environmental Protection Agency, 2018c).

The management of NPS pollution originates primarily from voluntary state programs, with federal grant money offered to states to aid in the development and implementation of NPS pollution management through programs like the Clean Water State Revolving Fund. Section 319 of the 1987 amendments to the CWA established the "Nonpoint Source Management Program." Under this program, states can receive grant money by proposing watershed-based plans including NPS restrictions if it is deemed that the waterbody cannot maintain water quality standards without these restrictions (33 U.S. Code § 1329, 1987). The EPA also works with the United States Department of Agriculture (USDA) to provide funding opportunities for the agricultural sector through the Farm Bill, such as the Environmental Quality Incentives Program (EQIP) and the Conservation

Reserves Program (CRP). USDA's Natural Resource Conservation Service and the EPA are partnering with states to implement the National Water Quality Initiative in order to facilitate involvement of the agricultural sector and investment in watershed-based plans which address NPS pollution (Environmental Protection Agency, 2013). The EPA's Key Components of an Effective State Nonpoint Source Management Program suggests, "The state strengthens its working partnerships and linkages to appropriate state, interstate, tribal, regional, and local entities (including conservation districts), private sector groups, citizens groups, and federal agencies" (Environmental Protection Agency, 2012). This is an important aspect, because watersheds can be complex hydrological units that spread across state lines and require coordination for the efficient reduction of nutrient concentrations and other pollution concerns.

Policy Framework in the Chesapeake Bay

The Chesapeake Bay is a prime example of a complex hydrological unit as it is an estuary of 64,000 square miles of water systems, spanning New York, Pennsylvania, Maryland, Delaware, Virginia, and West Virginia along with the District of Columbia (*Watershed*, 2019). Although the average hypoxic zone is 1.74 cubic miles, ecologists from the University of Maryland Center for Environmental Science and the University of Michigan predict the 2019 Chesapeake Bay dead zone will be about 2.1 cubic miles, with between 0.49 and 0.63 cubic miles of water containing no oxygen. This increase in volume is due to higher levels of precipitation and streamflow, which has enabled the Susquehanna and Potomac Rivers to transport 102.6 and 47.4 million pounds of nitrogen, respectively, to the Chesapeake Bay in the spring of 2019 (University of Maryland Center for Environmental

Science, 2019). The Chesapeake Bay Program Partnership's goal is to restore the bay through the collaboration of federal, state and local government agencies, as well as nonprofit and academic organizations. The partnership comprises committees, workgroups, action groups and goal implementation teams, which enables the exchange of scientific information, evaluates and promotes abatement actions, supports the implementation of strategies to achieve the TMDL and tracks progress towards the TMDL through modeling.

The TMDL established by the EPA for the Chesapeake Bay is an aggregate of 92 smaller TMDLs, which defines "watershed limits of 185.9 million pounds of nitrogen, 12.5 million pounds of phosphorus and 6.45 billion pounds of sediment per year. This equates to a 25 percent reduction in nitrogen, 24 percent reduction in phosphorus and 20 percent reduction in sediment" (Environmental Protection Agency, 2018b). To date, there has been little success in achieving these goals, so the jurisdictions have extended agreements to establish deadlines and accountability. Specifically, the Chesapeake Watershed Agreement is an extension of the Chesapeake Bay Program partnership, which defines Watershed Implementation Plans (WIPs) at the state level (Environmental Protection Agency, 2014). In 2018, Phase III of the WIPs for the seven jurisdictions was approved and outlined nitrogen and phosphorous reduction goals, as well as the corresponding action plans from 2019-2025. These plans include Pennsylvania's "revision of state trading regulations and NPDES permits to address trading program deficiencies and facilitate MS4 [Municipal Separate Storm Sewer System] and interstate trading in order to allow permittees to manage their compliance obligations cost effectively and leverage nitrogen and phosphorus reductions" (Environmental Protection Agency, 2018a).

Policy Instruments for Water Quality Management

PS pollution reductions have thus far been pursued through inefficient, technology-based effluent limits, which are specified within the NPDES permit program, and must be reported to the EPA (Stephenson & Shabman, 2017). This is an example of a command and control (CAC) policy, or a direct regulatory instrument, which does not meet the conditions for cost-effectiveness in general. These are generally the most politically feasible and easiest to implement but fail to induce innovation as they lack flexibility in how to make pollution reductions.

Alternatively, incentive-based instruments promote innovation and implementation of least-cost abatement (LCA) mechanisms throughout the production process. For example, input-based policies, in the form of taxes and subsidies, are feasible NPS abatement polices since these can be directly observed and measured, unlike emissions. Input taxes, such as taxes on fertilizer, represent a second-best policy since the tax is applied to only one input as opposed to all the inputs which contribute to the emissions output (Shortle et al., 1998). Uniform taxes are easier to enforce, but with heterogeneity in resources and producers as well as endogenous prices, they may not be the most equitable. Non-uniform taxes can address equity concerns but still can result in pecuniary externalities, such as a decrease in the price of fertilizer or an increase in the price of outputs. Pecuniary externalities are conveyed through markets and manifest as price changes, which in turn "impact environmental externalities in other sub-regions by altering social pollution control costs and hence the level of pollution control in these other areas" (Claasen & Horan, 2001). An example of a pecuniary externality would be a decline in demanded fertilizer in one region could allow the price to decrease, thus allowing an increase in fertilizer usage in another region. These indirect effects in other regions, or spillover effects, need to be accounted for along with substitution externalities, wherein extensification of agricultural production supplants intensification. Policies addressing NPS pollution reduction need to account for potential unintentional externalities, which can be difficult for smaller, voluntary programs to address.

Green payments are common incentivizes for voluntary participation through programs which pay for activities that reduce environmental impacts. Examples include the aforementioned CRP and EQIP. Within the framework of welfare economics, transfer payments have distortionary effects, so policymakers and regulatory bodies need to account for efficiency and equity objectives. Because NPS emissions cannot be precisely quantified, green payments are based upon observable actions of the production process and can be implemented as input subsidies or as contracts (Horan et al., 1999). These payments offset the producer's costs of implementing the emission-reducing activity or reducing the use of a polluting input while also providing social benefits by reducing environmental damages. When abatement actions and land management are correlated with emission levels and easily observed, they are used as environmental proxies and rewarded through green payments. By using BMPs as environmental proxies, water quality models can provide estimations of their effects.

Ambient policies involve quantifying aggregate emissions at the watershed outlet as opposed to at the firm or farm level. An ambient subsidy/tax scheme would pay firms when ambient pollution falls below a certain level and tax when the concentration exceeds the target. However, firms are not rewarded nor penalized for their individual performance, and the firm's response will depend on their expectations of the impacts of their choices and the choices of others, as well as stochastic, surrounding conditions (Shortle et al., 1998).

Cap and trade, or cap and tax, policies are the foundation for market-based environmental permit trading programs such as water quality trading programs (WQTPs). WQTPs are market-based programs that allow producers with high pollution abatement costs to purchase permits, or abatement credits, from sources that have relatively lower abatement costs. To participate in WQTPs, agricultural producers must implement BMPs and produce nitrogen abatement beyond the assigned baseline requirement. The EPA defines a baseline participation requirement as the pollutant control requirement, or minimum abatement threshold, which apply to a seller in the absence of trading (Environmental Protection Agency, 2007). Abatement that goes beyond this baseline can generate credits, and in turn, these permits are eligible for market sale. BMPs can be edge-of-field practices, which require installation or annually implemented integrated field practices. Examples of BMPs include nutrient management programs, cover crops, riparian buffers, planting of varieties with improved N use efficiency, biofilters, no-till methods, irrigation management, land retirement, and wetland restoration or construction (Environmental Protection Agency, 2010a).

Many BMPs aimed at nitrogen abatement efforts also mitigate phosphorus loading and soil erosion, aid in carbon sequestration, and potentially provide wildlife habitat. These ancillary benefits, often referred to as co-benefits, can be the result of one abatement strategy or a combination of actions. For example, wetland construction at the edge of agricultural land, with drainage leading to the site, can not only facilitate nitrogen mitigation but also act as a carbon sink, provide natural habitat for animal and plant species, and offer flood control (Zedler & Kercher, 2005). Because wetland restoration and construction involve removing agricultural land from production, the Natural Resource Conservation Service's Wetlands Reserve Program (WRP) provides payments to landowners to offset some of the incurred costs. Landowners could also participate in environmental credit markets due to the potential to generate carbon, nitrogen, and wildlife credits. In part because of the success of carbon markets, there is increasing interest in market-based policies for water quality goals, especially WQTPs, but there has been limited trading to date; of the 11 states that have trading programs, only Pennsylvania, Virginia, and Connecticut have seen significant trading (Davies, 2019). The US EPA's Assistant Administrator for Water, David Ross, recently encouraged states to establish WQT markets to address pollution issues in a memo to EPA regional administrators. The Chesapeake Bay's WIPs intend to aid the trading process and facilitate trade of permits produced and purchased within the same watershed sub-basin, even if the sub-basin crosses state boundaries.

For a well-functioning market to form, there needs to be clear definitions of tradable commodities, rules of exchange, and binding caps (Horan & Shortle, 2011). Theoretically, WQTP allow for cost-effectiveness through comparative advantage without the regulatory body needing to have firm-specific abatement cost information. WQTP have the potential to be second-best policies for attaining pollution reductions goals, as they incentivize low-cost agricultural abators to produce permits and sell to regulated industrial polluters with higher marginal abatement costs (Shortle & Horan, 2013). Market-based environmental permit trading has been largely successful for air quality (Kaupa, 2019) and is taking off for water quantity (Velloso Breviglieri et al., 2018); however, when it comes to water quality, there are deviations from the standard, theoretical markets. In particular, "fundamental features of textbook markets are that emissions (1) can be accurately metered for each regulated emitter and (2) are substantially under control of the emitter, and that (3) the spatial location of emissions is not relevant to the attainment of the environmental target" (Horan & Shortle, 2011). These conditions do not hold in the context of water quality, especially with the inclusion of NPS emitters.

Additional issues with environmental markets are setting a stringent cap and specifying trade ratios, which safeguard the environmental goals by limiting the aggregate supply of permits. To do so, the consideration of permit specifications, PS allowances, and the NPS trade ratios is important in relation to the abatement objective. Trade ratios ensure that the water quality goals are attained by considering the stochastic nature of runoff and transport attenuation rates (Shortle & Horan, 2013). The definition of water quality goals as the TMDLs for the watershed and the NPDES permit system set the pollution cap and framework for trading by enforcing the allocation of emissions.

Credit Stacking and Additionality

As mentioned previously, some water quality BMPs can create multiple environmental benefits to water, soil, air, and habitat conditions. With ancillary benefits of BMPs comes the question of whether a producer can generate different types of environmental permits, thus participating in multiple markets, from a spatially overlapping area (Fox et al., 2011). This is known as credit-stacking, and although additional revenue could promote greater abatement participation from agricultural producers or a more cost-effective mixture of practices, there exist concerns regarding additionality provisions. Additionality is the principle that permit sellers should not receive payments for a benefit that would have occurred without the additional payment. Greenhouse Gas Management Institute defines an activity as additional "if the recognized 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" (Gillenwater, 2012). For example, a riparian buffer installed on the edge of a field for water quality benefits could have occurred without the additional incentive of a wildlife habitat payment. The landowner is incentivized by water quality payments to install the riparian buffer, and the additional habitat payment could incentivize the installation of greater buffer area, or it may result in the same amount of buffer area at a larger cost.

