Three Essays on the Economics of Land Use, Environmental Value, and Public Spending

by

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ABSTRACT

Across the world, public spending on government programs profoundly alters land use, preservation of environmental value, and the wellbeing of rural populations. These essays explore three such programs and derive lessons for improving their targeting. Chapter 1 tests the effect of conservation easement tax incentives on land conservation in Virginia, using a difference-in-difference design around a 2002 tax reform. This finds that the environmental quality distribution of easements is wide and matches the statewide quality distribution of all undeveloped land, suggesting the program has considerable room to improve targeting. Increasing tax incentives attract donations of similar or lower quality, but targeting tax incentives only at high-quality land would substantially increase high-quality acres at a cost of 1.18 low-quality acres per high-quality acre. Chapter 2 investigates the targeting of short-term incentives for long-term behavior change, focusing on the case of the EQIP agricultural incentives program. The model connects the short-term and long-term effects of incentives as products of the immediate adoption costs and long-term repeated costs and benefits of a practice. If populations vary primarily by adoption cost, targeting groups with the greatest short-term effect will also maximize the long-term effect. If populations vary primarily by long-term costs and benefits, the groups with the greatest short-term impact are those for whom the practice is highly unprofitable in the long run, and a program can improve long-term impacts by instead targeting those for whom the practice is slightly profitable in the long run. A discontinuity analysis comparing successful and unsuccessful EQIP applicants shows that EQIP induces significant short-term change. Chapter 3 investigates the behavior of Mongolian livestock markets after severe weather shocks, and the role that a livestock insurance program may play in smoothing shocks. During severe Mongolian winters, livestock sales increase and prices fall as credit-constrained nomadic herders look to make necessary investments to protect their remaining herd. National integration in livestock markets absorbs a significant share of the weather-related shocks, as 40-60% of district price risk is due to national market fluctuations and 20-40% is due to province effects. This paper finds that national mortality strongly drives price variations, and livestock insurance reduces sales during high-mortality periods.

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Biographical Sketch

Kelsey Larson was raised in Bozeman, Montana and graduated from Bozeman High School in 2012. She received her B.A. in Economics from Yale University in 2016, graduating summa cum laude. At Yale she was an editor for the Yale Globalist. After graduation she then worked as a research analyst for Dean Karlan at Innovations for Poverty Action from 2016 to 2018. At MIT, she has received the Martin Family Fellowship for Sustainability, a National Science Foundation GRFP Fellowship, and the Lincoln Institute of Land Policy C. Lowell Harriss Dissertation Fellowship. She was a 2023 National Tax Association Climate Fellow, and she has received research grants from the Environmental Defense Fund and the MIT George and Obie Shultz Fund. Her research focuses on the intersection between land use, environmental value, and public spending.

After her PhD she has accepted a position as an Assistant Professor of Agricultural Economics at Montana State University in Bozeman.

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Chapter 1

The Environmental Value of Private Land Conservation and the Role of Conservation Easement Tax Incentives

1.1 Introduction

Much of the United States' conservation happens on private land. Conservation easements, where a landowner retains the right to use and sell their land but permanently gives up the right to develop it, now protect almost 56 million acres of land, an area more than eight times the size of Massachusetts (Land Trust Alliance 2021). The area under private conservation has ballooned in part due to substantial federal income tax incentives that have been in place since 1976 and expanded in 2006 and 2015 (Elkind 2017; Colinvaux 2012). These federal tax deductions now cost the government almost as much as the entire National Park Service (Looney 2017b), and fourteen states offer additional tax breaks (Land Trust Alliance 2019). Policymakers and conservationists are considering further expansions to these tax credits to help meet the 30x30 commitment to conserve 30% of Earth's land and water by 2030, an ambitious goal shared by the Biden administration and the 196 signatories of the Kunming-Montreal Global Biodiversity Framework, which would more than double the share of land protected compared to 2020 (Convention on Biological Diversity 2022; Gibbens 2021). While the scale of private land conservation in the US is unique, several other countries are currently expanding similar projects, such as the UK's 2021 conservation covenant law (Niker 2022) and France's "real environmental obligations" system created through the 2016 Biodiversity Law (Racinska and Vahtrus 2018).

Advocates support these policies as a way to preserve privately held natural lands more cheaply than purchasing them. However, these programs' designs leave room for land that produces little environmental value to end up permanently protected, possibly at a greater fiscal cost than the environmental benefit is worth. The size of the tax benefits given for a parcel of land depends on the development value of the parcel, defined as the difference in a parcel's market value with and without the restrictions of an easement. The benefits do not vary based on the environmental benefits provided by a parcel as long as the parcel preserves some minimum environmental, historical, or "open space" benefit and the donor can find a government or nonprofit land trust willing to accept the donation (J. Sundberg 2013).

Under this policy, private conservation areas range from pollutant-filtering wetlands housing endangered species to isolated stands of trees bordering golf courses. Elkind 2017 and others have documented cases of easements placed on parcels with low environmental quality but high development value. Quality greatly matters because the most valuable ecosystems can provide environmental services orders of magnitude more valuable than those of low-quality areas. For example, Ingraham and Foster 2008 estimates that New England grasslands in the National Wildlife Refuge system deliver approximately \$60 in combined carbon storage, water quality and regulation, and habitat provision services per acre per year while wetlands provide \$2,670. Even when limiting analysis to water-regulation services provided by riverbanks in the Chesapeake Bay watershed, different stretches provide from less than \$1,730 to more than \$8,400 in annual value per acre (Phillips and McGee 2016). As such, low-quality easements may receive large tax deductions but provide offer little benefit(Colinvaux 2012). When poorly located, private conservation may even cause a net loss of environmental value by pushing development away from low environmental quality land and onto high value parcels. In light of this, numerous experts have recommended investigating the environmental value of private conservation lands (Colinvaux 2012; Merenlender et al. 2004; Fishburn et al. 2009), but research on this remains scarce.

In this paper, I explore the environmental value of private conservation land and test the marginal quality of land attracted by increases in tax incentives. I investigate this in the state of Virginia, which has one of the nation's largest conservation easement subsidy programs (J. O. Sundberg 2011) and may serve as a model for states considering expanded private conservation programs. I measure environmental value using the Virginia Natural Landscape Assessment (VaNLA), a detailed set of conservation priority maps made in 2007 by the Virginia state government. These maps rank land according to its value for meeting the State of Virginia's conservation goals for biodiversity, forest conservation, agricultural preservation, recreation, and water quality. Using these measures of environmental quality allows this paper to directly test whether private conservation in Virginia is contributing to the state's established goals for conservation spending. It also provides a clear metric to compare the contributions of public and private conservation.

This paper's first contribution is to establish some basic and previously unexamined empirical facts about the conservation priority of private conservation parcels in Virginia. I find that the environmental quality of private conservation land is lower than the quality of state-owned conservation lands, and, strikingly, almost indistinguishable from the distribution of unconserved undeveloped land in the state. The estimated development pressure facing private conservation lands is also similar to the pressure on other undeveloped land. Only the top 40% of private conservation is similar in quality to public conservation lands. Environmental quality varies widely, so considering a policy change's value requires considering the particular quality levels of land affected. Private conservation lands do comply with the easement's restrictions on development, with pre-2006 easements showing a near-zero change in share of land developed between 2006 and 2016.

This paper also offers a model exploring several reasons why the quality of land selecting into a private conservation program might change as subsidy rates change. Using a simple two-period model of the easement donation decision, I model the donation decision as a product of the landowner's expected revenues from developing their land, the tax benefits of an easement, and their personal utility of keeping their land undeveloped. Landowners with higher utility from undeveloped land are more willing to donate, and landowners with higher development values on their land will be more elastic in response to the price of donation. The impact of a conservation price change on marginal quality is theoretically ambiguous. If the utility of nondevelopment correlates with environmental value, such as if landowners enjoy healthy habitats more than ecologically degraded ones, the lands donated when subsidies are large may be less valuable than those donated when subsidies are small. Conversely, if higher environmental value lands tend to have higher development values, policymakers may need higher subsidies to attract land with higher environmental value.

My model is unique in focusing on the donors' decision above the land trusts'. Past theoretical work has noted that land trusts are likely to lower their standards as donation incentives increase: Vercammen 2019's theoretical work on easements hypothesizes that land trusts that purchase easements may create marginal quality shifts by changing their decision to purchase or donate easements, and Suter, Sahan, and Lynne 2014 finds evidence suggesting land trusts spend less on targeting priority areas when subsidies increase. Virginia land trusts seem to do only marginal screening in my sample: all large land trusts accept donations with a wide quality range, and controlling for land trust fixed effects in my regression does not change the results. This means that the quality of land under easement largely depends on which landowners are willing to donate. The 2002 policy shift that changes incentives for donors rather than for land trusts shows significant effects on those landowner's marginal decisions.

I further find that in the Virginia setting, the marginal easements attracted by a tax incentive change are lower quality than the always-donated easements. For each donated easement, I estimate the landowner's post-tax "donation price," the share of a landowner's donated development value for which the tax incentives will not compensate them. I then examine the effects of a 2002 reform to the conservation easement program that differentially shifted the donation price by development value and donor income, increasing support for easements on high development value parcels and reducing support for low development value parcels. I run a regression interacting the expected impact of the 2002 reform on donation price and whether the parcel was donated pre- or post-reform. This shows that the environmental value rankings of parcels that became cheaper for landowners to donate fell relative to parcels with higher donation prices, indicating that the marginal parcels are of lower quality. The lower marginal agricultural scores can largely be explained by the inverse correlation between conservation and development value: high development value agricultural parcels, which are more elastic to shifts in the subsidy rate, tend to have lower conservation value.

Past research has found that the total acreage of private conservation donations responds strongly to the post-tax price of donating (Soppelsa 2017), with Parker and Thurman 2018 estimating an acreage elasticity of -2.4 to -6.1. I find a slightly smaller but similar acreage elasticity of -1.8, and I add a new quality elasticity to complement it. By allowing the marginal quality of land to shift as tax incentives change, this paper lowers our estimation of the environmental benefits of a tax incentive increase compared to the pure-acreage estimation. A modeling exercise shows that a constant quality model of Virginia private conservation land would have overestimated the increase in very high-quality acreage from large donations by 75% compared to a model allowing variation in marginal quality.

This shift adds a new angle to the literature on selection into private conservation programs. Research on such programs often finds adverse selection in terms of additionality, the behavior change induced by a program (Jack and Jayachandran 2019; Alix-Garcia, Shapiro, and K. R. E. Sims 2012), which results in low estimated overall additionality of many voluntary conservation programs (Braza 2017; Honey-Rosés, Baylis, and Ramírez 2011). This additionality problem usually becomes less severe as programs expand and attract more additional land. Conversely, I find that the selection problem in terms of environmental quality may worsen as more landowners join. This effect lowers the optimal subsidy rate for private conservation as policymakers use the subsidy rate to screen out undesirable donations.

If policymakers wish to expand private conservation incentives to meet the 30x30 acreage goals, this result suggests they should consider building in checks against donations of lower-quality land. My model estimates that targeted same-cost policy offering increased subsidies only to high-quality land could have conserved at least ten thousand more acres of high-quality land per year at a cost of 1.61 low- or medium-quality acres per high-quality acre. Given the high variation in per-acre environmental services, many land use planners would consider this a very worthwhile tradeoff.

I also contribute to the literature on optimal tax deductions for charitable donations. With the goal of inducing donations of private funds towards creating social goods, the US and many other developed countries give substantial tax incentives for charitable donations. In 2020, the charitable donation tax deduction cost the IRS an estimated $44.4 billion (Tax Policy Center 2022). Saez 2004's key framework notes that this policy is only treasury efficient if a dollar in reduced tax revenue increases donations by more than a dollar. Many authors have explored the elasticity of donations in response to the post-tax price of donation, finding an elasticity of -1.44 on average (Peloza and Steel 2005), though it varies considerably between types of charities (N. Duquette 2016). I join a newer strand of this literature that explores how effectively these marginal donations translate into social value. Grant and Langpap 2022 find that despite the high donation elasticity of water-focused-charity donations, the lower marginal benefits of charitable donations compared to public spending make donation incentives inefficient at creating water quality improvements. Galle 2015 estimates that fundraising spending is elastic to the tax incentive rate, reducing the social good provided by the increase in donations. This paper shows another way in which the marginal donations attracted by tax subsidies may produce less social value, in this case by attracting lower environmental quality land for preservation. In the optimal taxation framework, this effect lowers the optimal size of subsidies for easements.