Participating in distinct markets engenders the possibility of having to interact with different regulatory agencies and require distinct verification processes to ensure the abatement action is above baseline requirements. Participating in multiple, separate environmental markets also induces the potential of higher transaction costs that can take the form of opportunity costs, search costs, verification costs, and legal fees (Woodward & Kaiser, 2002). The exchange framework of environmental markets usually involves bilateral negotiations between buyers and sellers or on intermediaries, such as clearinghouses. Some nonprofit organizations have filled the gaps of regulatory bodies in providing guidelines for permit generation, as well as verification services to ensure transferability and enforceability. The Electric and Power Research Institute (EPRI) led the founding of the Ohio River Basin Trading Project, which is an interstate WQT framework with permits recorded online through the IHS Markit exchange (EPRI, 2019). Willamette Partnership, named after the tributary in Oregon, developed the Ecosystem Credit Accounting System to guide the provision of credits for water quality and habitat benefits (Willamette Partnership, 2019).

The Willamette Partnership's Ecosystem Credit Accounting System allows for the verification and certification of water quality, aquatic habitat, and terrestrial habitat. A site visit for a project that, for example, generates salmonid, thermal and wetland credits would allow a verifier to inspect the property and certify multiple credits simultaneously. If multiple, distinct credits are generated from a spatially overlapping project site, this would be considered credit stacking. Credit stacking would lower the net cost to the landowner, as multiple revenue streams could be vetted at once by an accredited verifier of an organization such as Willamette Partnership (K. Teige Witherill & C. Sanneman, personal communication, November 8, 2018). This allows the credit producer to diversify their "portfolio" of credit types and ideally sell the most profitable type or potentially sell multiple credits as a bundle to one buyer. In a 2015 news release, EPRI announced their new project commitment to investigating credit stacking of greenhouse gas and nutrient credits:

> "The U.S. Endowment for Forestry and Communities (Endowment) committed \$1.5 million to integrate forestry projects as a best management practice on farmland for reducing nutrient (nitrogen and phosphorous) runoff. The U.S. Department of Agriculture (USDA) awarded a \$300,000 Conservation Innovation Grant to develop "credit stacking" of nutrient and greenhouse gas emission reductions" (Perry & Mahoney, 2015).

Maryland's WQTP has begun setting guidelines for credit stacking of water quality and GHG permits (Gasper et al., 2012).

Research on credit stacking typically focuses on additionality concerns, specifically, this literature explores situations in which a net gain in abatement is possible and when there is potential for net losses. Lentz et al. (2014) examine wetland construction as an abatement option for water quality. Wetland investments are categorized as "lumpy" because of large, upfront fixed costs that result in discrete investment decisions. The study models WWTP demand for nitrogen permits and the supply of permits from wetland construction. The authors find that additionality may not hold when they allow for stacking in the market, because of the lumpy nature of the investment, and market outcomes depend on the demand for the primary credit.

A study concerning climate change and eutrophication mitigation in the Baltic Sea examined the changes in costs associated with abatement and the overall abatement level under different theoretical policy regimes (Gren & Ang, 2019). The three policy regimes studied here include trading systems, pollutant tax systems, and CAC. One of their main conclusions is that stacking can reduce total abatement cost under each policy regime. By allowing for stacking under CAC, cobenefits from an action targeted at a reduction in one nutrient were counted, if the action resulted in the reduction of an additional nutrient. The overall reduction in cost depends upon the relative magnitude of the cobenefits generated from the abatement measures and the stringency of the caps. Furthermore, their analysis demonstrates that without stacking, given that the abatement targets are sufficiently stringent, abatement surpassing the targets will occur. Although this may sound beneficial at first blush, "if the targets are optimally determined by considerations of values and costs of abatement measures, excess abatement would imply a net cost since the marginal abatement cost exceeds the marginal benefits for each pollutant" (Gren & Ang, 2019).

Prior work often assumes that pollution caps are set optimally, in which case participation in multiple markets could still result in a social optimum. However, the policymaker would need to have perfect information of the costs and benefits and implementation of all markets for all pollutants would need to be coordinated to attain this optimum (Woodward, 2011). In reality, prior regulations often define pollution regulations, making efficient caps and policy coordination potentially infeasible. Given this constraint, efficiency gains can be made by defining inter-pollutant trade ratios within an integrated market (Reeling et al., 2018a). An integrated market would allow for both intra-pollutant trading—trading of "like" permits—and inter-pollutant trading. With inter-pollutant trading—the trading of different pollutant types—there are also inter-pollutant trade ratios, which equate pollutant types (Reeling et al., 2018a). For example, N_2O emissions can be converted to CO_2 equivalents and traded in GHG markets. By combining multiple types of pollution permits into a single commodity, an integrated market could alleviate some of the transaction costs and uncertainty incurred by the producer (Reeling et al., 2018b).

There are previous studies of climate cobenefits within the framework of environmental markets within the Chesapeake Bay due to Maryland's promotion of credit stacking of GHG and water quality credits. Gasper et al. (2012) consider how participating in multiple markets could provide further incentives to adopt abatement practices, therefore expanding NPS market participation. They explore the differences between regulatory and financial additionality principles regarding credit stacking. An interesting BMP that has produced multiple cobenefits is the restoration or construction of wetlands, as they can provide abatement for water pollution and air pollution as well as habitat. Because the use of wetlands for nutrient abatement has ancillary benefits, there is justification to support the activity by supplementary incentives, such as subsidies. Herberling et al. (2010) explore how the additional incentive of a subsidy might affect the farmer's production decisions and the potential for unintended consequences. Such potential issues of increased fertilizer use and the expansion of cropland depend upon whether land is constrained. In the case of unconstrained land, they find that a marginal increase in the subsidy increases wetland area, fertilizer application and cropland area. In short, the wetland subsidy acts as a fertilizer subsidy and can encourage cropland expansion. The policy implications suggest a wetland subsidy would be effective in the constrained land case or when coupled with restrictions of cropland expansion.

While this literature has sought to answer under what circumstances additionality holds in theory as well as in specific applications to certain regions, the studies have not addressed additionality in conjunction with pecuniary externalities outside the region of study. By implementing a partial equilibrium that spans the continental U.S., I account for additionality with the watershed region, while also observing spillover effects. This approach allows me to account for any "exporting of pollution" to other regions due to policies within the Chesapeake Bay. By doing so, social welfare changes are accounted for on a larger scale.

Regional, Hydrological Analyses of WQTP

In watersheds with a large proportion of NPS polluters, the TMDLs could require NPS participation to meet the abatement goal. Because NPS participation is usually voluntary, incentive-based policies can induce abatement actions. Targeted green payments have had limited success, and states have turned to WQTPs to facilitate the necessary reductions to meet the TMDL (Shortle et al., 2012). In practice, WQTP have been characterized by limited trading to date due to "uncertainty over the number of discharge allowances for different management practices, difficulties in predicting pollutant reduction at the point in the watershed where the purchasing point source discharges, the reluctance of point sources to trade with unfamiliar agents, and the perception of some farmers that entering contracts with regulated point sources leads to greater scrutiny and potential future regulation"

(Ribuado & Nickerson, 2009). Ribaudo and Nickerson analyze 710 impaired, eight-digit HUCs and find 20% and 32% were best suited for nitrogen and phosphorus permits, respectively. This lack of suitability is attributed to low demand for permits relative to the high potential supply, given the agricultural sector dominates production in most watersheds. However, targeted watershed and/or sub-basin analyses can help evaluate the suitability of WQTPs in a specific region by evaluating the biophysical and economic conditions that are unique to the region.

Corrales et al. (2014) utilize water quality and economic modeling to assess the economic and environmental benefits of implementing a phosphorus credit-trading program in a sub-basin of Lake Okeechobee in Florida. They compare the LCA approach, in the form of WQTPs, to the CAC approach. The use of the Watershed Assessment Model (WAM) allowed for a water quality analysis and the estimation of attenuation rates of phosphorus for trade ratio calculation. The LCA scenario is a cost minimization of available abatement technology for PSs and NPSs throughout the basin, and the WQTP induces comparative advantage. In this scenario, there was selection of BMPs to meet the individual total phosphorus load allocations, as opposed to optimizing across the entire sub-basin. Overall, they find a cost-savings of 27% ($1.3 million per year) from implementing the LCA over the CAC scenario. An additional study by Corrales, et al. (2017) extends the previous study by including PSs in the form of WWTPs and concentrated animal feed operations for a more diverse trading pool and included two different, more hydrologically complex, sub-basins. The use of an optimization problem defined the optimal cap per sub-basin, as opposed to basing it on the existing TMDL. They find the optimal targets to be 46% and 32% reductions with an estimated net cost-savings of 76% and 45%, respectively.

While least-cost models optimize abatement measures within the study region, they often fail to account for changes that occur outside of the region. As mentioned in the previous study, the partial equilibrium modeling framework allows accounting of pecuniary externalities. Because SIMPLE-G is an economic framework across the U.S., market changes will be accounted for within and outside of the Chesapeake Bay region. Also, due to the grid-level biophysical information, pollution changes throughout the country will also be tracked. While LCA models have been applied to the CBW before, to my knowledge, this is the first application of a partial equilibrium model to this region.

General and Partial Equilibrium Models

Policy options and market schemes must consider both potential costs and benefits in terms of environmental and agricultural effects as well as account for unintended consequences. Market-based policies are a cost-effective option but can still create pecuniary externalities when addressing negative environmental externalities (Marshall et al., 2018). A benefit of both general and partial equilibrium models is the ability to capture behavior and interactions within the market, thus capturing potential feedbacks and pecuniary externalities that otherwise could be overlooked. While partial equilibrium models focus on one sector within the market, general equilibrium models model the whole economy. General equilibrium models were originally developed to answer global trade questions, while the partial equilibrium models described below focus on the agricultural sector's influence on nutrient emissions. Partial equilibrium models allow for more complexity within the represented sector as well as finer resolution for spatial data and effects, as its narrower scope avoids the need for data aggregation. A benefit of this grid-level data representation for environmental and natural resource economic modeling is that the resolution remains at the same level as physical models (e.g. nutrient loss models), thus allowing the combination of environmental models with economic models.

The application of general equilibrium models allows the examination of economy-wide effects of policy regulations relating to environmental quality concerns. Carbone and Smith examine how reductions in sulfur and nitrogen oxides contribute to environmental and health cobenefits, as well as use-based and non-use based environmental activities. They argue that by not including all facets of the value of ecosystem services, previous studies have not acknowledged "the idea that demand for environmental quality responds to relative price changes (and changes in other dimensions of the non-market services, environmental as well as other public goods, that are available outside markets)" (Carbone & Smith, 2016). By constructing a utility function that incorporates non-market services, the authors link emissions levels to non-market ecosystem services to create feedbacks that influence net benefits. Other studies apply the general equilibrium approach to model water quality improvements. Dellink et al. (2011) investigate water pollution reduction policies in the context of the Dutch economy using a dynamic applied general equilibrium model, which is linked to a water quality model. Within this analytical framework, the authors inspect the effects of different policy scenarios relating to the European Water Framework Directive.

It is common to use partial equilibrium models in the context of climate change policy and land use change to assess the environmental and economic impacts. Kesicki (2013) studies carbon emissions reductions in the United Kingdom utilizes an energy sector model, UK MARKEL, to estimate marginal abatement cost (MAC) curves for carbon abatement. Kesicki considers the uncertainty of various MAC curve parameters by performing sensitivity analysis to determine the most influential drivers. The model is run with the calibrated parameters and varying levels of a carbon tax to generate the MAC curve. Other studies focus on land use change and incorporate uncertainty by including individual decision-making. Morgan and Daigneault (2015) developed the Agent-based Rural Land Use New Zealand model, which combines partial equilibrium modelling and agent-based decision making to estimate responses to GHG price policies. Another study in New Zealand examines both climate and water policies concerning their impacts on farm income, land use and the environment (Daigneault et al., 2018). The authors employ a partial equilibrium economic model that contains spatial data of New Zealand land use that "tracks changes in land cover, enterprise distribution, land management, GHG emissions, N leaching, soil loss, water yield and P loss resulting from a variety of policy options." This enables the analysis of standalone climate change and water policy as well as a simultaneous climate change and water quality policy.

For water quality specifically, Schou et al. (2000) develop a partial equilibrium model for the Danish agricultural sector, along with geospatial information systems procedures and a nitrate-loading model that allows for policy analysis pertaining to nitrogen tax instruments in Denmark. Their results demonstrate variation in nitrate leaching with respect to the source, the spatial pattern of leaching vs loading and costs to different farm types. Savard (2000) develops an international model of the hog industry to examine the environmental implications of trade policies between the U.S. and Quebec due to the use of manure as an agricultural nutrient input. Land use and environmental impacts have also been studied in the context of water quantity policies (Daigneault et al., 2011).

Recent studies use partial equilibrium models to evaluate policies aimed at reducing the Gulf of Mexico hypoxic zone. Agricultural production along the Mississippi and Atchafalaya River Basins transports excess nutrients into the water system and is the main cause of eutrophication in the Gulf of Mexico. The Corn Belt region of the US lies within the Mississippi River Basin, and the agricultural sector is responsible for around 60% of the nitrogen load (Robertson and Saad, 2013). The EPA founded the Mississippi River Gulf of Mexico Watershed Nutrient Task Force to mitigate over-concentration of nutrients within the Gulf of Mexico and reduce the summer hypoxic zone from 5,236 square miles to 1,931, which translates to a 45% reduction in cropland nutrient loads (Marshall et al., 2018). Marshall et al. (2018) recently performed a study on behalf of the USDA's Economic Research Service to analyze how this hypoxic zone reduction would translate to economic impacts by utilizing the Regional Environmental and Agriculture Programming (REAP) model and Conservation Effects Assessment Project data. They find that employing the necessary abatement methods to reach the 45% nitrogen and phosphorus load reduction goal would decrease crop commodity production and increase the price of crop commodities. These initial findings also result in spillover effects, where intensification occurs in outside regions to offset the reduction in production and lessen the increase in crop price levels, therefore shifting water quality issues to other watersheds. In the case of conservation practices, such as land retirement or wetland conversion, extensification could also occur where marginal lands are available for cropland conversion. If regional land rents are affected, they can, in turn, affect the crop commodity prices depending upon the marginal land regional elasticities.

A similar study by Liu et al. (2018) looks specifically at the impacts of reductions in nitrogen losses from corn production. The authors evaluate the abatement methods that would be necessary to meet the 45% reduction in nitrogen leaching by layering the BMPs one at a time. SIMPLE-G, a partial equilibrium model, captures market effects, while Agro-IBIS models the changes in nitrogen leaching resulting from the layers of BMPs. The authors find the required leaching charge to incentivize the 45% reduction to be $0.75/lb., equivalent to a 130% ad valorem tax, if a reduction in the application of N fertilizer is the only BMP utilized. This results in a 17% reduction in corn production and thus a rise in prices, which the inclusion of split nitrogen application as an additional available practice lessens. In subsequent steps, controlled drainage and wetland conversion are also simulated and the combination of these practices can achieve the targeted reduction with about a 1.5% reduction in corn production and a $0.12/lb. leaching charge (Liu et al., 2018). This policy brief also looks at the spatial consequences of each policy to show the regional effectiveness of each BMP.

Partial equilibrium models provide a more holistic understanding of policy effects by accounting for pecuniary externalities and unintended consequences. In this way, they are "what-if" analysis tools, in that they demonstrate the outcomes of certain market specifications. However, for my analysis of water quality markets, my goal is to provide a behavioral modeling tool to examine how the agricultural sector would respond to production decisions for permit generation.

METHODOLOGY

I utilize the Simplified International Model of agricultural Prices, Land use and the Environment, on a Grid model (SIMPLE-G) to simulate point-nonpoint WQTPs within the CBW. The SIMPLE-G model is a gridded partial equilibrium model of cropland use, crop production, crop commodity consumption, and trade (Baldos et. al, 2020; Baldos & Hertel, 2012). SIMPLE-G models crop production over each grid cell covering the Continental U.S., with each 5-arcmin × 5-arcmin grid cell representing a single, aggregate agricultural producer/landowner. The area represented by each 5-arcmin × 5-arcmin grid cell changes depending upon the latitude at which the grid is overlaid, but they are often referred to as 10 km grids. Figure 1 shows the grid set layer over the CBW region.

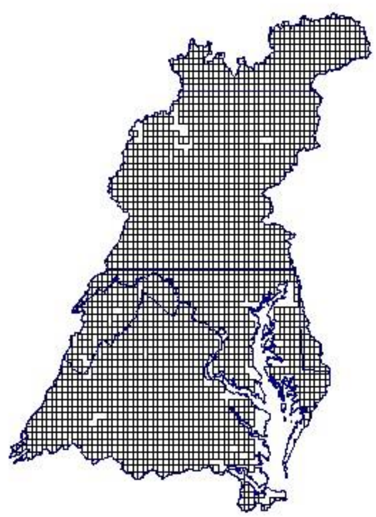

Figure 1: SIMPLE Grid Represented Over the Chesapeake Bay Watershed

Cropland within the model takes on two land types, irrigated and rainfed. If the quantity of a land type within the model changes—either through expansion or contraction—the land must shift to or from the other land type. Relative rental rates—representing opportunity costs—guide cropland providers' decisions in the allocation of land.

Modeling WQTPs requires modeling permit supply and demand functions for polluters within the CBW at the grid cell- and land type-level. The permits generated by each grid cell and land type are then aggregated to different spatial scales depending on our market-size scenario, which allows for the simulation of WQTPs between pollution sources at various market sizes. NPS are represented by grid cell aggregate profit-maximizing crop producers in the agricultural sector that emit nitrogen pollution due to fertilizer use, while the PS are represented by industries that emit nitrogen pollution.

SIMPLE-G Model Framework and Structure

The formal structure of SIMPLE-G comprises grid cell- and land type-level constant elasticity of substitution (CES) demand and supply nests (Baldos & Hertel, 2012). The model considers an aggregate crop commodity supply and demand at the regional level, where markets are assumed to clear. Zero-profit equations and economic responses ensure long-run market equilibrium for all the sectors. Crop output depends on input supply elasticities, technological efficiencies, and relative prices. Derived demand for production inputs at each grid cell depends on substitution elasticities, input efficiencies, and relative prices. Derived demand for crop commodity outputs at regional level depends on income, population, substitution elasticities, and relative prices. The baseline data is a global database for 2010 constructed from the World Development Indicators (World Bank, 2003; 2011), the World Population Prospects (United Nations, Department of Economic and Social Affairs, Population Division, 2011), the GTAP V.6 database (Dimaranan, 2006), FAOSTAT (2011), and Angel et al. (2010).

Figure 2 shows the demand structure of SIMPLE-G, represented in the CES nested framework, for crop commodities. The quantities of regional consumption ($Q_r^{cons,i}$) depend on the population and per capita consumption within the region. These quantities of regional consumption are broken down into the categories of processed foods, crops, and livestock feed. Aggregate demand for crop commodities in region r ($Q_r^{D,crop}$) comprises the derived demand for crops as a final good from individual households ($Q_r^{crop,direct}$), as production inputs by the processed food sector ($Q_r^{crop,proc}$), and as feedstock for the livestock sector ($Q_r^{crop,lvstck}$). The derived demands depend on the elasticity of substitution between local and global markets as well as endogenous market prices. The processed food and livestock sectors can also substitute between crop and non-crop inputs (indexed by the superscript *ncrop*), as shown by the elasticity of substitution terms, σ, in Figure 2.

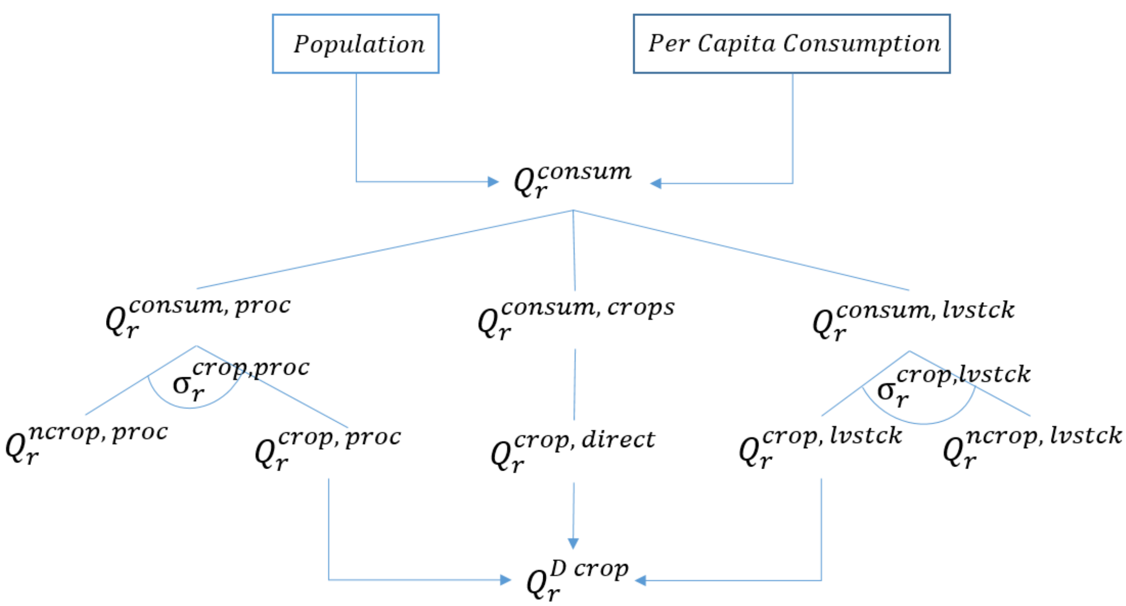

Figure 2: Constant Elasticity of Substitution Demand Nest for Crop Commodities

The demand for crops from the biofuel sector ($Q_r^{crop,bio}$) is an additional demand but is not included in Figure 2 as it is set exogenously. The market-clearing condition,

$$(1) \qquad \sum_r Q_r^{D,crop} \equiv \sum_r \left(Q_r^{crop,lvstck} + Q_r^{crop,proc} + Q_r^{crop,direct} + Q_r^{crop,bio}\right) = \sum_r Q_r^{S,crop}$$

ensures that the aggregate supply across all regions (the final right hand-side term in (1)) equals the aggregate demand across all regions from direct consumption and production inputs within region *r*, $Q_r^{D,crop}$, (the left hand-side term in (1)).

Within each region, grid cell- and land type-level crop production decisions depend on aggregate demand for commodities and the relative prices of inputs. I denote the quantity (in metric tons, MT) of crop output produced in grid cell g with land type l (discussed later) as $Q_{g,l}^{crop}$, with $Q_r^{S,crop} = \sum_{l \times g \in r} Q_{g,l}^{crop}$. I denote by $P_{g,l}^{crop}$ the corresponding grid cell price of crop output (in USD/MT).