This paper proceeds with Section 2 describing the setting and data. Section 3 presents a model for the supply of private conservation land. Section 4 describes some empirical facts comparing private and public conservation land. Section 5 discusses the empirical methodology for the difference-in-difference analysis of the effect of changing the cost of donation, and Section 6 shows the results. Section 7 estimates the effect of same-cost but targeted easement policies. Section 8 presents robustness checks, and Section 9 concludes.

1.2 Data and Setting

1.2.1 Setting

Conservation easements have been part of the conservation landscape in the United States since the 1970s. Conservation easements add permanent restrictions to a land deed, with a land trust such as a government agency or nonprofit holding the development rights that the landowner relinquishes. Conservation easements let landowners continue current land uses such as agriculture or keeping a residence that are compatible with the environmental value to be protected, but prevent further development by any current or future landowner. Earlier common law allowed landowners to sell partial land rights allowing someone to build on, mine, or pass through their land, but the courts rarely upheld contracts where a landowner sold partial rights to prevent an action instead. The growing environmental movement of the 1970s, looking for new tools to protect vulnerable land, pushed successfully for states to pass laws making easements for non-development legally enforceable. These laws gave land trusts the responsibility for enforcing the terms of easements that they accepted. If the easement terms were violated, the land trust could take the landowner to court to correct the violation.

Easement policy then took a further step from legalization to subsidization. In 1976, the federal government made conservation easements deductible as charitable donations (Parker and Thurman 2019). Federal legislation set the amount deductible as the development value forgone by putting the land under easement. Since then, fourteen states have also established tax incentive programs for private conservation, most of which were introduced in the 1990s and 2000s (Land Trust Alliance 2019).

Virginia's conservation easement incentive program is particularly large. The Virginia Land Conservation Incentives Act of 1999[1] allowed taxpayers to claim tax credits worth up to 50% of the fair market value of an easement donation. Taxpayers could use these tax credits to reduce their tax liability one-for-one, claiming up to $75,000 in credits per year, and they could spread use of these credits over the decade after the initial donation.

However, these tax credits could not be used to reduce liability below zero, and any credits not used within ten years would expire. Donors with high development value parcels or small incomes might be able to claim only a fraction of the tax benefits that the development value of the land made them eligible for. Policymakers were concerned that this discouraged large donations and disadvantaged low-income donors, so in July 2002 the Virginia house passed Bill 1322 raising the cap on annual credit usage to $100,000 and specifying that all easements made after January 1st 2002 would be allowed to transfer tax credits to other taxpayers. This sales process requires paying a 2% transfer fee to the Department of Conservation, and a wave of private brokers entered the market to help facilitate matching buyers and sellers of credits for a further fee. At present, at least 6 brokers searchable online handle these transactions. Sale values range, but donors typically receive 70 to 80 cents per dollar of credit after discounting and fees.[2] At the same time, Bill 1322 reduced the claimable credits for a parcel from 50% to 40% of the development value. This meant that donors with small

[1] https://law.lis.virginia.gov/vacodefull/title58.1/chapter3/article20.1/

[2] Land Conservation Assistance Network, https://www.landcan.org/article/Buying-and-Selling-Virginia-Tax-Credits/2545

easements and high incomes, who could use all of their credits even without the sale option, faced a higher price of conservation after the reform.

The Virginia private conservation program also notably includes some provisions to check against fraud. Nationally, fraud is a serious concern in conservation easements. Unscrupulous easement donors hire land assessors to dramatically inflate the expected development value of the land, allowing the donor to claim tax benefits that may be far larger even than the fair purchase price of the parcel. This is particularly common in syndicated easements where a group of wealthy individuals with substantial tax liability join together to buy a parcel and get easement tax benefits. In one such transaction, investors purchased a South Carolina golf course for $5.4 million, and then claimed a disproportionate $40 million in tax deductions from placing an easement on the parcel (Elkind 2017). As a result, syndicated conservation easements are on the IRS's "Dirty Dozen" list of frequently abused transactions. Fraud is particularly prevalent in states like Florida, South Carolina, and Georgia, the "'Southeast Triangle' in which federal income tax liabilities mysteriously disappear" (Feld, T. Sims, and Nielson 2022), which created state-level incentive programs but did not improve enforcement. This concentration of fraud is visible in the national credit use statistics: Georgia has 1.5% of private conservation acreage nationally, but Georgia landowners claimed 36% of federal deductions (Looney 2017a).

Experts have suggested several common-sense measures to reduce fraud such as increasing IRS enforcement funding, requiring land trusts to report the development value of easement donations they accept, and establishing panels to evaluate high development value easement donations (Feld, T. Sims, and Nielson 2022). Virginia's state credit program built in some such checks: all easement donations must send in a full application to the Virginia Tax Bureau for approval of development valuations, and donations claiming development values of over $1 million must also apply to the Department of Conservation and Recreation to prove that the easement restrictions will effectively protect at least one environmental value on the land. According to the DCR's annual reports to the Virginia legislature, the DCR requests revisions of easement terms in approximately half of applications. By proactively monitoring valuations and restrictions, Virginia seems to prevent the abuse seen elsewhere. Virginia has 7% of the nation's acres under private conservation, but claims only 4% of deductions, suggesting it is not a center for overinflated land values (Looney 2017a). This makes Virginia a good setting to analyze the effect of shifts in incentives since the amount of tax incentives landowners are legally allowed to claim and the amount they actually claim should closely align. It also makes the state a potential model for other areas considering expanding easement incentives.

Private conservation has come to dominate Virginia's conservation spending strategy. $1.7 billion of Virginia's $1.8 billion spent on land conservation over the last 20 years has gone to easement tax credits (Vogelsong 2021). Since public purchases of conservation land have become so rare, Virginia landowners largely choose between easements and no binding conservation.

1.2.2 Conservation Priority Data

I measure environmental quality of land using the 2007 Virginia Conservation Lands Need Assessment (VaNLA). Commissioned by the Virginia Department of Conservation and

Recreation (DCR), VaNLA was a green infrastructure mapping project designed to rank the conservation value of all the lands in Virginia along several dimensions that are key goals for Virginia's land conservation programs. The maps separately rank land from one to five on ecological integrity, watershed value, recreational value, forest value, and agricultural value. Locations that have a rank of 5 for a metric are assigned an "outstanding" environmental value and are highest priority for conservation, while areas with a 2 have moderate value and 1 has only a general value. Since the VCLNA was created for the state to prioritize purchases of land and to measure progress towards conservation goals, it is a useful measure of the extent to which the private conservation tax breaks have achieved Virginia policymakers' own targets.

The ecological integrity metric is designed to prioritize land that makes the greatest contribution to preserving biodiversity in Virginia. Recognizing that species benefit most from large and connected patches of preserved habitat (Haddad et al. 2015), VaNLA uses satellite-derived land cover data to identify cores of intact habitat and corridors that connect them. The ranking system then uses results from DCR ecologist's past mapping exercises to prioritize cores and corridors that are large and compact, that overlap with the ranges of endangered species, that have diverse terrain likely to host a range of species, and that have wetlands and interior streams.

The DCR's watershed integrity metric highlights locations that make a valuable contribution to water quality in Virginia. Driven by ecologists' models of runoff, the ranking system awards priority points based on proximity to drinking water sources, proximity to rivers and streams, slope steepness and therefore erosion risk, health and biological diversity of nearby waters, and health of terrestrial habitat.

The forest value map denotes land that the state of Virginia believes create the most value as forests, taking into account both tax revenues from wood production and the aesthetic and cultural value of preserving continuous and healthy forested cover. For lands with current forest cover, the forest layer prioritizes areas with high soil productivity and that are part of areas that produce high value timber. It also gives higher rank to areas that incorporate wetland and riparian features and that are part of natural heritage resource areas or continuous forests.

The agricultural value metric looks to protect historic farmland and high productivity agricultural land on the basis that the presence of agricultural land supplies air quality, scenery, agricultural products, and cultural resources. For areas currently used as agricultural land, the agricultural value index was based 80% on the likelihood of prime farmland as calculated from soil type, land cover, and elevation maps and 20% on the presence of culturally significant agricultural resources (usually historic farms) as designated by the Virginia Department of Historical Resources.

The recreational score encourages protecting lands that are useful for hiking, boating, birding, hunting, or other recreational sources, and gives greater weight to areas that are accessible from major population centers. Using maps of recreational locations from the Virginia Department of Game and Inland Fisheries, the index gives 1 point for each recreational use possible in the location.

These measures naturally have some limitations. They do not necessarily include all environmental benefits that the land creates. In addition, these measures also do not translate well to a dollar value, or even offer a clear ratio of benefits across different rankings. However, these limitations are common to most attempts at multidimensional environmental value estimations, and VaNLA includes the major environmental values most important to Virginia policymakers. Creating a clear monetary estimate would require detailed value estimates of traits like biodiversity, which is something that economists are only beginning to quantify. I accommodate this uncertainty by analyzing quality both in individual variables and indices.

As part of the VaNLA exercise, the state also mapped created a model of development risk statewide. This model is designed to highlight locations with the greatest development pressure. Using satellite and census measures, they calculate housing growth and imperviousness growth from 1990 to 2000 to find development "hot spot" census tracts that have seen the most growth. They then calculate development risk as a function of road commuting times to the nearest such hot spot. They reduce their estimated development risk in areas that would be difficult to develop, such as parcels with steep slopes. The model outputs a 1-8 scale of development pressure, where 8 is at greatest risk of development. I use this score separately from the environmental value scores to test the extent to which conservation is targeting land at immediate development risk.

I also use a set of mapped measures discussed in Appendix A.1 to measure other traits of the land such as its value in current use, development risk, local demographics, land use, weather, and soil quality. I explore land use, development risk, and area demographics as outcome variables, and I use those variables and the others as inputs to my development value estimation model.

1.2.3 Easements and Conserved Lands

I locate private conservation lands using the National Conservation Easement Database (NCED). The NCED gathers voluntary data from land trusts on the locations, purposes, and date of establishments of easements. The NCED estimates that their database contains all government-held easements in the state of Virginia and 88% of nonprofit-held easements. Since government organizations hold 82% of all easements in Virginia, this amounts to 98% of total private conservation acres and makes Virginia one of the most complete states in the NCED.

The NCED contains data on 6,816 Virginia land parcels under private conservation. 94% have listed easement establishment dates. Of those, 2,473 easements were established between 1998 and 2006, placing them in my central period for analysis. This period marked a dramatic increase in private conservation following the establishment of the Virginia tax credit program. Before this data, only 1,022 easements had been established in the state, with 20 to 60 easements established a year since the 1970s.

To map the locations of public conservation lands in the state of Virginia, I use the Virginia DCR's Virginia Conservation Lands Database. This database tracks state, federal, and local government preserved lands in the state. These lands include 2.3 million acres of federally owned land, 430 thousand acres of state owned land, and 47 thousand acres of locally owned land. I also make use of the statewide set of tax parcels on the Virginia Department of Transportation's Virginia Parcels map. I use this to create an comparison group of unconserved land, and to test the number and completeness of tax parcel coverage by individual easements.

1.3 Model

Conservation easement policy has a central contradiction. While the policy seeks to correct an externality created when landowners develop land with high environmental quality, the incentives given to landowners are tied to development value, the value of preserving the option to develop. I use a simplified two-period model to illustrate this tension. The policymaker wants to attract parcels for private conservation that have high environmental benefits and would have developed in absence of an easement.

From an economist's perspective the potential solutions to this externality might seem obvious: pay or charge landowners according to the amount of environmental value they protect or destroy. However, several factors prevent this option from being implementable. First, pinning down the exact amount of environmental value a parcel provides is extremely difficult. In the past, there were few good estimates of where environmental value was even concentrated. In recent decades, helped by the growth of remote sensing, many researchers have contributed to a rich literature trying to estimate the value of every environmental service from water management to biodiversity to birdwatching. Nonetheless, these exact value estimates remain contentious and incomplete. Therefore, conservation easement policy has gravitated towards using quality thresholds for eligibility instead: policymakers can choose some objective and attainable measurement or ranking to decide whether or not a parcel meets the criteria for a program. The VaNLA project itself shows why policymakers prefer this method. While the scientists who created it do not have an exact benefit ratio between any two tiers of VaNLA priority, they have confidence that the higher tiers held more environmental value per acre than the lower ones.

While environmental value is hard to measure, our legal system already has a clear structure to measure development value. In most places, assessors already assess land and property values for local taxes. Assessing the development value just requires an additional step of separating the value of the development option from the land's value in its current use. Fraudulent assessments can distort these measures, but active enforcement bodies like the Virginia Department of Tax can detect and correct these overvaluations if they significantly differ from the best practice approach. Using development value as a basis for incentives was a particularly natural policy move because it makes easements legally similar to other charitable donation tax incentives, where a taxpayer receives the same incentive for donating to any nonprofit as long as the nonprofit meets the IRS's criteria for tax exemption.

The policymaker is therefore constrained to an incentive program where the government allows all landowners who meet an environmental quality level to get a tax subsidy that is a share of the land's development value in exchange for donating an easement. This policy format creates several inefficiencies. Attracting land at high risk of development requires higher subsidy rates to balance the benefits of development for landowners. However, higher incentive rates may also attract landowners with lower intrinsic utility from preserving their land. If that utility of conservation correlates with the environmental quality – if owners of high biodiversity forests or historic farms get more utility from conservation than landowners with fragmented habitat strips alongside golf courses or low-productivity plots – increasing incentives will attract lower environmental quality donations. In addition, higher development value parcels will be more elastic to changes in the development value subsidy, potentially leading to lower marginal quality of easements if development and environmental value are inversely correlated. In setting the subsidy rate, the policymaker therefore needs to balance the costs of inframarginal payments and lower quality easements against the benefits of additional acreage and lower development risk land.

To illustrate this challenge, I use a single-period game between a policymaker and landowners. The policymaker moves first. They choose a conservation easement subsidy rate as a share of environmental value and an environmental quality threshold for easement subsidies. The landowner then chooses whether to donate an easement and then, if not restricted by an easement, whether to develop their land. Society then receives benefits from tax revenues and the environmental quality of undeveloped land, while the landowner receives revenues from their land, subsidies from easements, and a warm glow utility if the land remains undeveloped.

1.3.1 The Landowner's Decision

In this model, landowners derive utility from two sources: the income the landowner receives from their developed and undeveloped land, and a warm glow utility that landowners may receive from undeveloped land. In absence of a conservation policy, landowner i's utility function is:

$$U = \begin{cases} a(1-\tau)v_i^d & \text{if developed} \\ a(1-\tau)v_i^u + au_i & \text{if undeveloped} \end{cases} \tag{1.1}$$

where a is their land parcel's acreage, τ is the tax rate on income, u_i is the landowner's private utility from undeveloped land, and v_i^d and v_i^u are the revenue per acre a landowner receives if they develop or do not develop respectively. $V_i \equiv v_i^d - v_i^u$ is the development value of the land. This land also provides an environmental externality e_i per acre if undeveloped and 0 if developed. This value does not directly enter the landowner's utility function.

If the landowner chooses not to develop, they effectively pay a monetary price $a(1-\tau)V_i$ in exchange for au_i in utility of keeping the land undeveloped. This social benefit of conserving will be higher than this monetary price if $e_i > \tau V_i$, so the policymaker may choose to offer some landowners an easement tax incentive. The policymaker chooses a quality floor e^{min} and a subsidy rate s. If a landowner places an easement on a parcel with $e_i > e^{min}$, they receive a tax incentive worth saV_i. The easement binds the landowner to nondevelopment, so the landowner's utility function becomes

$$U = \begin{cases} a(1-\tau)v_i^d & \text{if developed} \\ a(1-\tau)v_i^u + saV_i + au_i & \text{if undeveloped} \end{cases} \tag{1.2}$$

So the landowner, who previously would have left land undeveloped only if $(1-\tau)V_i < u_i$, now keeps land undeveloped (and under easement) so long as $(1-\tau-s)V_i < u_i$. The price of conservation p, defined as the post-tax monetary loss per dollar of development value kept in conservation, lowers from $1-\tau$ to $1-\tau-s$. The landowner donates an easement if:

$$p < p_i(u_i, v_i) \equiv \frac{u_i}{V_i} \tag{1.3}$$

The landowner's maximum price of conservation at which they are willing to conserve is decreasing in u_i: landowners who get greater personal satisfaction from keeping their land undeveloped are willing to pay more to protect it. On the other hand, increases in the development value V_i mean the landowner will demand a lower price to conserve, since they are giving up more development value.

The correlation between e_i and p_i that creates the quality shift seen in my results could come through two potential channels in this model: a relationship between e_i and u_i, or a relationship between e_i and V_i. In the first category, we might expect that higher environmental value is correlated with higher conservation utility for the landowner. A healthier, more environmentally valuable ecosystem may offer more psychological benefits from exposure (Wyles et al. 2019), better opportunities for outdoor recreation like hunting or birdwatching, and perhaps a greater "warm glow" for playing a role in protecting it. As such, these landowners need less of a tax incentive to place their land under easement.

In regards to the second channel, the parcels with the greatest environmental value are often located far from the biggest development opportunities. Development value V_i tends to be highest near urban areas, while the greatest environmental value often comes from preserving contiguous and relatively undisturbed habitats more frequently found in rural regions. Table 1.7 illustrates that in the Virginia context, e_i and V_i are indeed inversely correlated. While development value distributions alone cannot explain the full effect, this likely plays a role in Virginia and can be tested for elsewhere.

1.3.2 Costs and Benefits of Subsidies

Next, I scale up the individual landowners' decision into a supply curve. In this model, landowners vary by their values of u_i, V_i, and e_i. The maximum price p_i at which a landowner will choose to conserve is a function of u_i/V_i, the ratio of a landowner's private utility of conservation to the value of their development rights. The total acres landowners are willing to donate then becomes

$$A(p) = a \int_{p=p}^{1} f(p)dp \tag{1.4}$$

where $f(p)$ is the probability density function of p_i. For simplicity, I set acreage per parcel $a = 1$. $A(p)$ denotes the total number of acres upon which landowners want to place an easement on their land when the conservation price is p, but some share of that acreage may have $e < e^{min}$ and therefore not qualify for the easement subsidy. A share of offered acres $P_{e^n}(p)$ has quality $e_i \geq e^n$ for any value $e = e^n$.

To calculate the full environmental benefits of an easement, I also need to consider the marginal development decision. Some landowners may have $u_i/V_i > 1 - \tau$, meaning they would not choose to develop their land even without an easement subsidy. These never-developers are referred to in the environmental literature as "non-additional:" since they behave the same whether or not they participate in the easement program, their participation does not add environmental value. For these non-additional landowners, $p_i \geq 1 - \tau$: they will always choose to place their land under easement even without a subsidy. Therefore in this model, $A(1 - \tau)$ acres of prospective donations are nonadditional, and $A(p) - A(1 - \tau)$ acres are additional.

This entry of low-additionality easements into the program is consistent with the broader literature on selection into voluntary conservation programs (Jack and Jayachandran 2019; Alix-Garcia, Shapiro, and K. R. E. Sims 2012; Vercammen 2019), where the landowners with the least interest in developing their land are the most willing to join a program. My model creates steeper adverse selection than seen in empirical examples because I do not include dynamic uncertainty around development opportunities or transaction costs. When there is uncertainty around development opportunities, landowners who had low development expectations when they placed an easement may later be forced to decline a good development offer, making their easement additional.

I choose not to include these complexities in this model because additionality is difficult to measure and less relevant in my setting. Unlike most kinds of voluntary conservation, easements lock in land use in perpetuity. They create permanent reserves of protected land that should remain undeveloped no matter how future development pressure evolves. Even examining development effects twenty years later cannot fully tell us the program's impact. In addition, the Virginia setting of this paper only compares across subsidy rates that are always well above zero, so a lack of a selection effect in this range fits with my models predictions. The total amount of environmental value preserved by easements given subsidy rate s would therefore be

$$B(s) = (A(p) - A(1-\tau)) * P_{e^{min}}(p) * E[e|p < p_i < 1-\tau, e_i \geq e^{min}] \tag{1.5}$$

Two factors enter the benefits equations: the additional acres protected $(A(p)-A(1-\tau)) * P_{e^{min}}(p)$, and the average environmental value per additional acre $E[e|p < p_i < 1-\tau, e_i \geq e^{min}]$.

In regards to costs, the policymaker must pay for both additional and nonadditional land, though differently on each. Altogether, the cost is

$$\begin{aligned} C(s) =& A(p) * P_{e^{min}}(p) * sE[V|p_i > p, e_i \geq e^{min}] \\ &+ (A(p) - A(1-\tau)) * P_{e^{min}}(p) * \tau E[V_i|p < p_i < 1-\tau, e_i \geq e^{min}] \end{aligned} \tag{1.6}$$

Easement subsidy payments cost sV_i for each easement acre regardless of additionality. On land that would have developed, the government also loses the τV_i in tax revenue they would have received from development.

1.3.3 Costs and Benefits of Subsidy Shifts

This model can illustrate how differences in marginal environmental quality can impact the costs and benefits of changes in easement tax incentives. If the quality of potential donations declines as the price of donation falls, a naive estimate ignoring quality effects will overstate the benefits of a subsidy rate increase. It will also overestimate the program's costs and total protected acreage if the program has a higher environmental value floor for eligibility.

As subsidy rates change, the shift in benefits is the derivative of Equation 1.5:

$$\begin{aligned} \frac{dB}{ds} =& - A'(p)P_{e^{min}}(p)E[e|p, e^{min}] \\ &- (A(p) - A(1-\tau))P_{e^{min}}(p))\frac{dE[e|p, e^{min}]}{dp} \\ &- (A(p) - A(1-\tau))P'_{e^{min}}(p)E[e|p, e^{min}] \end{aligned} \tag{1.7}$$

The change in benefits is composed of three key terms: the change in the number of acres offered, the change in the share of offered acres meeting the environmental requirement, and the average environmental value. Since the donation price $p = 1 - s - \tau$, an increase in s will cause the price of donation p to decrease. An estimate with fixed marginal quality considers only the first term, the change in the number of acres offered, since a lack of quality changes sets $P'_{e^{min}}(p) = 0$ and $\frac{dE[e|p,e^{min}]}{dp} = 0$. The latter two terms will be negative if marginal quality is declining. First, declining marginal quality means $\frac{dE[e|p,e^{min}]}{dp} > 0$. This lowers the average environmental value preserved per conserved acre as the subsidy for conservation increases, driving down the overall benefits from a subsidy increase.

Second, $P'_{e^{min}}(p) > 0$ reflects that fewer marginal acres meet the policy's required environmental threshold as the price of conservation falls, so more potential donations will be rejected and fewer acres will be conserved than the naive estimate expects. This also slows down the growth of costs in response to a subsidy rate change since fewer marginal acres are eligible for incentives.If the floor e^{min} is low as in current easement policy, few potential easements will be disqualified under any subsidy rate, so this term may be unimportant. However, this will impact evaluation of alternate policies with higher floors for environmental quality because it slows the growth in acreage and costs at any specific subsidy rate. If policymakers hope to reach a specific acreage goal despite declining marginal quality, they will need to set a higher subsidy and pay a higher marginal cost than they would in the fixed-quality case.