Figure 3 shows the supply nest structure of SIMPLE-G. Production inputs include nitrogen fertilizer (in MT), $Q_{g,l}^{nitro}$, cropland (in hectares), $Q_{g,l}^{cropland}$, and non-land inputs. Non-land and cropland inputs used in grid cell g on land type l are nested together as an "augmented land" input, $Q_{g,l}^{aug}$. At the first level of the production nest shown in Figure 3, the inputs of $Q_{g,l}^{nitro}$ and $Q_{g,l}^{aug}$ are chosen optimally to produce $Q_{g,l}^{crop}$. The subscript l denotes land type, which takes the form of *irrig* for irrigated land and *rain* for rainfed land. The non-land inputs include groundwater for irrigated land and a non-land aggregate (i.e., labor costs and capital investment). Grid cell- and land type-level output depends on total factor productivity, $\theta_{g,l}^{crop}$, the input quantities' corresponding input efficiencies ($\theta_{g,l}^{aug}$ and $\theta_{g,l}^{nitro}$), and the substitution parameter $\rho = \left(\sigma_{g,l}^{crop} - 1\right)/\sigma_{g,l}^{crop}$, where $\sigma_{g,l}^{crop}$ represents the elasticity of substitution between augmented land and nitrogen as inputs in crop production. In addition, the ν_r parameters under the input quantities represent the supply elasticities by region and the τ_r parameters represent the transformation elasticities.

Crop production follows a CES specification:

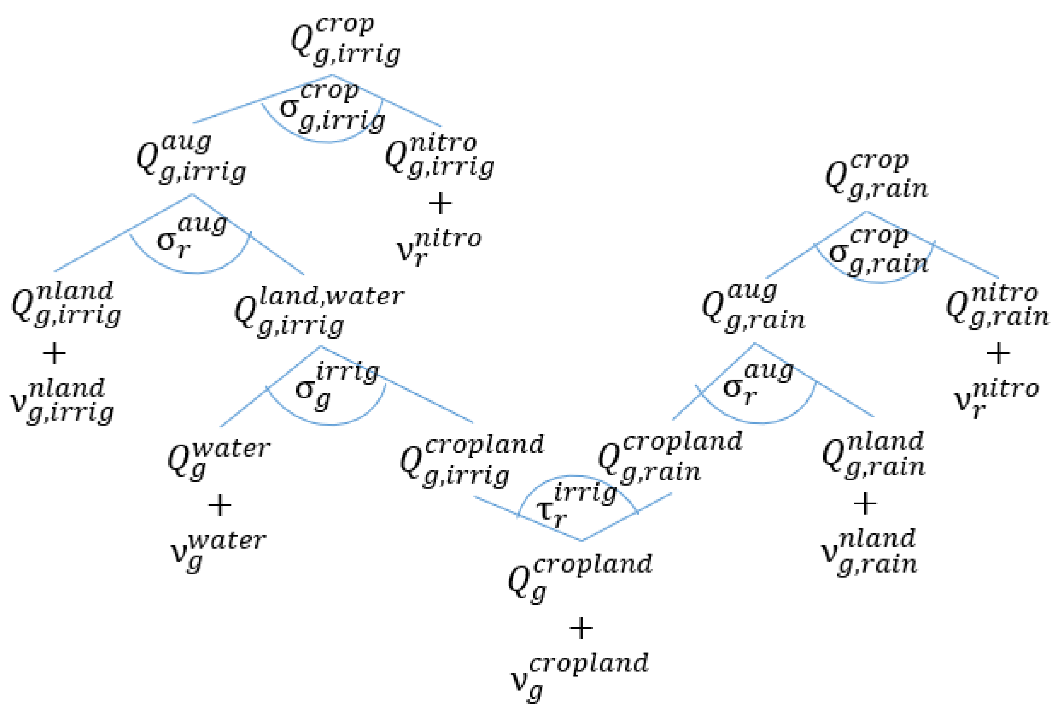

Figure 3: Constant Elasticity of Substitution Supply Nest for Crop Commodities

(2) $$Q_{g,l}^{crop} = \theta_{g,l}^{crop}\left[\left(\theta_{g,l}^{aug} Q_{g,l}^{aug}\right)^{\rho} + \left(\theta_{g,l}^{nitro} Q_{g,l}^{nitro}\right)^{\rho}\right]^{\frac{1}{\rho}}.$$

I derive input demands by solving a cost-minimization problem following Yang (2019). Formally, the input demands solve

$$(3)\qquad \min_{Q_{g,l}^{aug},Q_{g,l}^{nitro}} \; Q_{g,l}^{nitro}P_{g,l}^{nitro} + \; Q_{g,l}^{aug}P_{g,l}^{aug} \quad \text{subject to (2),}$$

where $P_{g,l}^{aug}$ is the price (in USD/ha) of the augmented land bundle and $P_{g,l}^{nitro}$ is the price (in USD/MT) of nitrogen fertilizer used in grid cell g on land type l. Baseline prices from market data for both inputs are updated within the model as demand and supply adjust given shocks to the system.

Long-run equilibrium in the crop production sector requires producers at the grid cell- and land type-level earn zero profits, denoted $\pi_{g,l}^{NPS}$. The zero-profit condition satisfies

$$(4)\qquad \pi_{g,l}^{NPS} \equiv Q_{g,l}^{crop}P_{g,l}^{crop} - \left[Q_{g,l}^{aug}P_{g,l}^{aug} + \; Q_{g,l}^{nitro}P_{g,l}^{nitro}\right] = 0.$$

I use the first-order conditions (FOCs) from (3) and (4) to solve for equilibrium crop price, $P_{g,l}^{crop}$, and the input demand functions

$$(5)\qquad Q_{g,l}^{aug} = \left(\theta_{g,l}^{crop}\right)^{\sigma_{g,l}^{crop}-1}\left(\frac{P_{g,l}^{crop}}{P_{g,l}^{aug}}\right)^{\sigma_{g,l}^{crop}}\left(\theta_{g,l}^{aug}\right)^{\sigma_{g,l}^{crop}-1}Q_{g,l}^{crop}$$

$$(6)\qquad Q_{g,l}^{nitro} = \left(\theta_{g,l}^{crop}\right)^{\sigma_{g,l}^{crop}-1}\left(\frac{P_{g,l}^{crop}}{P_{g,l}^{nitro}}\right)^{\sigma_{g,l}^{crop}}\left(\theta_{g,l}^{nitro}\right)^{\sigma_{g,l}^{crop}-1}Q_{g,l}^{crop}\,.$$

SIMPLE-G simulates nitrogen leaching from nitrogen fertilizer application at the grid cell- and land type-level. The quadratic leaching function estimates the rate at which each grid cell emits nitrogen runoff (in kg/ha), given the land type and other physical characteristics.

Extending SIMPLE-G to Incorporate Environmental Permit Trading

Nonpoint Source Permit Supply

For my study, I utilize a regional model of SIMPLE-G comprising the 2,300 All Crops database grid cells contained within the Chesapeake Bay Watershed. This model deviates from the framework explained above in that it assumes the price of crop outputs, nonland inputs, and nitrogen fertilizer are exogenous. However, the price of land and water inputs are endogenous, consistent with the notion that these markets are local in scope. The demand for nitrogen fertilizer depends on the opportunity costs of using nitrogen, which will vary under the WQTPs I describe in the next section. Therefore, grid cells do not communicate via the crop market, but instead through permit markets; the changes in crop production in one grid cell can change the corresponding leaching amount, and thus the permit prices, which feeds back to crop production decisions in other grid cells.

I simulate environmental permit trading in SIMPLE-G by modifying the supply nest in Figure 3 to include permits as additional commodities. Because some of the water quality BMPs I model in SIMPLE-G generate ancillary environmental benefits (e.g., GHG abatement and wildlife habitat provision), I also consider various trading scenarios in which conservationist producers can receive revenues from selling carbon and wildlife habitat permits. This is relevant as the United States has several regional carbon markets (e.g., the

Regional Greenhouse Gas Initiative and California's Cap-and-Trade market established under AB 32) and conservation incentive payments (e.g., the Conservation Reserve Enhancement Program).

I use a linear approximation of the original quadratic leaching function within SIMPLE to enable the derivation of the new nitrogen input demand function under permit trading, described below. Under this linear specification, the leaching rate is a proportion $\delta_{g,l,b}^{nitro}$ of $Q_{g,l}^{nitro}$, where b denotes the BMP that is applied (described later). Formally, leaching is

$$Q_{g,b}^{emissions,N} = \sum_l \delta_{g,l,b}^{nitro} Q_{g,l}^{nitro}. \tag{7}$$

I simulate water quality permit trading using SIMPLE-G (described later), where crop producers generate water quality permits by adopting BMPs that reduce nitrogen leaching (7). These BMPs include nutrient management plans and wetland construction. In this study, the nutrient management plan involves decreasing the amount of nitrogen fertilizer applied. Decreasing the amount of nitrogen applied, $Q_{g,l}^{nitro}$, decreases leaching via (7) and can lead a profit-maximizing producer to substitute for more $Q_{g,l}^{aug}$. With a reduction in the application of nitrogen fertilizer, there could be an increase in $Q_{g,l}^{aug}$ because of the substitution effect on inputs required to hold output constant. Therefore, the opportunity cost of a reduction in applied nitrogen fertilizer is the foregone profit from increasing input costs.

In contrast, wetland installation reduces the grid cell-level leaching rate to $\delta_{g,l,wetland}^{nitro} Q_{g,l}^{nitro} < \delta_{g,l,0}^{nitro} Q_{g,l}^{nitro}$. The opportunity cost of wetland construction equals the reduction in profit due to a loss in crop output plus the cost of installation. Crop output could decline because of the reduction in $Q_{g,l}^{aug}$ due to less available land for crop production, unless the producer substitutes additional non-land inputs into crop production.

I assume producers in each grid cell influence GHG emissions via nutrient management. Emissions increase with nitrogen fertilizer application since nitrogen fertilizer can volatilize into nitrous oxide (N_2O), a potent greenhouse gas with 298 times the global warming potential of CO_2 (Environmental Protection Agency, 2017). Hence, nutrient management BMPs that reduce fertilizer application will also reduce GHG emissions. The 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories estimates different emissions factors for N_2O emissions from nitrogen inputs depending on cropping activities, nutrient management plans, soil type, and climate factors (Task Force on National Greenhouse Gas Inventories, 2019). CO_2 equivalent N_2O emissions are

$$Q_g^{emissions,G} = \sum_l \left(\delta^{GHG} Q_{g,l}^{nitro}\right) \times 298. \tag{8}$$

The term in parentheses equals total N_2O emissions, where δ^{GHG} is the emissions factor from nitrogen fertilizer. Scaling these emissions by 298 accounts for the difference in global warming potential between N_2O and CO_2. Converting croplands to wetlands can also provide habitat for imperiled species (e.g., migratory birds (De Steven & Lowrance, 2011)). Hence, producers in each grid cell generate wildlife habitat permits by allocating land to wetlands.

Now consider the producer's optimization problem. In the absence of trading, the revenue from crop production equals the quantity of crop produced in 1000 MT of corn-equivalent multiplied the price of the crop output in USD/MT, or $Q_{g,l}^{crop} P_{g,l}^{crop}$. With trading, I add the revenue from permit sales, equal to the grid cell- and land type-level environmental permit output, $Q_{g,l,c}^{permit,m}$ multiplied by the market price of permits, $P_r^{permit,m}$. The subscript c denotes the abatement action that generates the permit is that of the cropland activity—or the reduction in applied nitrogen fertilizer to cropland. The superscript m denotes the type of permit, with $m = N$ denoting a nitrogen permit, $m = G$ denoting a greenhouse gas permit, and $m = H$ denoting habitat permits. Grid cell- and land type-level revenues from crop production with trading are then $Q_{g,l}^{crop} P_{g,l}^{crop} + \sum_m Q_{g,l,c}^{permit,m} P_r^{permit,m}, m = N, G$. The quantity of permits generated from abating each type of emissions equals

$$(9) \qquad Q_{g,l,c}^{permit,m} = \varepsilon_{g,l,c}^{m} (\bar{Q}_{g,l,c}^{emissions,m} - Q_{g,l,c}^{emissions,m}), \;\; m = N, G,$$

where $\bar{Q}_{g,l,c}^{emissions,m}$ is the producer's profit-maximizing level of emissions in the absence of trading. Not all producers that generate emissions reductions will necessarily receive permit revenues, as some producers may not enroll in a given market (e.g., due to perceptions of "loss of autonomy regarding farm operations; opportunities for increased government oversight; and negative publicity about agricultural pollution"; Breetz, et. al, 2005). I denote the enrollment rate for each market and grid cell as $\varepsilon_{g,l,c}^{m}$. Currently within the model with fixed enrollment rates, overall enrollment changes as the total supply of permits changes. In principle, this enrollment rate should be endogenous and will depend on crop and permit prices. Here, I assume it is exogenous because of uncertainty as to the factors that determine enrollment rates, such as fixed costs, preferences, and perceptions. Enrollment rates could be based solely on price changes within the respective permit markets, but this would require a functional form or parameterization method that is beyond the scope of this project.