1.4 Conservation Land Characteristics

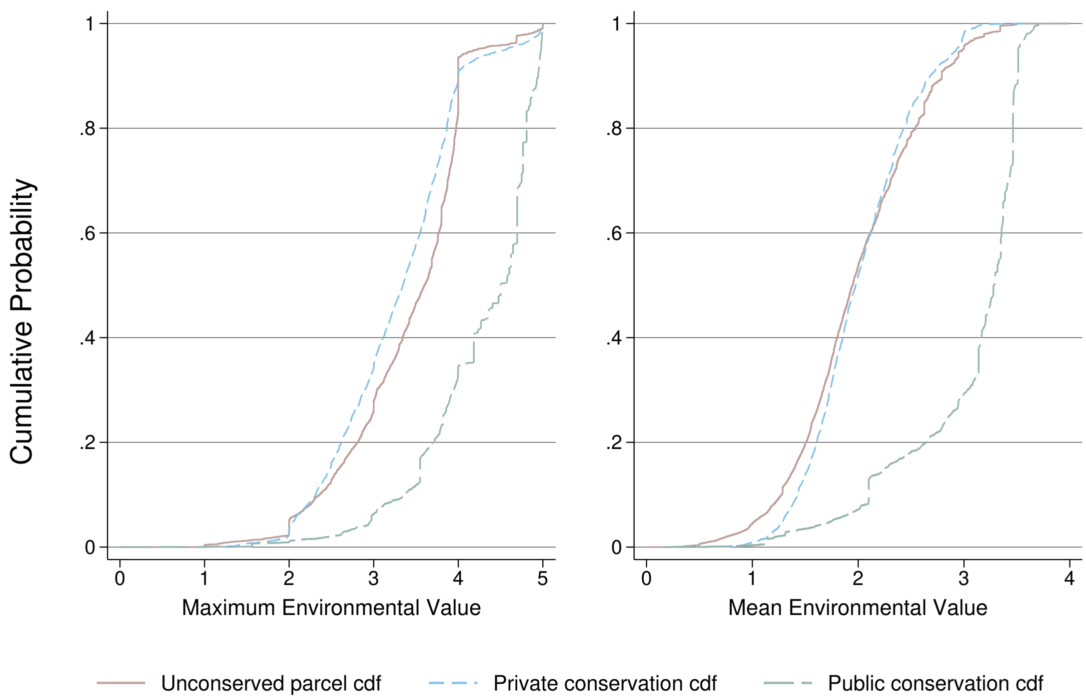
In this section I explore the attributes of conservation land in the state of Virginia, an important step in understanding what value private conservation programs are producing. I compare private conservation land to public conserved land and a random sample of 19,000 private unconserved parcels with at least 33% undeveloped land. Public conservation land clusters around the highest conservation priority scores, as expected. Private conservation land, in contrast, rates poorly. Private conservation land looks similar in quality to the unconserved undeveloped land. It has lower and more variable quality than public conservation land for every dimension except agricultural value. This variability in quality leaves room for differences in marginal quality to be important for valuing the benefits of a policy change.

I also test whether private conservation lands comply to the restrictions on development and find that land use remains fixed after easements are put in place, suggesting a high level of compliance to easement terms. The distribution of development threat level for easements is similar to that of statewide undeveloped land.

1.4.1 Conservation Land Environmental Value

First, I plot a cumulative distribution function of environmental quality among private and public conservation parcels, as well as unconserved undeveloped parcels statewide. Aside from easements, governments otherwise must buy land to preserve it in parks or reserves. Since governments must either directly pay a market price to own land or at least forgo revenues from selling this land to private landowners, owning land is more expensive for the

Figure 1.1: Quality distribution of public and private conservation lands



state than holding easements. However, the government can directly choose which parcels to purchase with more control than the easement process exerts, and land purchases allow stronger control over the land's use.

Figure 1.1 shows the acreage-weighted CDF of the highest environmental value on conserved and unconserved land parcels. State-owned public conservation land encompasses mostly land in the highest VaNLA rankings: two-thirds of public land scores between a 4 (very high conservation value) and a 5 (outstanding value). Only 5% of publicly owned parcels score below a 3 (high value). This suggests that the VaNLA scores do encapsulate the state's conservation priorities well, since the areas that the government actively chose to purchase are clustered at the high end of this scale. It also implies the state places little conservation value on lands below a 3 on this scale.

Strikingly, the quality distribution of private conservation lands is almost indistinguishable from that of a random statewide sample of land. As Figure 1.1 shows, the cdfs for private conservation and unconserved land track closely. By Virginia's metrics, the environmental quality of land preserved by easements looks as though the state had thrown darts at a map, excluding only heavily developed parcels. The DCR-measured development risk in Figure 1.2 is also similar to the state-wide distribution, so easements are not targeting areas with more or less development pressure. Indeed, the private conservation quality distribution is similar to the unconserved distribution for every VaNLA-measured environmental value category. This suggests that the minimum quality checks on easements are doing little more than confirming that parcels have natural or agricultural land to conserve.

This private conservation land quality is on average one priority point below public conservation lands and has a high level of variability. Almost a third of private conservation land has a maximum score below a 3, which is rarely seen in public conservation land. This is not inherently non-optimal: private conservation is less costly for the state than purchasing land for public conservation, so private conservation might optimally cover some lands that would not be worth putting under public conservation. However, this result is far less optimistic for easements than Villamagna, Scott, and Gillespie 2015, which focused on a smaller sample of easements included in a previous, less complete version of the NCED and found that ecosystem services on private conservation lands in Virginia and North Carolina were largely similar in quality to public conservation lands.

Private conservation land also differs from public land in terms of the types of environmental services it provides. As shown in Figure 1.2, public land has the biggest advantage in recreational and biodiversity value. Easement land does better than public land only in terms of agricultural value. This difference shows up in land use as well as land value. Figure 1.3 shows that Virginia has almost no agricultural land in public conservation, but one third of private conservation land is used for agriculture.

This higher level of agricultural land in easements makes sense with the relative advantages of easements compared to purchasing land. Public conservation allows the state to fully control human activity on a parcel, which makes it particularly optimal for purposes like protecting fragile habitats or opening recreation opportunities to the public. On the other hand, in cases where the environmental value can be conserved by restricting only a few behaviors, private conservation can do so without requiring the state to pay for the land's full value. This can perform well for agriculture or water quality, where easement-added restrictions preventing large developments and specifying pollution-reducing agricultural practices can protect key environmental services.

The distributions in Figure 1.2 also draw attention to an important fact shaping private conservation: preserving productive farmland is an explicit major goal of current conservation policy. As written in Virginia's Agricultural and Forestal Districts Act, the state looks to preserve farmland both as an environmental resource that provides positive externalities like clean air, watershed protection, and pleasant views, but also because the state wants to "protect and enhance agricultural and forestal land as a viable segment of the Commonwealth's economy."[3] Virginia commits funding to preserve farmland through multiple channels, from conservation easement subsidies to the $4 million in funding given to Virginia's Farmland Preservation Fund annually through the state budget.[4] This preference is common among policymakers, with many policies citing preserving agriculture as important to preserving rural amenities, protecting cultural heritage, and maintaining food stability (Hellerstein and Nickerson 2002).

1.4.2 Easement Compliance

Second, I test whether private conservation lands actually remain undeveloped. This understudied question is an important first-order condition for private conservation to have any impact: it must be able to prevent development if it is to create environmental value.

[3]Code of Virginia § 15.2-4301, https://law.lis.virginia.gov/vacode/title15.2/chapter43/section15.2-4301/

[4]Virginia state budget for 2023, https://budget.lis.virginia.gov/amendment/2022/1/SB30/Introduced/CA/98/1s/

Figure 1.2: Quality Distribution of Conservation Values

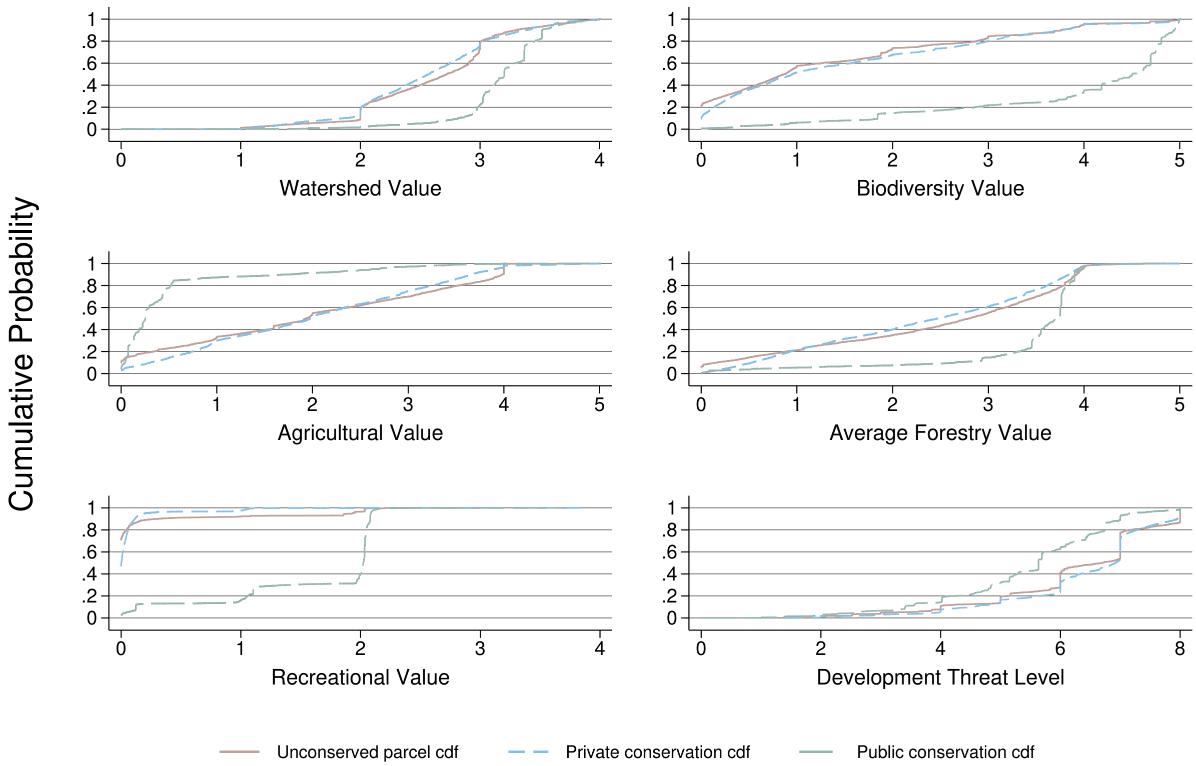


Figure 1.3: Land use of public and private conservation lands in Virginia

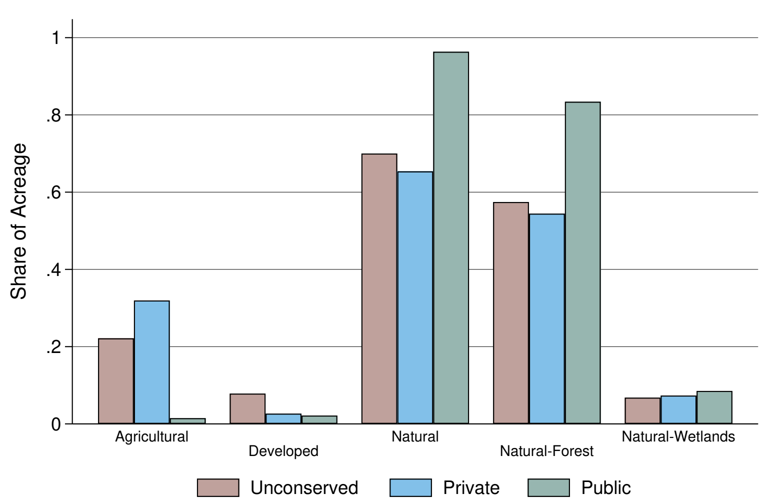
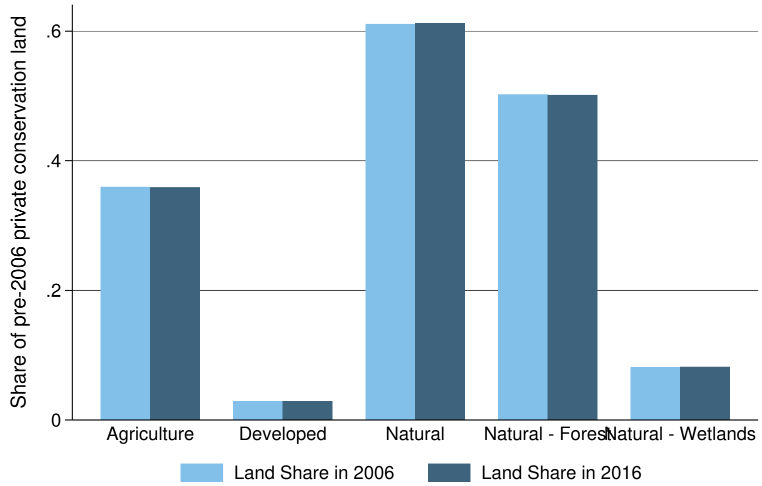


Figure 1.4: Land Use Change on Pre-2006 Easements



This has been a widespread source of concern since the the responsibility for monitoring and enforcement of easement terms falls on the land trust that accepts the donation of the easement, and land trusts range widely in size, mission, and resources (Merenlender et al. 2004).