The nest system in Figure 4 shows the tradeoff between allocating inputs to the competing actions of agricultural commodity production and environmental permit production. For example, if a producer uses land, labor, and capital to produce crops, then she foregoes the opportunity to generate and sell environmental permits. This means that the cost of using polluting inputs (i.e., $Q_{g,l}^{nitro}$) comprises the actual input cost plus foregone permit revenues. Therefore, the quantity of nitrogen and greenhouse gas permits—represented by $Q_{g,l,c}^{permit,m}$ with $m = N, G$—is added below $Q_{g,l}^{nitro}$ within the nesting system to indicate that these permits are implicit inputs. This is because there is less potential for nitrogen and CO_2 equivalent permits to be produced as more nitrogen fertilizer is applied. The same is true for land used for agricultural production instead of conservation; as discussed later in this section, cropland area ($Q_{g,l}^{cropland}$) and wetland area ($Q_{g,l}^{wetland}$) compete for available land area, and nonland inputs are necessary to convert land area to wetlands. I therefore add permits to the land nest in the bottom of Figure 4.

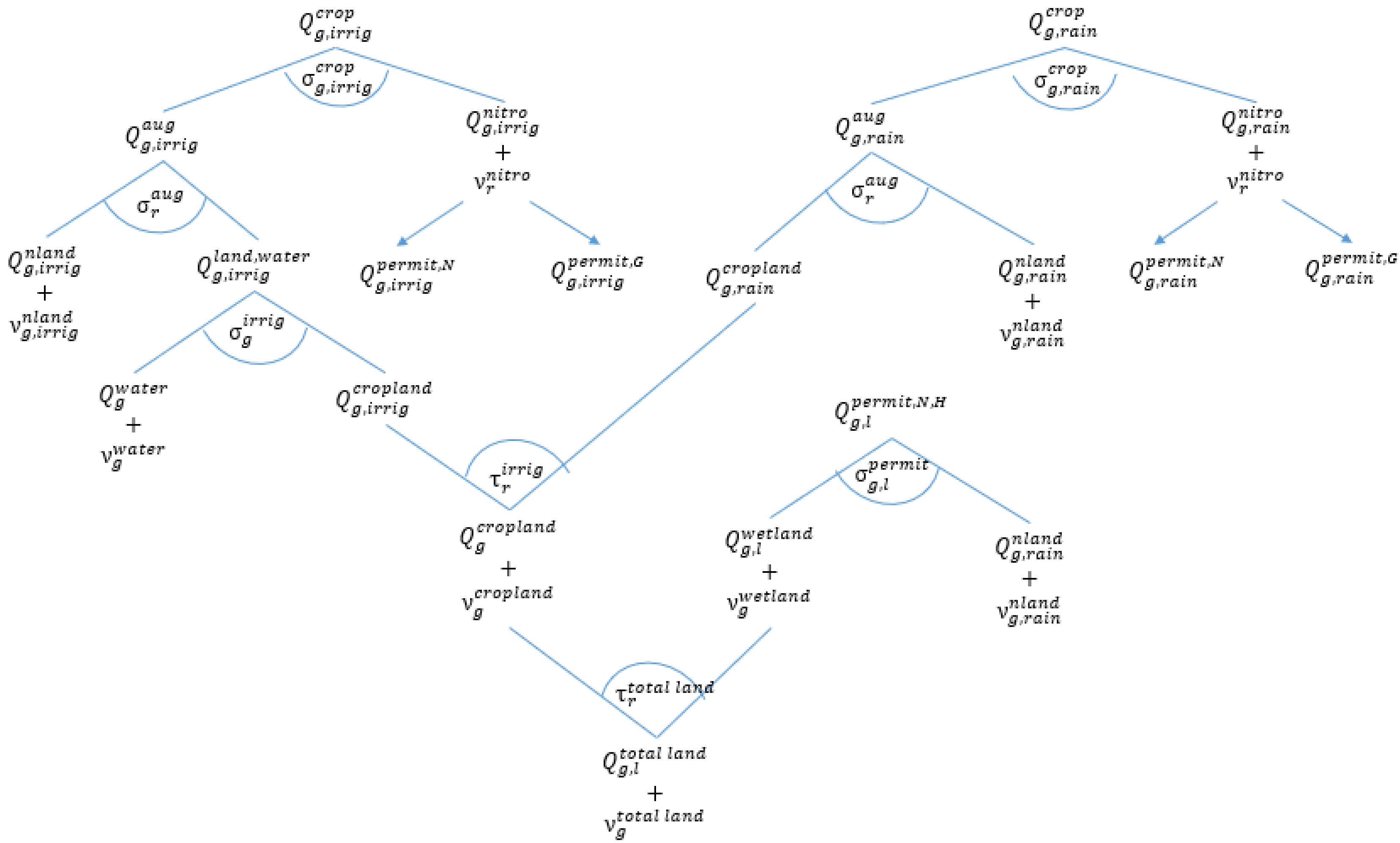

Figure 4: Constant Elasticity of Substitution Supply Nest for Crop Commodities and Permits

I modify the zero-profit condition (4) to include the opportunity costs of ignoring the potential profits from permits

$$(10) \qquad \pi_{g,l}^{NPS} \equiv Q_{g,l}^{crop} P_{g,l}^{crop} - Q_{g,l}^{aug} P_{g,l}^{aug} - Q_{g,l}^{nitro} P_{g,l}^{nitro} + \sum_{m \in \{N,G\}} Q_{g,l,c}^{permit,m} P_r^{permit,m} = 0.$$

To solve for the derived input demands with permit trading, I construct and solve a new optimization problem:

$$(11) \qquad \min_{Q_{g,l}^{aug}, Q_{g,l}^{nitro}} Q_{g,l}^{nitro} P_{g,l}^{nitro} + \; Q_{g,l}^{aug} P_{g,l}^{aug} - \sum_{m \in \{N,G\}} P_r^{permit,m} Q_{g,l,c}^{permit,m}$$

subject to (2) and (7).

The FOCs from (11) and the zero-profit condition (10) yield input demands (5) and

$$(12) \qquad Q_{g,l}^{nitro} = \left(\theta_{g,l}^{crop}\right)^{\sigma_{g,l}^{crop}-1} \left(\frac{P_{g,l}^{crop}}{P_{g,l}^{nitro} + \sum_{m \in \{N,G\}} P_r^{permit,m} \delta_{g,l}^{m}}\right)^{\sigma_{g,l}^{crop}} \left(\theta_{g,l}^{nitro}\right)^{\sigma_{g,l}^{crop}-1} Q_{g,l}^{crop}$$

along with equilibrium crop price, $P_{g,l}^{crop}$.

Note that the demand for nitrogen fertilizer, (12), is similar to (6) except the effective opportunity cost of fertilizer use (denominator of second right hand-side term in parentheses) increases to account for foregone permit revenues. Further, both the supply of crops and permits depend on relative market prices—as determined by their respective demand functions—and input costs. The market-clearing condition for water quality permits is $\sum_{l\times g\in r} Q_{g,l}^{permit,N} = \sum_{g\in r} q_g^{permit,N}$, where $q_g^{permit,N}$ is the quantity of N permits demanded by PSs derived at the end of this section. I assume the prices of greenhouse gas and habitat permits are exogenous.

For the BMP of N reduction, permit generation depends on decisions made by the crop producer. However, for the BMP of wetland construction, the landowner makes permit generation decisions. The addition of the CET nest below the cropland land-type CET nest in Figure 4 represents the inclusion of a landowner at the grid cell-level. Prior to making any crop production decisions, the landowner first must decide to allocate land to either cropland or wetland. Therefore, the demand for wetland area is related to the relative rental rates for cropland and wetland, $R^{cropland}$ and $R^{wetland}$. The landowner's decision also depends on the total land available in a given grid cell, which depends on the constant elasticity of transformation (CET) parameter, or the share of land devoted to each use at the benchmark, $\alpha_{g,l}^{j}, j \in \{cropland, wetland\}$, and the CET parameter, κ, between cropland and wetland. The CET parameter, κ, is a function of the CET transformation elasticity, τ. Formally, the landowner chooses the quantity of land to place in both cropland and wetlands to solve

(13) $$\max_{Q_{g,l}^{cropland},Q_{g,l}^{wetland}} R^{cropland} Q_{g,l}^{cropland} + R^{wetland} Q_{g,l}^{wetland}$$

$$\text{s.t. } Q_{g,l}^{total\ land} = [\alpha_{g,l}^{cropland}(Q_{g,l}^{cropland})^{\kappa} + \alpha_{g,l}^{wetland}(Q_{g,l}^{wetland})^{\kappa}]^{\frac{1}{\kappa}}.$$

(14) $$Q^{j} = Q^{total\ land}\left(\frac{R^{j}}{R\alpha^{j}}\right)^{\tau} \text{for } j \in \{cropland, wetland\}$$

$$\text{where } \tau = \frac{1}{\kappa-1} \text{ and } R = \left[\sum_{j\in\{C,W\}} \alpha^{j^{-\tau}} R^{j^{1-\tau}}\right]^{\frac{1}{1-\tau}}.$$

Once the landowner decides to provide wetland area, the profit function for the wetland provider becomes

$$(15) \qquad \pi_{g,l}^{wetland\ provider} = \sum_{m\in\{N,H\}} P^{permit,m} Q_{g,l,w}^{permit,m} + P^{cons} Q_{g,l}^{wetland} - P_{g,l}^{nland} Q_{g,l}^{wetland} - P_{g,l}^{wetland} Q_{g,l}^{wetland}$$

This profit function comprises four terms. The first two terms represent revenues to the wetland provider. Specifically, the first term is the permit revenue the wetland provider earns for the nitrogen and habitat permits corresponding to wetland activity, denoted by the w subscript of $Q_{g,l,w}^{permit,m}$. The second term represents incentive payments for conservation practices from the federal and state government. The per-hectare payment is P^{cons}, and $Q_{g,l}^{cons}$ represents the hectares that are eligible for compensation. One form of compensation is the Conservation Reserve Enhancement Program (CREP), a voluntary program implemented by the USDA in partnership with state governments to promote conservation practices to improve environmental quality. Through CREP, agricultural producers within the CBW can receive payments for wetland restoration. The CREP incentive payments vary by state and county, because each state decides on their own cost-share terms, and the base soil rate varies by county. Based on the CREP payment scheme, I calculate the value of P^{cons} for eligible grid cells.