To test this, I compare land use in 2006 and 2016 on lands that were put into private conservation before 2006. This analysis shown in Figure 1.4 shows that lands in private conservation saw almost no change in land use over this decade. Developed land went from 2.87% to 2.90% of private conservation acreage, a statistically insignificant change. This small fluctuation could easily reflect activities allowed by easement terms, since most easements continue to allow some low-intensity use of the land. Only 2.3% of parcels saw any increase in developed land share, and .8% saw development increase by more than 1%. Natural land acreage slightly increased, while agricultural and forest acreage marginally decreased. This is consistent with the requirements on some agricultural easements that farmers create a natural buffer along any waterways or streams within their property. All of these changes reflect less than a percentage point change in total land use.

This lack of land use change suggests that landowners upheld their easement terms, meeting one of the first-order conditions for this policy to have an impact. Two potential reasons could cause this lack of change: either enforcement kept landowners from developing, or landowners had no interest in development in the first place. There is at least some development pressure on easement land, since Figure 1.2 shows that the development threat levels calculated for easement land match the statewide distribution. However, private conservation land is less likely to be developed for reasons the development threat model cannot consider. Landowners with private information that reduces their interest in development have a good incentive to select into an private conservation program since they can reap the tax benefits while giving up only an option that they considered low-value. In line with this, Braza 2017's matching design study on fields under short term agricultural easements estimated that only 14% of the program land would have been cultivated in absence of an easement. Combining this selection effect with the fact that developed land across the the state of Virginia increased by on 2.6%

in this decade, and the counterfactual level of development that would have happened without private conservation on this land may have been quite low. Most construction during this period replaced or intensified construction on already-developed land, rather than cutting into forest or farmland. However, since easements are permanent, continued enforcement may have a more substantial effect on preventing development in future decades as undeveloped land grows more scarce.

This measure of easement compliance captures the most substantial potential violations of easement terms, but there are more subtle types of violations that the satellite land use measure cannot detect. For example, agricultural easements designed to protect water quality may require or prohibit implementing certain agricultural practices to reduce erosion and pollution. Since these clauses vary between easements and often cannot be detected without on-the-ground monitoring, this paper cannot test for compliance with them.

1.5 Methodology

1.5.1 Difference in difference design

Seeing this variation in environmental quality among conservation lands, I set out to test whether changing easement incentives changes the quality of land donated. For each conservation easement parcel donated between 2000 and 2006, I estimate the development value of the parcel using a model discussed in Section 1.5.3 and the likely income of the donor using census data. With this income and development value information, I create a tax calculator, discussed in Section 1.5.2, to estimate the after-tax price of donating this parcel before and after the 2002 reform. The 2002 reform lowered the price of conservation for large parcels, especially for those with less wealthy donors, and increased the price of conservation for smaller parcels with higher income donors.

I then run the following difference-in-difference regression, omitting the policy transition year of 2002:

$$y_i = \alpha + \beta_1 \Delta p_i * post_i + \beta_2 post_i + \beta_3 \Delta p_i + \epsilon_i \tag{1.8}$$

where y_i is the environmental outcome of interest for easement parcel i. Δp_i is the "treatment effect" of the 2002 reform, the change the 2002 reform would have induced in parcel i's donation price per dollar of development value. $post_i$ is a dummy variable that equals one after 2002 and 0 before. The interaction coefficient β_1 is the coefficient of greatest interest. If β_1 is positive, this means that the average environmental quality of parcels with newly increased prices rose relative to those with decreasing prices. It also shows that as the price of conservation increases, average quality falls. In similar logic to Einav, Finkelstein, and Cullen 2010, this would indicate that the marginal donations influenced by a change in the donation price are lower quality than the average easement parcel.

I run this regression with a range of outcome variables y_i. I use robust standard errors. For the key environmental value regressions, I use the maximum and mean VaNLA environmental quality scores, as well as each of the separate VaNLA scores. I also test the impact on land use and on census tract demographics as measured in the 2000 census. In my main regressions, I use 1999 to 2001 as the before period and 2003 to 2006 as the after period. Easement tax credit policy was stable during each of those periods at both the state and federal level. I omit 2002 as the year of transition: the Virginia legislature began considering the reform in January 2002 and passed it in July 2002, with the new law applying to donations made in 2002 onward.

The difference-in-difference design requires the parallel trends assumption that aside from the subsidy rates, nothing differentially shifted the quality of small parcels relative to large parcels. In my search through records and discussions with Virginia land trusts, I have found no policy shifts that would challenge this assumption. I check for parallel trends using the 1999 period onward. Using 2002, the year the policy was passed, as the base year, I run the regression

$$y_i = \alpha + \sum_{y=1999}^{2006} \beta_1^y \Delta p_i * (year = y)_i + \sum_{y=1999}^{2006} \beta_2^y (year = y)_i + \beta_3 \Delta p_i + \epsilon_i \tag{1.9}$$

I test the treatment effects for each year. I find that the pre-treatment years of 1999 through 2001 are not statistically different from 2002. Years before 1999 are omitted because the Virginia easement tax incentive was first passed in 1999.

To estimate the elasticity of easement supply in response to the price of conservation, I also run a regression on acreage and number of donations in groups of easement value and donor income. I define 16 groups g based on pre-2002 quartiles of pre-2002 price and Δp. For each group and year, I calculate the total acreage and number of donations for each year falling in that bin. I then regress

$$\ln(y_{gt}) = \alpha_g + \beta_1 \ln(p_{gt}) + \sum_{y=1999}^{2006} \beta_2^y (year = y)_t + \epsilon_{gt} \tag{1.10}$$

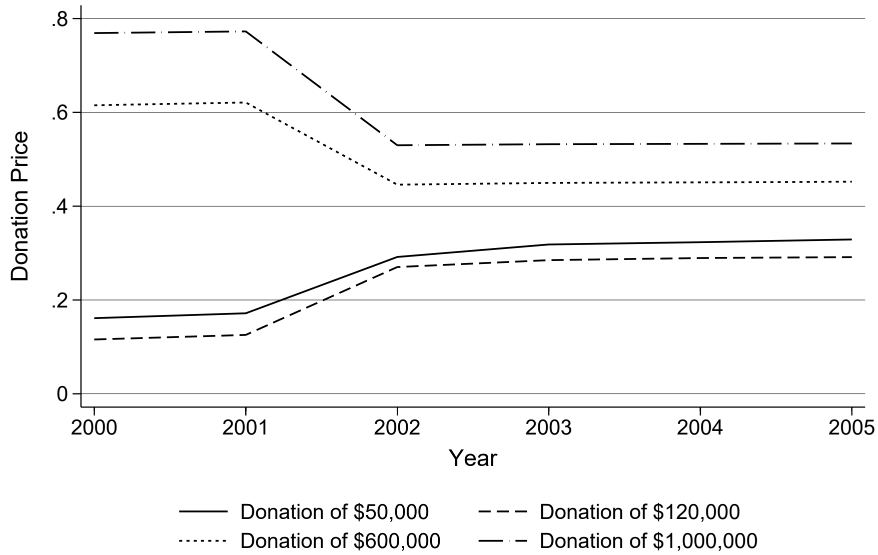
Where p_{gt} is the average expected donation price in the group. The β_1 resulting from this regression is the elasticity of donation numbers or acreage.

1.5.2 Understanding donation price shifts

To measure the effects of the 2002 reform, I estimate the price of conservation for easement donations before and after the reform. Tax benefits for private conservation come from several sources: federal charitable contribution deductions, state tax credit usage, post-2002 tax credit sales, local property tax reductions, capital gains, and estate tax reductions. In this paper, I focus on the first three factors. Most agricultural land in Virginia is covered by a policy that sets local property tax according to the land's current use, so easements rarely change local property taxes in this setting. Property taxes are generally low in the state, with rates between 0.42 and 1.12 dollars per \$100 of valuation. The estate tax and capital gains tax saw no relevant policy change over the 1998 to 2006 window and only 4 of Virginia's 96 counties adjusted their property tax policies during this period (Kulp 2019). The price of conservation p in year t for a parcel with landowner income y and development value V is estimated as

$$p_t(v, y) = 1 - \frac{\sum_{x=t}^{t+15} (1-\delta)^{x-t} \sum_{l=f,s} (tax_l^{nc}(V, y) - tax_l^{c}(V, y) + .8 * sales_s(V, y))}{V} \tag{1.11}$$

Figure 1.5: Tax incentives over time

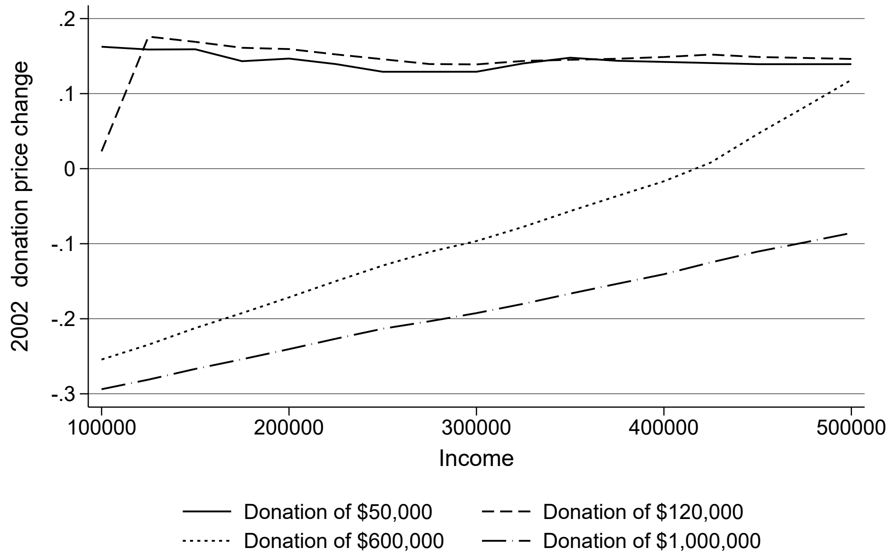


Where tax^{nc} is tax payments without a conservation easement and tax^{c} is payments with an easement, and s is state and f is federal tax payments. δ is an annual discount rate, set at 5% in this calculator. $sales_s$ is revenues from sales of state tax credits. To get the difference in federal taxes, I use federal easement donation law to calculate the size of charitable deduction a donor with development value V could claim in year t, then use the NBER's TaxSim35 tax calculator to estimate the federal taxes a household with income y would owe with and without that charitable deduction. To estimate the state-level tax change, I calculate the household's state tax liability for each year with TaxSim35, then reduce each year's tax liability in turn until the time limit is reached or until the donor hits the cap of claimable credits. Post-2002, I assume that the donor uses credits to first reduce their own tax liability, and then sells all remaining credits for 80 cents on the dollar.

Pictured in Figure 1.5, the results of these calculations show that the 2002 reform dramatically compressed the conservation price for different parcels. The reform allowed taxpayers to sell tax credits if they exceeded what the landowner could use on their own tax liability. Before 2002, a donor with an income of $200,000 would lose only 18 cents on the dollar of development value if they donated a parcel valued at $120,000, but a full 78 cents on the dollar from a million-dollar parcel. After the 2002 reform, the million-dollar parcel's donation price fell by 22 cents as the taxpayer sells the credits they cannot use, and the $120,000 parcel's price rises by 10 cents as the reduction in the credit cap takes effect. This change is large in absolute terms: the donor of the million-dollar parcel in this example would get almost $300,000 more in tax benefits from donating after the reform, while the $120,000 donor would get $12,000 less. The remaining gap in prices is due to federal tax policy, which had caps on the share of income that could be reduced with easement credits and a limit similar to Virginia's on years of redemption eligibility until a 2006 reform.

Focusing in on the 2002 reforms, Figure 1.6 shows how the effect of the reform varied by donor income across different donation sizes. The y axis depicts the difference between 2001

Figure 1.6: Variation in 2002 tax incentive shifts



and 2002 subsidy rates for donations of different donor incomes and donor size. The reform's effect was largest for low-income donors, for whom the past limits had been most binding. Looking at this graph, we can see that there are some parcel sizes above which donors of almost all likely incomes would have benefited, and some below which almost all easement donors would have faced a higher price.

1.5.3 Measuring Income and Development Value

To calculate the subsidy rate for each parcel, I need to use an estimated landowner income and parcel development value for each easement. The exact income and development value claimed for each donation is confidential tax data. Instead, I estimate both using the following procedures, and I include robustness checks for a range of different model assumptions.

Since the median easement donor in 2005 had a household income of approximately $200,000 (Wilson 2005), 4 times the median annual household income, I estimate the likely donor income for each parcel as 4 times the median income for the parcel's census tract. The estimated donor incomes for my sample thus range from $37,500 to $792,600. I vary this assumption in my robustness checks, testing the results if I assume 2 times the annual income, 6 times the annual income, or a constant annual income of $200,000.

Next, I estimate the parcel's development value. Virginia easement law defines the development value of land as "the reduction in the fair market value of the land that results from the inability of the owner of the fee to use such property for uses terminated by the easement."[5] I decompose the development value V_i for a plot as

$$V_i = salevalue_i - usevalue_i \tag{1.12}$$

[5] Code of Virginia § 10.1-1011 B

where $salevalue_i$ is the fair market value of a parcel unencumbered by an easement, and $usevalue_i$ is the fair market value of a parcel of land bound to its present use by an easement. V_i is then the market value of the ability to change the land's use. I estimate the sale value of my easement sample using a hedonic regression of sales value on a dataset of land that is not affected by easements. Since there are too few sales of parcels under easement to estimate a similar regression for use value, my land use values instead come from the estimates of land's value per acre in agricultural or forestal use from Virginia's Use-Value Assessment Program, which creates these estimates for use in local property tax assessment. I discuss the details of the estimation methodology in Appendices A.2 and A.3.

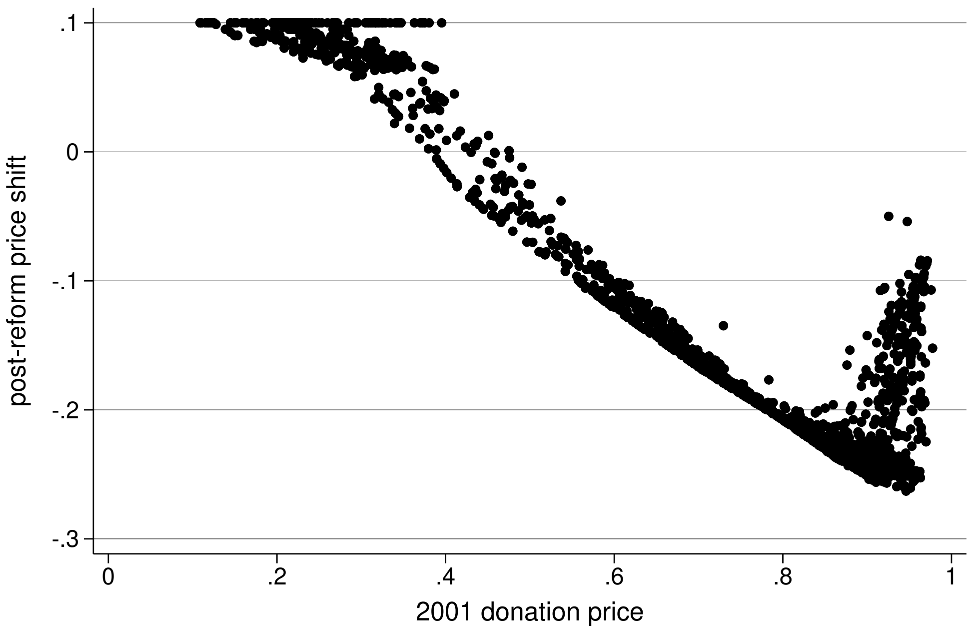
I use a machine learning lasso regression to estimate the sales value per acre, resulting in a model that produces an out-of-sample R^2 of .21. To lower prediction error, the lasso model shrinks the size of coefficients. The combination of this coefficient shrinking and the difficulty in predicting exact parcel value mean that my results are a lower bound on the size of the program's effect.

Entering these estimated incomes and development values into my tax calculator, I return an estimated pre-2002 and post-2002 price of conservation for each easement parcel in my sample. The difference between pre- and post-reform price is the price shift used as a treatment effect in my regressions. Despite the compresion of development values and incomes in the estimation procedure, the estimated donation prices displayed in Figure 7 still show considerable heterogeneity in the pre-reform price of conservation and the price shift. The 2002 reform compressed subsidy rates towards a 40% price, decreasing the price of conservation for most parcels in the sample. The greatest possible increase in price was 10%, capped by the 10 percentage point decrease in the maximum claimable credits. On the other end of the spectrum, some of the highest development value parcels saw smaller price decreases than other parcels with a similar pre-reform price because the post-reform $100,000 per year cap on credits (either to sell or use personally) was still binding. Overall, development value drives most of the variation in pre-reform prices and in the size of the price shift, with income creating the slight variation from the general trend line.

1.6 Results

In this section I measure the impact of the 2002 tax credit reform to test the quality of marginal easement donations. I find that easement donation rates are substantially responsive to the price of conservation. However, these marginal easements differ from the previous average: as the price of conservation falls, the quality of donations declines as well, especially in terms of agricultural priority. In a more neutral shift, the marginal easements tend to be more oriented towards preserving natural land and less towards agriculture. Marginal easements are more likely to be in lower-income areas, though they remain in areas with low levels of racial diversity. The decline in agricultural quality can be largely explained by a correlation between high development value land and low agricultural value. Controlling for land trust fixed effects, on the other hand, has almost no impact on the entry of low-quality lands into easements.

Figure 1.7: The Sample Distribution of Price Shifts



1.6.1 Elasticity of Conservation Easements

I first use Equation 1.10 to test the elasticity of easement donation numbers and acreage to changes in the donation price. The results in Table 1.1 show that around the 2002 reform, donation numbers and acreage both responded to the donation price change. Groups with lower donation prices increased donations in comparison to groups with price increases. The donation count elasticity is -1.79 and statistically significant, while the acreage elasticity of -.83 is not. The acreage elasticity is likely less reliable since the groups for this regression are defined by development value and acreage is a strong predictor of development value. The donation count elasticity is in the same range as some other estimates in the literature, such as Parker and Thurman 2018's excellent estimates of the national acreage elasticity ranging from -2.0 to -5.1. This supports that my estimations are able to identify a true differential shift in conservation incentives, enough to meaningfully impact the number of marginal donations.

This reform's impact on easement numbers needed several years to take full effect. Figure 1.8 shows the number of easements in each price shift quantile over time. The shares of easements in each quantile stayed fairly constant between 2002 and 2003 aside from a small decrease in the share of donations from the group for whom the donation price increased by 10 percentage points. The real change in size trends happened in 2004, as the numbers of parcels in the large price decrease group almost doubled and the number of parcels in the price increase quantiles dipped. This response time makes sense given the slow processes involved in an easement donation. According to internal documents from the Virginia Outdoors Foundation (VOF), the largest land trust in Virginia, putting a parcel under easement with the VOF requires a ten-step process with multiple site visits, legal documents, and board and

Table 1.1: DiD Effect on Easement Donations

VARIABLES	(1) Donation Count (log)	(2) Acres Donated (log)
Price (log)	-1.790***	-0.829
	(0.359)	(0.584)
Constant	7.499***	10.77***
	(1.057)	(1.730)
Year Fixed Effects	Yes	Yes
Group Fixed Effects	Yes	Yes
Observations	80	80
R-squared	0.710	0.770

Robust standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.1

committee meetings. This process requires at least six months to complete, but it may take far longer if donors are not rushing the process or if complications arise. 2003 is therefore the first year where we might expect to see any impacts on private conservation. Delays in informing potential donors about the new program and the time it takes for a donor to decide whether they want to make such a permanently binding commitment with their land mean it is unsurprising that the full treatment effect did not manifest until several years later.

Figure 1.8 also suggests that the 2002 reform led to a long-term increase in the easement supply. Total numbers declined in 2003, since it is a much quicker process to cancel a potential donation than to form a new one. However, the median pre-reform donation faced a 10 percentage point lower cost of donations post-reform, so this reform largely encouraged donations on net. By 2006, we see a doubling in annual donation counts over the 2002 numbers.

1.7 Environmental Quality Impacts

Where conservation prices fell, the average conservation priority scores declined. Column 1 of the Table 1.2 results of estimating Equation 1.8 for the conservation priority measures show that easements with larger price increases had differentially higher mean conservation priority post-reform. The treatment size of 0.6 is almost the size of the pre-treatment difference by price shift, -.755, which suggests that the reform's compression of donation prices also compressed quality across the development value distribution. The effect on maximum conservation priority similarly shows that higher prices reduce quality and the 2002 reform smoothed the previous distribution, though this coefficient is not statistically signficant.

As well as decreasing the average quality, lowering the price of conservation decreased the share of easements that meet any given quality threshold. Using a logit regression to predict the odds that the maximum priority score of a parcel meets a given threshold, Table 1.3 shows that the odds of meeting the "very high" quality threshold of 4 falls most starkly

Figure 1.8: Conservation Donation Counts by Price Shift Quantile

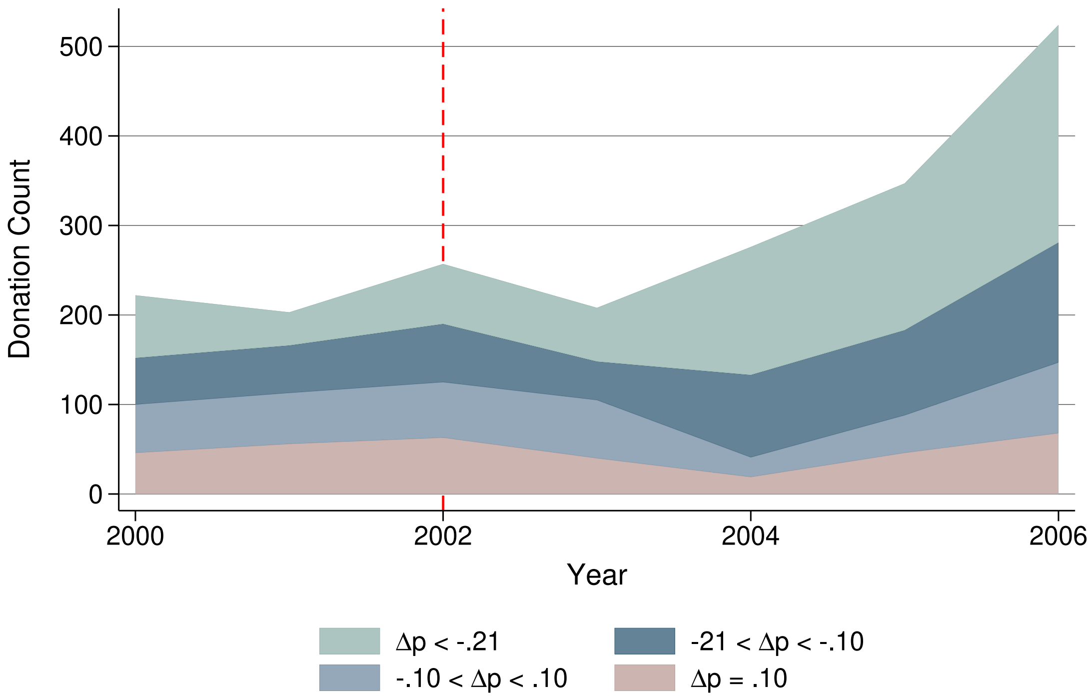


Table 1.2: Effect of Conservation Price on Conservation Value

VARIABLES	(1) Mean Priority	(2) Maximum Priority	(3) Biodiversity Priority	(4) Watershed Priority	(5) Agricultural Priority	(6) Forestry Priority	(7) Recreational Priority
Post * Price Shift	0.632***	0.489	-0.314	-0.0705	1.605***	-0.985*	0.0213
	(0.240)	(0.344)	(0.540)	(0.258)	(0.549)	(0.548)	(0.0663)
Post Reform	0.0499	0.0729	0.348***	0.144***	-0.0842	0.192**	0.00715
	(0.0408)	(0.0572)	(0.0897)	(0.0431)	(0.0907)	(0.0869)	(0.0122)
Price Shift	-0.755***	-0.359	1.222***	0.324	-2.249***	1.348***	0.149***
	(0.202)	(0.292)	(0.436)	(0.217)	(0.466)	(0.470)	(0.0506)
Constant	1.851***	3.160***	0.840***	2.295***	2.022***	1.900***	0.0688***
	(0.0325)	(0.0464)	(0.0681)	(0.0346)	(0.0724)	(0.0696)	(0.00852)
Observations	1,767	1,780	1,780	1,780	1,780	1,780	1,780
R-squared	0.009	0.002	0.019	0.011	0.017	0.011	0.022