The two terms on the last line of (15) ($P_{g,l}^{nland} Q_{g,l}^{wetland}$ and $P_{g,l}^{wetland} Q_{g,l}^{wetland}$) represent costs to the wetland provider. The first term, $P_{g,l}^{nland} Q_{g,l}^{wetland}$, is the conversion cost. This cost equals $P_{g,l}^{nland}$, the per-hectare non-land costs related to capital and labor for construction and installation, multiplied by the area of wetland converted in each grid cell, $Q_{g,l}^{wetland}$. The index of rent for one hectare of wetland—which can be interpreted as the opportunity cost of setting the land aside for conservation—is $P_{g,l}^{wetland}$, whereas wetland non-land costs, $P_{g,l}^{nland}$, depend on grid cell-level soil properties and the state in which the grid cell is located. To classify the share of each grid cell that is suitable for wetland construction, I use the EPA's Potential Restorable Wetlands on Agricultural Land (PRW-Ag) dataset (Environmental Protection Agency, 2018d). PRW-Ag classifies wetland suitability at the 30-meter resolution into four categories, including unsuitable, low, medium, and high wetland restoration potential, based on land coverage, soil drainage, and wetness. I take a weighted share of the amounts of low, medium, and high wetland restoration potential land to calculate to grid cell-level shares of potential wetland area. By mapping current wetland area, provided by the National Land Cover Database, I can observe the share of wetlands within the grid cell (Multi-Resolution Land Characteristics Consortium (U.S.), 2016). Then by using the USDA's Conservation Reserve Program Statistics, I can differentiate between naturally occurring wetlands and wetland area that has been constructed or restored for the purpose of conservation payments (Farm Service Agency, 2020). The final term in (14) is foregone rents from wetland installation, equal to index of rent for one hectare of wetland (or the opportunity cost of setting the land aside for conservation), $P_{g,l}^{wetland}$, times the wetland area.

Water quality permit production from wetland area depends on nitrogen removal efficiency. I assume removal efficiency affects leaching from each grid cell linearly as in (7). I model the nitrogen removal rate following Simpson and Weammert (2009), whose approach is used by the Chesapeake Bay Program for water quality modeling. The removal rate depends on the proportion of the grid cell containing wetland area—the share of naturally occurring wetland area plus converted wetland within each grid cell—represented by ρ_g, as well as a fitted value (7.90) from an analysis of reported studies in the literature:

(16) $$\delta_{g,l}^{wetland} = 1 - e^{-7.90(\rho_g)}.$$

To calculate the nitrogen permits generated from wetland activity, $Q_{g,l,w}^{permit,N}$, I multiply the difference between the reference nitrogen removal rate and the nitrogen removal rate corresponding to the increase in constructed wetland ($\bar{\delta}_{g,l}^{wetland} - \delta_{g,l}^{wetland}$) by the reference amount of nitrogen ($\bar{Q}_{g,l}^{nleach}$) that leaches off of the cropland area within the grid cell:

(17) $$Q_{g,l,w}^{permit,N} = \varepsilon(\bar{\delta}_{g,l}^{wetland} - \delta_{g,l}^{wetland})\, \bar{Q}_{g,l}^{nleach}.$$

Within the partial equilibrium framework, changes in permit quantities are calculated as percentage changes from a baseline level which requires non-zero benchmark quantities for permits within the database. Therefore, I assume there is some trading occurring at the benchmark prior to simulating how stacking affects the permit trading markets. WQT markets are relatively new, and data regarding trade amounts and prices is limited. In order to anchor the simulations with benchmark values for the quantities of permits related to reductions in fertilizer application and wetland construction, I estimate baseline permit quantities based on data relating to these two BMPs.

The database for the model was constructed with 2010 data, and the grid cell fertilizer application rates are based on USDA 2010 estimates of nitrogen fertilizer use. For greenhouse gas permits and nitrogen permits from cropland, I use data on trends in fertilizer use in the CBW. Total nitrogen application within the CBW peaked in 2000, before decreasing by 15.42% by 2012 (Keisman et. al, 2018). I use this 15.42% change in nitrogen fertilizer application to create grid cell-level reference nitrogen quantities by increasing the 2010 database quantities to the 2000 level. I then use the 2000 grid cell-level reference nitrogen quantities to calculate the baseline emissions for nitrate leaching and nitrous oxide, $\bar{Q}_{g,l,c}^{emissions,m}$. With the baseline emissions levels, I am then able to calculate the corresponding baseline quantities of nitrogen permits from cropland and greenhouse gas permits at the grid cell level as shown by (9).

For baseline habitat permits and nitrogen permits from the construction of wetlands, I use estimates on the change in constructed wetland area on cropland from the Chesapeake Bay Program (2020). I reduce the amount of restored wetland area at the grid-cell level to that of 2000 reference levels. I then calculate the grid-cell level reference nitrogen removal rates based on (16) and then use these to calculate the baseline nitrogen permits from wetland activity following (17). The baseline habitat permits are based on the grid-cell level changes in wetland area. However, the reduction in fertilizer and the construction of wetland habitat are not entirely due to ecosystem markets, and therefore the resulting permit amounts are discounted via the enrollment rates.

Point Source Permit Demand

Next, I consider industrial PS demand for nitrogen emissions. PS producers maximize profits, $\pi_{g,l}^{PS}\left(q_{g,l}^{nitro}\right)$, subject to the constraint that any emissions they generate, $q_{g,l}^{nitro}$, must be covered by a permit. Formally,

$$\max_{q_g^{nitro}, q_g^{permits,N}} \pi_g^{PS}\left(q_g^{nitro}\right) - P_r^{permit,N}\ q_g^{permits,N} \tag{18}$$

$$\text{s.t. } q_g^{nitro} \leq \hat{q}_g^{permits,N} + q_g^{permits,N}$$

where $\hat{q}_g^{permits,N}$ is the original allocation of permits issued to the PS and $q_g^{permits,N}$ is the quantity of permits purchased. Assuming an interior solution and substituting the constraint into the objective function yields the unconstrained problem

$$\max_{q_g^{nitro}} \pi_g^{PS}\left(q_g^{nitro}\right) - P_r^{permits,N}\left[q_g^{nitro} - \hat{q}_g^{permits,N}\right] \tag{19}$$

with first-order condition

$$\frac{d\pi_g^{PS}}{\partial q_{gl}^{nitro}} - P_r^{permits,N} = 0 \tag{20}$$

which, when solved, implies the nitrogen demand function $q_g^{nitro}\left(P_r^{permits,N}\right)$. Substituting this demand function into the constraint in (18) yields PS demand for permits, $q_g^{permits,N}\left(P_r^{permit,N}\right)$.[1] The market-clearing condition requires that the quantity of permits

[1] Technically, PS demand for permits is also a function of the permit allocation, $\hat{q}_g^{permits,N}$. I am implicitly assuming that each PSs' emissions are subject to a binding cap.

demanded at the regional level equal the quantity of permits supplied:

(21) $$\sum_{g\in r} q_g^{permits,N}\left(P_r^{permit,N}\right) = \sum_{g\in r} Q_g^{permits,N}\left(P_r^{permit,N}\right).$$

ECHO contains discharge monitoring data that has been reported to the EPA (Environmental Protection Agency, 2019). In particular, ECHO contains nitrogen loading data and locational information for each PS in the CBW[2]. The locational information enables me to map each polluting facility to the grid cells defined within SIMPLE-G.

ECHO provides information on 97 PSs categorized as significant as defined by Van Houtven, Loomis, Baker, et al. (2012):

> "'Significant' point sources (SigPS) include municipal wastewater treatment facilities with wastewater capacity exceeding 0.4 million gallons per day (MGD) and industrial wastewater discharge facilities with ≥ 3,800 lbs/yr annual total phosphorus or ≥ 27,000 lbs/yr total nitrogen loads. SigPS do not include any federally regulated urban stormwater sources (MS4s) or CAFOs as defined under the Clean Water Act."

[2] ECHO also provides facility characteristics along with loading information, which I attempted to use to calculate grid cell-level permit demand elasticities from marginal abatement cost curve estimations. Due to data limitations, I was unable to confidently estimate elasticities for each of the grid cells containing point sources. I assume a permit demand elasticity of -5, which is similar in order of magnitude to the estimations.

I match these significant PSs to 94 different grid cells in the CBW, as shown in Figure 5.

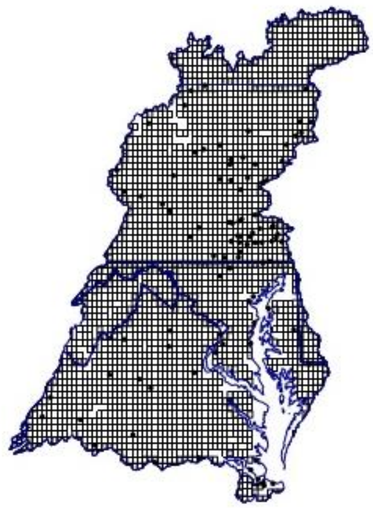

Figure 5: Location of Significant Point Sources

Of these 94 grid cells, 81 (86%) contain data on crop production and are modeled within the SIMPLE-G All Crops model. These significant PSs also match to 35 (65%) out of the 54 HUCs within the CBW and are contained within Maryland, Pennsylvania, Virginia, and DC. Because 19 of the HUCs—along with the states Delaware, New York, and West Virginia—do not contain significant point sources, I assume these markets do not have demand for nitrogen permits. Therefore, these markets do not contain active trading markets.

Experimental Design

I test both the scale and the scope of ecosystem trading markets by modeling trading across varying market sizes and with multiple levels of permit stacking allowed. The stacking levels represent the expansion of the scope of the markets, while the market sizes correspond to the expansion of the scale of the markets. The scale of the market ranges from trading within individual HUCs—the smallest of the market sizes—to trading within state boundaries contained in the CBW, to trading across the entire CBW. As the market sizes increase, the grid cells that were previously excluded from trading due to lack of significant PSs can participate as they are included in areas with active markets. The experiments discussed in the Results section follow a 2x3 experimental design, in that I model two levels of stacking for each of three different market sizes, as shown in Figure 6.

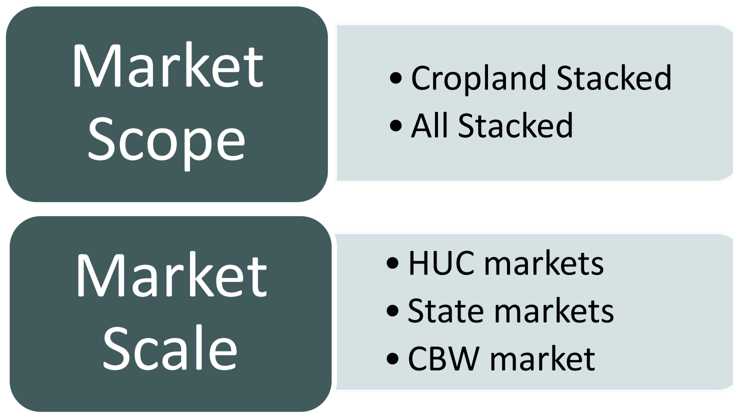

Figure 6: Experimental Design

The scope of the markets refers to the ability to "stack" permits that are produced from a single abatement action. Allowing the crop producer to collect revenues from multiple types of permits on spatially overlapping areas allows me to examine the effect of permit trading on additionality. Additionality holds if the additional permit revenue streams incentivize the crop producer to increase abatement. To test for additionality, I run the model initially with estimated enrollment rates allowing portions of each grid cell to be involved in only one of the conservation practices. I then compare the outcome with a counterfactual case in which producers can collect revenues from GHG and wildlife habitat permits, as well as water quality permits from spatially overlapping areas of the grid cells.

This comparison allows me to determine whether additional environmental benefits are generated when credit stacking is allowed.