Robust standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.1

Table 1.3: Effect of Conservation Price on Maximum Priority

VARIABLES	(1) Max Priority ≥ 2	(2) Max Priority ≥ 3	(3) Max Priority ≥ 4
Post * Price Shift	3.441*	0.432	2.980***
	(2.006)	(0.871)	(1.086)
Post Reform	0.450*	0.0136	0.294*
	(0.270)	(0.138)	(0.178)
Price Shift	-4.898***	-0.107	-0.820
	(1.684)	(0.743)	(0.935)
Constant	2.537***	0.528***	-1.731***
	(0.199)	(0.110)	(0.149)
Observations	1,780	1,780	1,780

Robust standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.1

as quality falls. Considering that 72% of public lands in Virginia have a maximum score of 4 or above, failing to attract quality 4 parcels with subsidy increases fed the gap between private and public conservation quality in Virginia.

The marginal easements also shifted towards more ecological and natural land uses and away from agriculture. Of the environmental values shown in Table 1.2, agricultural priority is most strongly correlated with price. Forestry and biodiversity priority are both higher for marginal easements than for the average easements. Table 1.4's land use results show a similar pattern with more agricultural land at higher prices and more forested and natural land when prices fall. The median decrease in prices of 10% would have increased the likely share of total natural land by 2.5% and of forest land specifically by 3.5%, while agricultural land share fell by 2.7%. Again, for most land uses, the post*price shift interaction points the opposite direction as the price shift coefficient. This indicates that making prices smoother across development values also made the average value of easements more similar, suggesting that potential easements with different development values and donor incomes have a similar underlying elasticity and quality distribution.

Given these shifts in marginal easement characteristics, whether policymakers consider the marginal changes an improvement or a reduction in quality depends partially on their weighting of different environmental qualities. While mean conservation scores declined, that decline is strongly concentrated in agriculture. A policymaker who prioritizes protecting forests and biodiversity over preserving Virginia's agricultural heritage might prefer the marginal easements for their higher biodiversity and forestry priority.

I next test the effects of the donation price on the share of easements in areas with different income levels and racial composition. Given that the tax benefits from easements are mostly claimed by higher income and whiter households that have greater landholding wealth and therefore greater capacity to donate easements, it is a serious social justice concern

Table 1.4: Effect of Conservation Price on Land Use

VARIABLES	(1) Development Threat Level	(2) Agricultural Land Share	(3) Developed Land Share	(4) Natural Land Share	(5) Forest Land Share	(6) Wetland Land Share
Post * Price Shift	1.118**	0.272**	-0.0170	-0.255*	-0.354**	0.0956
	(0.435)	(0.127)	(0.0646)	(0.134)	(0.139)	(0.0625)
Post Reform	-0.411***	-0.0615***	-0.00126	0.0627***	0.0102	0.0413***
	(0.0739)	(0.0200)	(0.0119)	(0.0212)	(0.0224)	(0.0107)
Price Shift	-1.584***	-0.754***	0.190***	0.564***	0.466***	0.0565
	(0.302)	(0.112)	(0.0543)	(0.116)	(0.119)	(0.0450)
Constant	6.967***	0.357***	0.0728***	0.570***	0.515***	0.0357***
	(0.0500)	(0.0166)	(0.00960)	(0.0174)	(0.0179)	(0.00684)
Observations	1,780	1,780	1,780	1,780	1,780	1,780
R-squared	0.033	0.056	0.045	0.029	0.011	0.017

Robust standard errors in parentheses
*** $p<0.01$, ** $p<0.05$, * $p<0.1$

if the viewshed and environmental benefits from easements also disproportionately benefit wealthy white households. In regards to income, lowering donation prices does shift private conservation towards lower income census tracts. Table 1.5 shows that lowering the price of conservation increases the average share of households with incomes over $100,000 and reduces the share of households with less than $25,000. However, racial composition does not change. This means that easements tend to stay in whiter areas of Virginia: while the 2000 census estimated that 72% of Virginia is white, easements are in census tracts that are on average 88% white.

1.7.1 Effects over Time

As with the treatment effect on acreage, the treatment effect on average quality also builds over time. Figure 1.9 and Appendix A.4 show the treatment coefficients produced by estimating Equation 1.9. The coefficients across years are not statistically different from one another due to the wide confidence intervals, but both maximum and mean quality coefficients trend upwards from 2003 to 2006. The specific category and land use scores look somewhat different: agricultural priority and land share was notably higher in 2003 and leveled out in later years.

1.7.2 Intensive Margin Shifts

By lifting the expiration problem with credit usage, the 2002 reform greatly lessened the "penalty" for donating a large, high development value easement. In this section, I explore whether that smoothing of donation prices led donors to change the intensive margin of their donations: did they become more likely to donate larger shares of their land, or less likely to split their land into multiple donations made over longer periods of time?

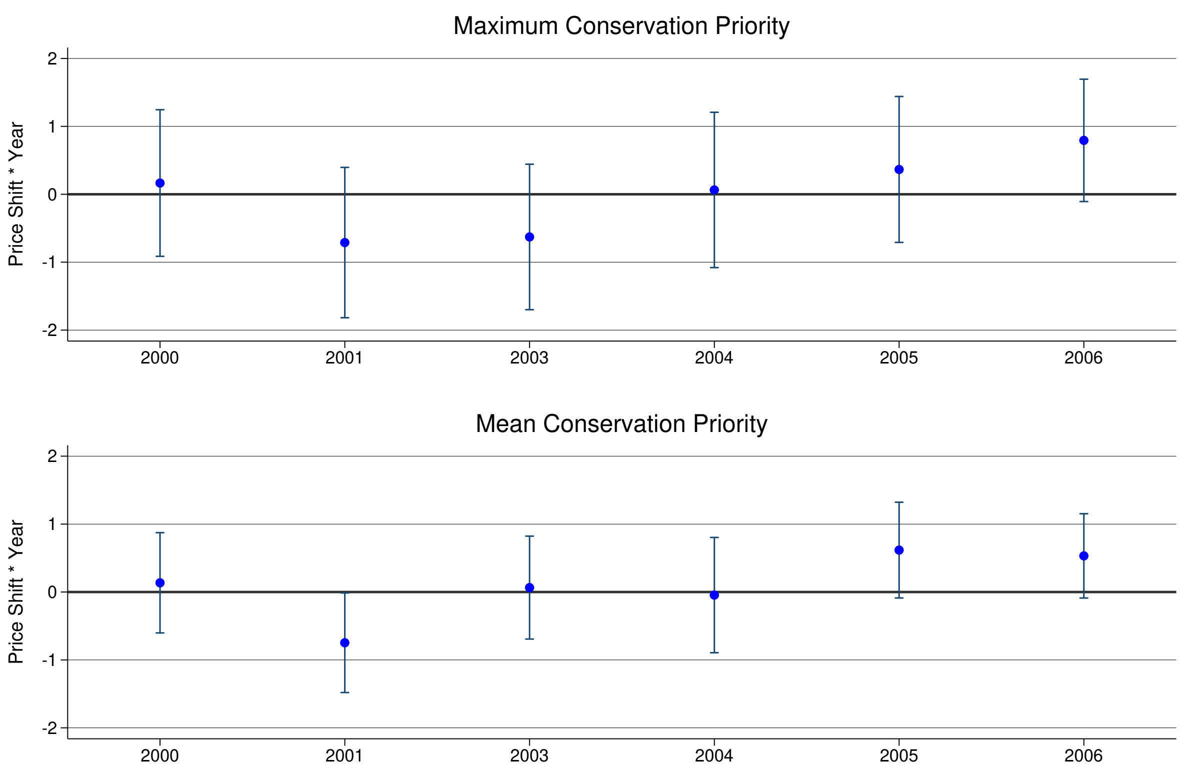
This is both a question of interest in itself and a potential threat to my identification strategy. The intensive margin matters for policy because large, continuous parcels are more

Table 1.5: Effect of Conservation Price on Area Demographics

VARIABLES	(1) Local median income	(2) % of households with income >$100,000	(3) % of households with income< 25,000	(4) % white
Post * Price Shift	17,826	21.43***	-18.47***	0.0203
	(10,879)	(6.341)	(4.167)	(0.0480)
Post	-3,378*	-1.408	2.144***	-0.0200**
	(1,966)	(1.135)	(0.745)	(0.00824)
Price Shift	-2,161	-11.62**	20.34***	-0.0950***
	(9,652)	(5.529)	(3.324)	(0.0363)
Constant	53,755***	17.06***	23.42***	0.883***
	(1,682)	(0.944)	(0.571)	(0.00601)
Observations	1,780	1,780	1,780	1,780
R-squared	0.022	0.024	0.034	0.011

Robust standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.1

Figure 1.9: Event Study Treatment Effect on Environmental Value



Graph denotes coefficients on year * high value, with 2002 omitted as the baseline year of the reform.

valuable for biodiversity, and because breaking donations into multiple small transactions increases the transaction costs for landowners, nonprofits, and the government. It is important for my identification strategy because I assume that donations each have a fixed development value, with landowners deciding only whether to donate or not. If landowners are changing their donation size, my estimate will be biased by the easements that changed development values and thus price shift groups. In an extreme case, we can imagine that the reform did not alter any donor's choice of whether to donate or not, but it did cause the small parcel donors with the highest quality land to increase their donation size enough to be in a lower price increase category. This would result in an overstated positive treatment effect because the same parcel's change reduces the quality of the increased-price group and increases the quality of the decreased-price group.

First, I check how common repeat donations of easements are to see whether donors seem to be managing the caps on credit usage by spreading their donations over time. The NCED easement database lacks information about donors, so I instead use data from the Virginia Outdoor Foundation (VOF) to explore the frequency of repeat donations. VOF is the largest land trust in Virginia, representing more than half of the NCED easements in Virginia, and they provide a dataset including dates and donor names of their easement projects. This data shows that repeat donors do happen, but donors do not seem to be timing their donations around the ten-year credit expiration threshold. Of the 3,926 easements recorded, 80.7% are from unique donors. Another 6% of easements were from donors making multiple donations in the same year, and 75% of repeat donors completed all of their donations within a five-year span. These donations seem to reflect multiple locations that are all part of the same conservation action rather than an attempt to manage tax credit timing. These split donations would incorrectly show up as multiple lower-value donations rather than a single higher-value one in my dataset, and so may reduce my measured elasticities and treatment effect. VOF sees a small and not statistically significant decrease in repeat donors after the 2002 reform. 87.7% of donors between 1998 and 2001 were unique donors, as were 91.4% of 2002-2006 donors. The relatively low rates of repeat donors over time may reflect the transaction costs involved in making a donation. Given that the VOF's easement donation process involves multiple rounds of paperwork, negotiations, and evaluations, landowners may not find it worth the trouble to break a possible donation into separate transactions.

Landowners also can choose whether to cover their entire property or only a subset of it with an easement. I explore this margin by comparing the maps of private conservation parcels to the map of Virginia tax parcels. Tax parcels are an important unit of landownership, used for much official record-keeping. Landowners may own multiple parcels, but each parcel must be owned by a single entity. These parcel maps from Virginia's GIS service correspond reasonably well to conservation parcel areas. With cleaning to remove parcels that marginally overlap due to differences in mapping precision, I am able to match 96% of conservation parcels to at least one tax parcel and 47% of conservation parcels to exactly one tax parcel. Matched tax parcels cover 93% of acreage under easement.