Given the limited data on permit trading amounts, I assume 25% of NPS polluters participate in nitrogen/WQT markets another 25% participate in greenhouse gas markets. Likewise, I assume 25% of wetland providers are enrolled in nitrogen trading markets and 0.1% are enrolled in habitat markets. Through correspondence with ecosystem credit brokers, I attain a benchmark price of $3.92 per nitrogen permit, in which a one-pound reduction in nitrogen pollution equals one permit. For the greenhouse gas permit price, I use auction data from the Regional Greenhouse Gas Initiative to attain a baseline of $1.92 per MT of CO_2 equivalent emissions. In the case of habitat permits, I start the payment at $1,000 per hectare. This scenario represents the benchmark case in the database without the stacking of permits on spatially overlapping area.

I model permit stacking by aggregating the enrollment rates for the different permit types generated by the same BMP. I model two stacking scenarios. For the "Cropland Stacked" experiment, I assume agricultural producers can collect permit revenues for both nitrogen and greenhouse gas abatement related to a reduction in fertilizer application. Since I assume 25% of producers are initially enrolled in nitrogen and greenhouse gas markets, and fertilizer application jointly influences both pollutants, stacking in this manner implies 50% enrollment rate in the stacked market. The "All Stacked" experiment is the same, except that I add the enrollment rates of producers supplying wetland-related permits. Therefore, the All Stacked case has an enrollment rate of 50% supplying nitrogen permits from cropland and greenhouse gas permits, as well as an enrollment rate of 25.1% supplying nitrogen permits from wetlands and habitat permits.

RESULTS

Table 1 presents the results of these two stacking scenarios across the different market size specifications.

Table 1: Results of Key Variables from Additionality Experiments with Varying Enrollment Rates under each Market Size

	Cropland Stacked			All Stacked		
Key Variables	HUC Markets	State Markets	CBW Market	HUC Markets	State Markets	CBW Market
N permits wetland	5,464lbs	12,652lbs	13,594lbs	23,456lbs	38,626lbs	38,190lbs
Wetland area	21,514ha	21,529ha	21,514ha	23,557ha	23,886ha	23,627ha
Habitat permits	21.51ha	21.53ha	21.51ha	5,564ha	5,849ha	5,823ha
N permits cropland	4,508,311lbs	5,565,875lbs	6,516,067lbs	-1,417lbs	7,392,510lbs	7,878,728lbs
Avg price of N permit	$4.21	$4.14	$3.97	$5.83	$3.92	$3.87
Applied N fertilizer	258,318MT	254,837MT	252,223MT	275,577MT	243,386MT	247,676MT
Avg N application	0.1360MT/ha	0.1344MT/ha	0.1331MT/ha	0.1448MT/ha	0.1284MT/ha	0.1309MT/ha
Cropland area	1,899,889ha	1,896,172ha	1,895,666ha	1,902,547ha	1,895,870ha	1,891,778ha
GHG permits	96,789MT	104,797MT	103,150MT	55,031MT	125,071MT	121,483MT

Equilibrium permit output increases with both the scale and scope of the markets. The largest category of ecosystem permits is the nitrogen permits from cropland area, where each permit represents a pound of N, with 4.5 million, 5.6 million, and 6.5 million lbs traded in the HUC, State, and CBW market specifications, respectively. Cropland permit generation is directly correlated to the reduction in applied nitrogen to the reduction, which steadily decreases as the markets expand in scale. Likewise, the quantity of nitrogen permits produced from wetland installation increases with the scale of the markets. Equilibrium permit supply is 5,464 lbs, 12,652 lbs, and 13,594 lbs for the HUC, State, and CBW markets, respectively. The wetland area and habitat permits do not vary much across the different scales of the market. In fact, while the CBW market case produces the largest amount of nitrogen permits from wetland implementation, this case does not contain the largest amount of wetland area.

The equilibrium quantity of GHG permits for the HUC, State, and CBW market scenarios, where each permit represents 1 MT of CO_2 equivalent, is 96,789 MT, 104,797 MT, and 103,150 MT, respectively. The State market case produces more GHG permits under the Cropland Stacking scheme, although more fertilizer is applied overall in this case. Figure 7a shows aggregate permit results across market sizes under the Cropland Stacked scenario.

For the HUC market specification in both stacking schemes, there is a tradeoff between nitrogen permits produced from cropland activity and nitrogen permits produced from wetland activity. In the Cropland Stacked scenario, more nitrogen permits from cropland are produced, while there is small amount of nitrogen permits produced from wetland activity. In the All Stacked case, this relationship is reversed due to the increase in the profitability of wetland activity. The increase in the enrollment rate for habitat permits from the Cropland Stacked to the All Stacked case is quite large—from 0.1% to 25.1%—which causes a large increase in habitat permits produced across these stacking scenarios. In the All Stacked case, the total quantities of habitat permits are 5,564 ha for HUC markets, 5,849 ha for State markets, and 5,823 ha for the CBW market, as opposed to ~22 ha for each market size in the Cropland Stacked case.

The aggregate quantities of nitrogen permits produced from cropland activity in the All Stacked case are –1,417 lbs, 7,392,510 lbs, and 7,878,728 lbs for the HUC, State, and CBW markets, respectively. This is a substantial increase in permits from the Cropland Stacked case, except in the case of the HUC market structure due to the tradeoff between cropland and wetland abatement activities. Nitrogen permits from wetlands also increase with the scope of the markets; wetland providers supply 23,456 lbs, 38,626 lbs, and 38,190 lbs worth of permits in the HUC, State, and CBW markets, respectively. These figures correspond to wetland areas of 23,557 ha, 23,886 ha, and 23,627 ha of wetlands.

There is a decrease in the aggregate quantity of GHG permits produced under the All Stacked HUC market case relative to that under the Cropland Stacked case. I attribute this to an increase in applied nitrogen, as the permit production under this experiment is primarily due to wetland activity as opposed to cropland activity. Therefore, the trend for the quantity of GHG permits across the different experimental designs does not follow the same trend as the other permit types, with 55,031 MT, 125,071 MT, and 121,483 MT worth of GHG permits produced under the HUC, State, and CBW markets, respectively. As is the case with the Cropland Stacked case, more GHG permits are produced under the All Stacked State markets case than the CBW market case. The higher amount of GHG permits produced in the State market case is primarily driven by a few grid cells that experience large decreases in nitrogen application. Figure 7b shows aggregate permit volumes across market sizes in the All Stacked scenario.

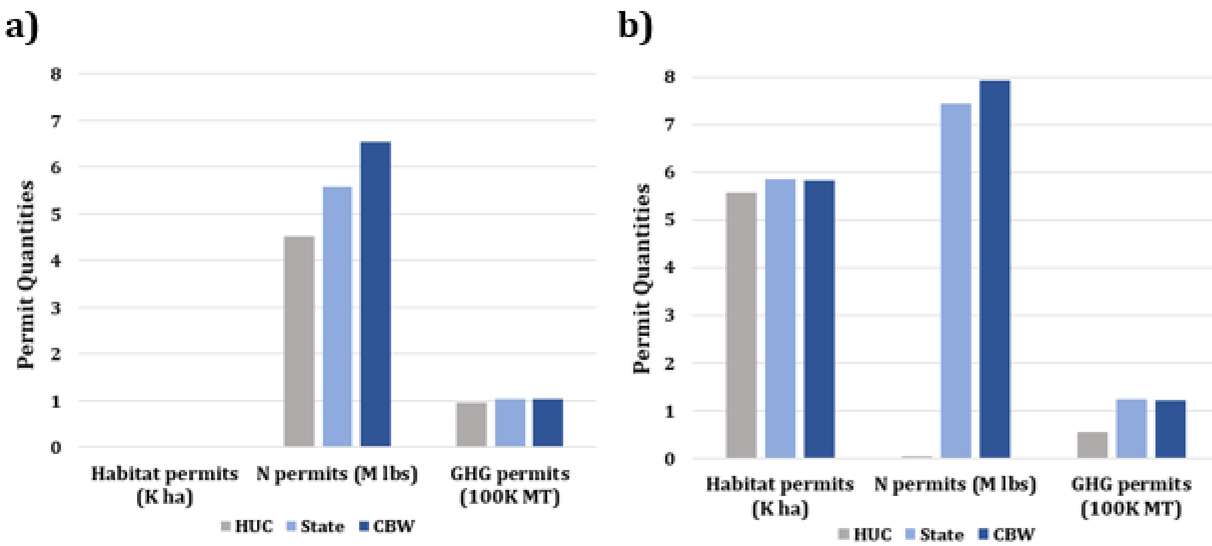

Figure 7: Aggregate Permit Quantities for a) Cropland Stacked and b) All Stacked

Rainfed cropland grid cells have higher leaching parameters overall, so the reduction of nitrogen fertilizer applied to rainfed land allows for the largest reductions in leaching. This leads to more nitrogen permits produced from any given reduction of nitrogen on rainfed area as opposed to irrigated area. This demonstrates that crop producers on rainfed cropland area can earn greater returns from a reduction in nitrogen fertilizer, as they can generate more with the same reduction as a crop producer on irrigated land. When comparing leaching outcomes spatially for the HUC, State and CBW markets under All Stacked, it becomes clear that the nitrogen application reductions are occurring primarily on rainfed cropland as opposed to irrigated cropland, as shown in Figure 8. Maryland experiences the largest leaching reduction, followed by Pennsylvania, Virginia.

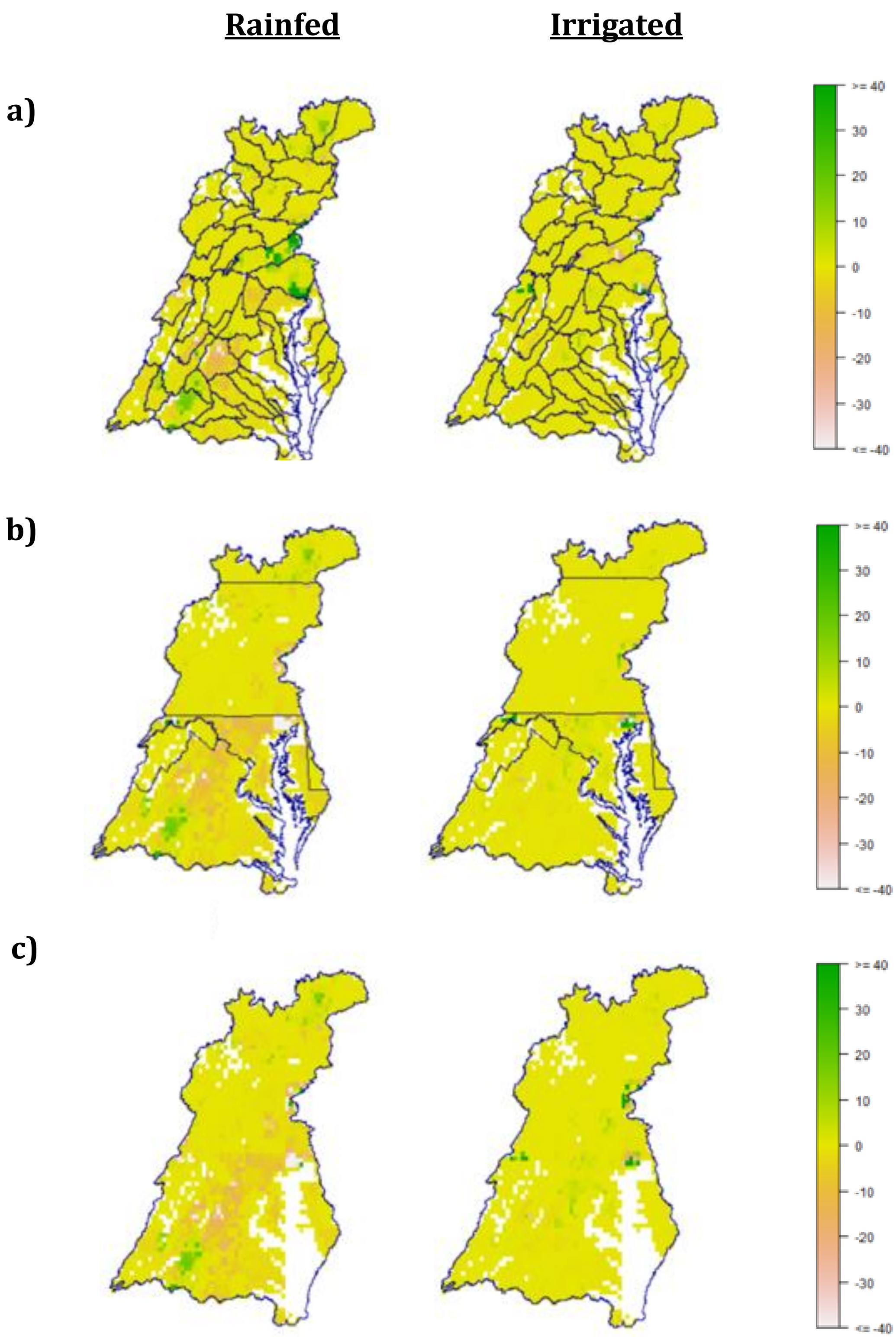

Figure 8: Percentage Changes from Baseline of Nitrogen Leaching on Rainfed and Irrigated Cropland for All Stacked under a) HUC markets, b) State markets, and c) the CBW market

DISCUSSION & CONCLUSIONS

This study lays the foundation for ecosystem credit modeling within the SIMPLE-G framework. While this study focuses specifically on the CBW, it provides a better understanding of the consequences from implementing a multi-state, ecosystem credit stacking market. By modeling NPS permit supply decisions, we observe potential unintended policy consequences and additionality consequences. The results provide insights to both economists and policy makers on the intricacies of WQT and potential measures to alleviate market challenges in the future.

I model ecosystem permit production across various market sizes, with two different stacking schemes, to analyze how the different combinations of scale and scope perform regarding permit production and environmental benefits. One concern with regards to the scope of environmental markets is related to stacking and the additionality principle. If the additional revenue streams generate no additional abatement, then the crop producer received an additional payment for an action that would have occurred without the extra payment. Previous studies show that additionality is violated when BMPs comprise discrete investment decisions with large capital costs (e.g., wetland construction; Lentz, et. al, 2014). Within the SIMPLE-G model, wetland conversion is modeled as a continuous choice once the landowner decides to allocate land to wetland. Given this modeling framework, additionality holds as marginal increases in wetland area results in marginally higher prices per hectare area. However, further analysis is necessary to determine if the constant elasticity of transformation nest is the best modeling framework for wetland construction as a continuous function may not capture the full nature of the investment.

In the case of HUC markets, there should be concern about conservation payments for wetland area possibly crowding out the potential nitrogen credits produced by a reduction in nitrogen fertilizer application. With the stacking of wetland and habitat payments, the producers' decisions within the model show a conflict between nitrogen permits produced from wetland and cropland abatement activities. While the smaller scale of HUC markets may make them easier to manage, this market structure does not result in environmental benefits of the same magnitude as the State and CBW market cases. However, local effects would need to be quantified to ensure that emission "hotspots" do not occur due to the movement of emissions from regions with lower abatement costs. The Chesapeake Bay TMDL is made up of 92 tidal segments that are monitored by stations in Delaware, D.C., Maryland, and Virginia (Environmental Protection Agency, 2010b). These stations capture nutrient loading at the tidal water outlets and may not fully capture the changes at local levels.

The State market simulations show large increases in permit production in comparison with the HUC market simulations. This supports the EPA's recent decisions to encourage state WQT markets to aid in the reduction of water pollution (Ross, 2019). The most active states, in order of most permit production, are Maryland, Pennsylvania, and Virginia. These are the three states within the CBW that currently have WQT markets which allow for NPS participation (Maryland Nutrient Trading Program, 2015; Pennsylvania Department of Environmental Protection, 2016; Virginia Department of Environmental Protection, 2008).

The experiment with the largest market scale and scope, the All Stacked CBW market case, has the largest amount of nitrogen credits produced, which is the primary credit in the market. This experiment, however, does not result in the largest reduction in the amount of nitrogen applied to the field nor the largest increase in constructed wetland area. This is due to the ability to increase the amount of nitrogen permits produced with a smaller reduction in the amount of applied nitrogen and less constructed wetland area. The All Stacked CBW market case allows for all grid cells to participate in abatement actions, as there is nitrogen permit demand throughout the entire region. This supports the potential for interstate trading, which is currently being considered by Pennsylvania and Maryland (Environmental Protection Agency, 2018a; Maryland Nutrient Trading Program, 2015).

Liu et al. (2018) utilizes hydraulic loading information along with climate, soil, and vegetation characteristics to model the annual percent decrease of leaching at the grid cell level. Their study finds that wetland construction is the most effective best management practice for reducing the leaching charge they employ to incentivize a 45% reduction in nitrogen leaching. In comparison, my study demonstrates that a reduction in the quantity of applied nitrogen on cropland is the preferred abatement action. My study differs from Liu et al. (2018) in a few key ways. The first difference is in the modeling approach. Liu et al. (2018) determine the "leaching charge" (analogous to a tax) required to attain a 45% reduction in nitrate leaching under various BMPs. They do not model permit trading explicitly, nor do they allow producers to choose from among a suite of practices to generate abatement. Liu et al. (2018) model wetland restoration via a technological change that increases the input-biased efficiency for land, $\theta_{g,l}^{land}$. By modifying the productivity of land so slightly less land is needed for production, a small percentage of cropland can be set aside as wetland area. Their approach differs from the land-use decision via the CET nest I use in this study.[3] My results imply that working lands practices (e.g., nitrogen application reduction) are more cost-effective than wetland conversion given the implied permit caps. Moreover, the implied permit caps are different. The highest leaching reduction within my simulations (12.9% in the All Stacked CBW market case) is much smaller in magnitude than the 45% reduction Liu et al. (2018) consider. These differences could all be contributing to the difference in the corresponding effectiveness of wetland restoration for nitrate leaching mitigation.

Future Research

Future work should quantify changes in welfare measurements such as producer and consumer surplus to better understand the changes to social net benefits from changing market scale and scope. I perform the experiments within a regional model of SIMPLE-G that does not calculate the changes in crop commodity prices. In order to analyze how these policies impact the crop market, future work must integrate this framework into the full version of the SIMPLE-G model. Changes in the input use of fertilizer and land affect crop output, which would alter the corresponding market prices in the full model. Overall, there is a decrease in the quantity of nitrogen fertilizer leading to a decrease in crop output within the experiments of this study. This loss in crop output would translate to an increase in the corresponding price that would incentivize additional crop producers to enter the market. While the average nitrogen application rate would remain low, additional cropland would

[3] We assume grid cell-level transformation elasticities for the total land CET nest based on the amount of potential wetland area within each grid. Further work can be done to investigate alternative calibration methods.

enter production, which could lead to a higher level of nitrogen fertilizer being used than what is represented in these results.

Additionally, future work should develop the permit demand side of the model by looking into additional methods for calibrating the nitrogen permit demand elasticities, which was hindered by data availability issues. Future studies might investigate alternative data sources as well as methods for the calibration of the marginal abatement cost curves that are used to calculate the permit demand elasticities. For the simulations, I assign nitrogen removal efficiency rates for the hydric soil-dominant grid cells based on changes in the proportion of the grid cell's total wetland area—based on the construction of additional wetland area—compared to the baseline removal rate from naturally occurring wetlands. Further work could estimate unique nitrogen removal efficiency rates, based not only on the proportion of the wetland area, but also on grid cell-specific characteristics.

Further analysis can be done to ensure that the leaching parameters are as precise as possible and estimate how different BMPs affect this crucial parameter. I simplify the original quadratic leaching function within SIMPLE-G to a linear approximation to enable the derivation of the nitrogen input demand function (12). The linear approximation may underestimate the leaching output corresponding to the nitrogen application rates in comparison to the quadratic form, and hence, may underestimate nitrogen permit generation from nitrogen application reductions within the model results. As water quality monitoring abilities and environmental modeling advance, more precise estimations of NPS emissions and abatement are becoming available. With the expansion of remote sensing, precision agriculture, and big data, targeted conservation practices become more feasible. Remote sensing could also aid in the nutrient management plans by targeting fertilizer application based on field specific needs which could help lower the leaching potential, especially for rainfed cropland area. These advances enable researchers to be better informed on policy recommendations and analyses.

An extension of this work could incorporate soil carbon data to model greenhouse gas emissions from converting land to cropland. This would enable a deeper analysis of additionality by capturing the tradeoff between intensification and extensification for crop production. To capture a more complete picture in the changes of greenhouse gas emissions, further work would need to be done on the effects of wetlands. While wetlands are carbon sinks—meaning they capture carbon and store it in the soil—they can also be a source of methane. Wetlands also remove phosphorus from water—another nutrient that causes hypoxia—and aid in mitigating soil erosion.

Phosphorus and sediment reductions are additional potential permit types that could be incorporated into the model. While additional permit types could allow farmers to mitigate conservation risks by allowing them to construct a conservation portfolio, the model still makes underlying assumptions about farmer behavior. The SIMPLE-G model assumes that all producers are rational, profit maximizers, and does not account for risk preferences or environmental conservation preferences. While our enrollment rate allows us to investigate how the model responds to varying participation rates, future studies could integrate behavioral models to understand the probability of enrolling in ecosystem markets and adopting conservation practices given producer preferences and perceptions.

APPENDIX

Given uncertainty about the parameters of the PS permit demand function, I perform a sensitivity analysis of the permit demand elasticity for nitrogen permits. I use a permit demand elasticity of –5 in the experiments shown in Table 1. For the sensitivity analysis, I vary the permit elasticity by ±20% (to –4—the "Inelastic Permit Demand" case—and –6—the "Elastic Demand" case). I implement this change across all market size specifications within the All Stacked scenario. Table 2 summarizes the results.

Table 2 shows that the model is largely insensitive to the permit demand elasticity; the results are not qualitatively different from those derived under the baseline specification in Table 1. The results for the All Stacked State Market case are not included as there were issues in performing the experiment. Further investigation is needed to find the root of this issue and mitigate the volatility of the amount of applied nitrogen fertilizer.

Table 2: Results of Key Variables from Sensitivity Analysis of Nitrogen Permit Demand Elasticity in All Stacked Design*

	Elastic Permit Demand			Inelastic Permit Demand		
Key Variables	HUC Markets	State Markets	CBW Market	HUC Markets	State Markets	CBW Market
N permits wetland	33,311lbs	35,904lbs	38,381lbs	35,220lbs	—	39,964lbs
Wetland area	23,601ha	23,728ha	23,655ha	23,607ha	—	23,615ha
Habitat permits	5,087ha	5,814ha	5,952ha	5,819ha	—	5,937ha
N permits cropland	7,265,285lbs	7,798,502lbs	11,114,122lbs	5,476,828lbs	—	9,436,629lbs
Avg price of N permit	$4.05	$4.05	$3.65	$3.71	—	$3.13
Applied N fertilizer	250,489MT	249,332MT	238,391MT	253,930MT	—	240,307MT
Avg N application	0.1321MT/ha	0.1316MT/ha	0.1261MT/ha	0.1340MT/ha	—	0.1272MT/ha
Cropland area	1,895,959ha	1,895,337ha	1,889,764ha	1,894,342ha	—	1,888,863ha
GHG permits	115,145MT	118,613MT	142,689MT	103.816MT	—	136,716MT

* Inelastic is defined as a nitrogen permit elasticity of -4, while elastic is -6.

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