While there are some partial easements, most landowners seem to make conservation decisions as a binary choice over full parcels. To avoid distortions from missingness in the tax parcel dataset, I limit the sample for this analysis to the 92% of conservation parcels for which at least 80% of their acres can be matched to tax parcels. In this group, the median conservation parcel covers 98% of the largest matched tax parcel. I classify 81% of private

Table 1.6: Changes in Tax Parcel Coverage

VARIABLES	(1) % of Parcel Eased	(2) Full-Parcel Easement	(3) Parcels Per Easement
Post * Price Shift	0.112	0.450***	2.501
	(0.0868)	(0.169)	(1.886)
Price Shift	-0.326***	-0.725***	1.054
	(0.0688)	(0.144)	(0.706)
Post Reform	-0.00887	0.0286	1.030***
	(0.0165)	(0.0300)	(0.368)
Constant	0.906***	0.808***	2.184***
	(0.0125)	(0.0248)	(0.116)
Observations	1,482	1,482	1,482
R-squared	0.031	0.028	0.013

Robust standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.1

conservation parcels as full-parcel easements where the conservation area covers at least 90% of the largest matched tax parcel. 78% of conservation parcels matched to a single tax parcel are full-parcel easements. This suggests that most landowners handle easements as a binary decision at the parcel level: they either place an easement on a parcel or they do not. This does leave the possibility that landowners with multiple adjacent parcels are choosing how many of those parcels to include, since 49% of conservation parcels are spread over at least 2 tax parcels, with 12% spread across 5 or more tax parcels.

To test whether this intensive margin seems to change with the reform, I run my diff-in-diff regression on the number of tax parcels per easement and the share of the tax parcel that the easement covers. The changes this regression identifies could come from two potential sources. First, donors could change the size of their donations, such as donating 100% of a parcel post-reform where they would have donated 50% pre-reform. Second, this could come from changes in the composition of donors, such as a landowner who only owns one parcel deciding against donating an easement post-reform while a landowner with four parcels is newly attracted into the program.

The results are presented in Table 1.6, and are more consistent with the change in donor composition story. The number of parcels per easement did increase post-reform, with a significant increase of 1.03 parcels on average. However, overall parcel coverage shares did not change, with a fairly precise null coefficient on Post Reform for both the share of a parcel eased and whether an easement covered a full parcel. The interaction between post-treatment and price shift is positive and significant for whether an easement covers a full parcel, meaning that the coverage share increased for the low-development-value easements that saw higher post-reform donation prices and fell for the high-development-value easements that received larger post-reform subsidies. This is the opposite of what the intensive margin theory would

Table 1.7: Correlations Between Development Value and Environmental Value on Unconserved Parcels

VARIABLES	Max Priority	Mean Priority	Biodiversity Priority	Watershed Priority	Agricultural Priority	Forestry Priority	Recreational Priority
Development Threat	0.00	-0.11***	-0.12***	-0.18***	0.15***	-0.09***	-0.07***
	(0.53)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)
Development Value Per Acre	-0.19***	-0.19***	-0.15***	-0.26***	0.01	-0.15***	-0.05***
	(0.00)	(0.00)	(0.00)	(0.00)	(0.31)	(0.00)	(0.00)
Development Value / Sales Value	-0.04***	0.10***	0.13***	0.02**	0.01	0.15***	-0.03***
	(0.00)	(0.00)	(0.00)	(0.00)	(0.19)	(0.00)	(0.00)
Acreage	0.01	0.07***	0.13***	0.07***	-0.04***	0.07***	0.10***
	(0.24)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)
Development Value	-0.01	-0.02*	-0.03***	-0.02*	0.01*	-0.01	-0.11***
	(0.19)	(0.02)	(0.00)	(0.02)	(0.04)	(0.21)	(0.00)

p-values in parentheses

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

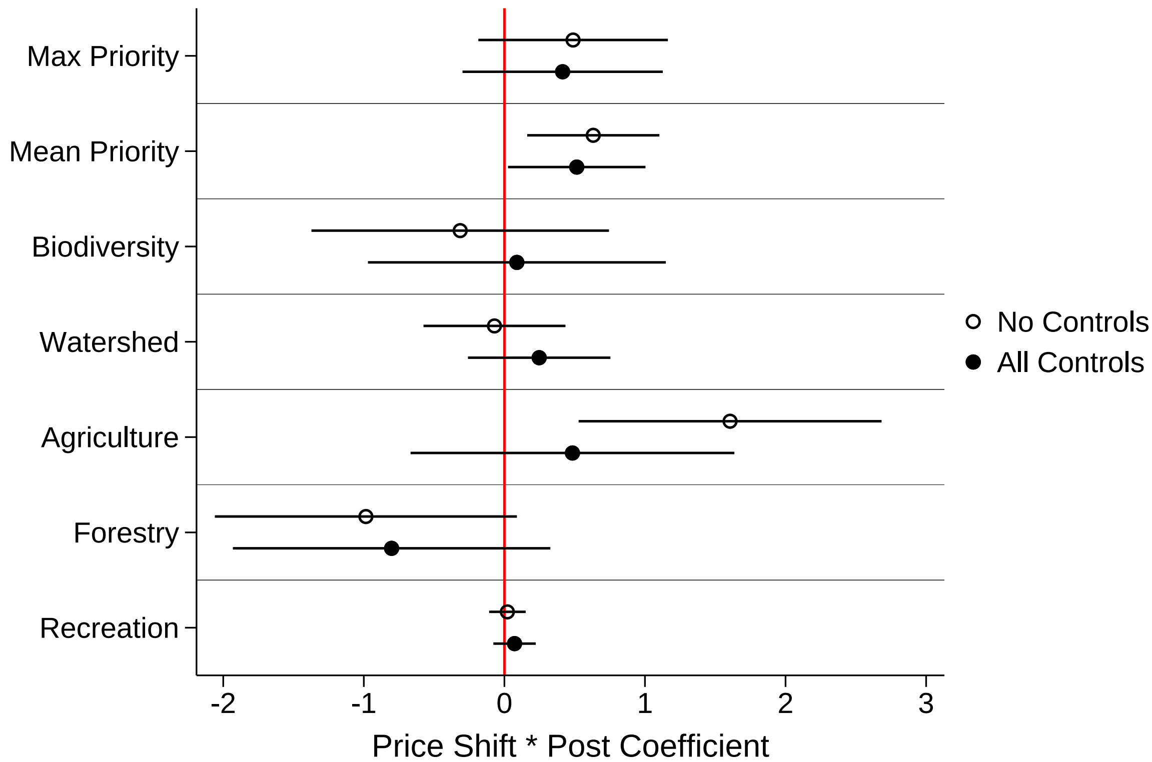
expect: the same landowner, when offered higher subsidies, should want to donate a larger share of their land. This, combined with the fact that we do not see an overall increase in parcel coverage, suggest that the changes in Table 1.6 are due to changes in the kinds of collections of parcels that newly incentivized landowners own, not due to intensive margin changes.

1.7.3 The Role of Development Value Correlations

In this section I explore the possibility that the shift in marginal quality are the result of a link between development values of land and their environmental quality. First, I check whether there is a relationship between development value and environmental value. In Table 1.7, I compute the correlations between a range of development value measures and conservation priority metrics among the sample of nondeveloped noneasement parcels. When we look at measures that explore how significant the development opportunity of land is given a parcel's size or total value, some clear relationships emerge. Parcels that have a higher per-acre development value and development threat level tend to have lower mean and max conservation priorities. Development value as a percentage of total value has a relatively weak negative correlation with conservation priority among easement parcels, and a positive one among nondeveloped parcels. When broken into categories, this negative correlation particularly shows up in biodiversity, watershed, and forestry value. Agricultural priority is the only one that is positively correlated with these development value measures. Together, this suggests that the parcels with lower development cost densities may be quiet rural property with high environmental value where the land can produce decent revenues when left undeveloped.

Including these development values as controls in my difference-in-difference regression shows that this correlation accounts for some of the marginal quality decline in agriculture. Figure 1.10 plots the large * post-reform coefficients with and without including the development controls. For most coefficients, the variation is too large to determine whether the

Figure 1.10: Treatment Coefficients With Development Value Controls



deveopment value controls had an impact. Agriculture is the exception: with development controls included, the size of the treatment effect declines by 75%. This reflects the distribution of agricultural land in Virginia. Agricultural lands closest to cities and particularly in the areas near Washington DC will have the highest development value, but many of the historic and most productive farmlands are in the center of the state and further from these pressures.

1.7.4 The Role of Land Trusts

I next test whether this quality shift is mediated through land trusts. Land trusts have an ongoing responsibility to monitor and steward conservation lands, but different types of land trusts, such as government or NGO land trusts, may face different funding incentives and hold their donations to different standards as a result. Contrary to this, I find lower quality easements are accepted by a wide range of land trusts and land trust types, suggesting that the easement quality problem cannot be solved simply by regulating a handful of low-quality land trusts.

First, I test whether controlling for land trust fixed effects can reduce the treatment effect. If the Post*Price Shift coefficient vanished with the land trust fixed effects, it would indicate that the quality within an individual land trust is fairly constant despite any changes in the price of donating. In that case, we could attribute the decline in quality to a relative increase

Figure 1.11: Adding Fixed Effects for Land Trusts

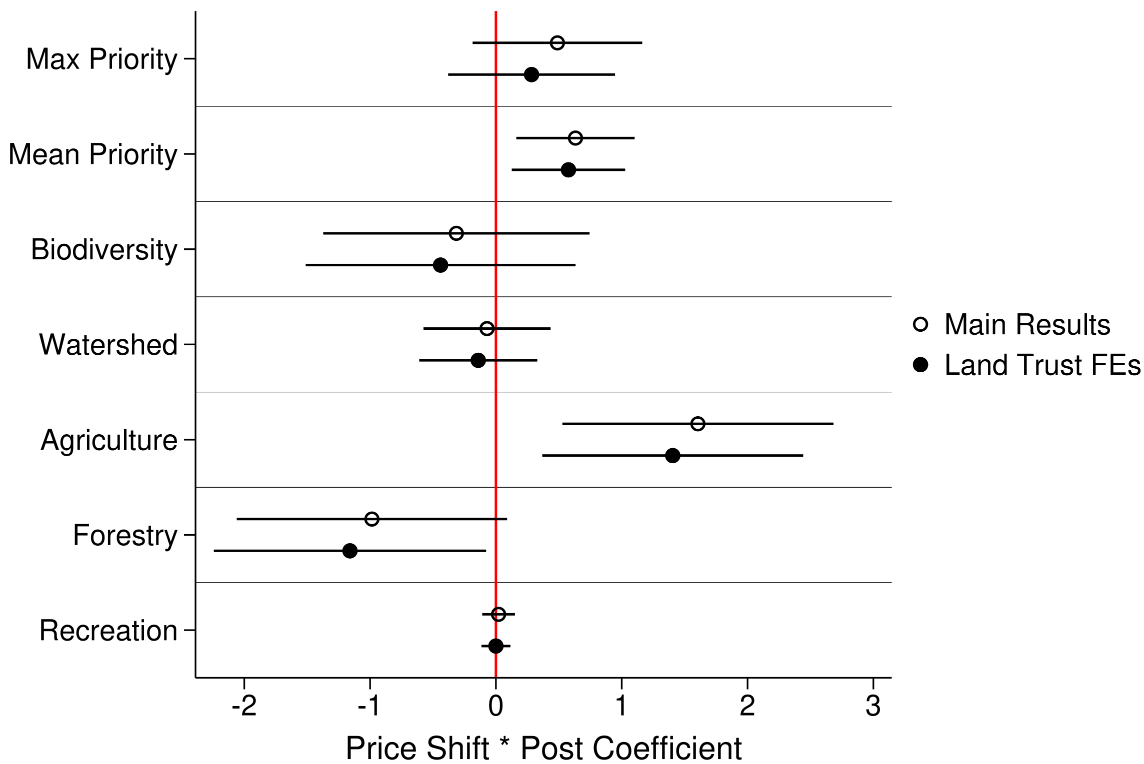
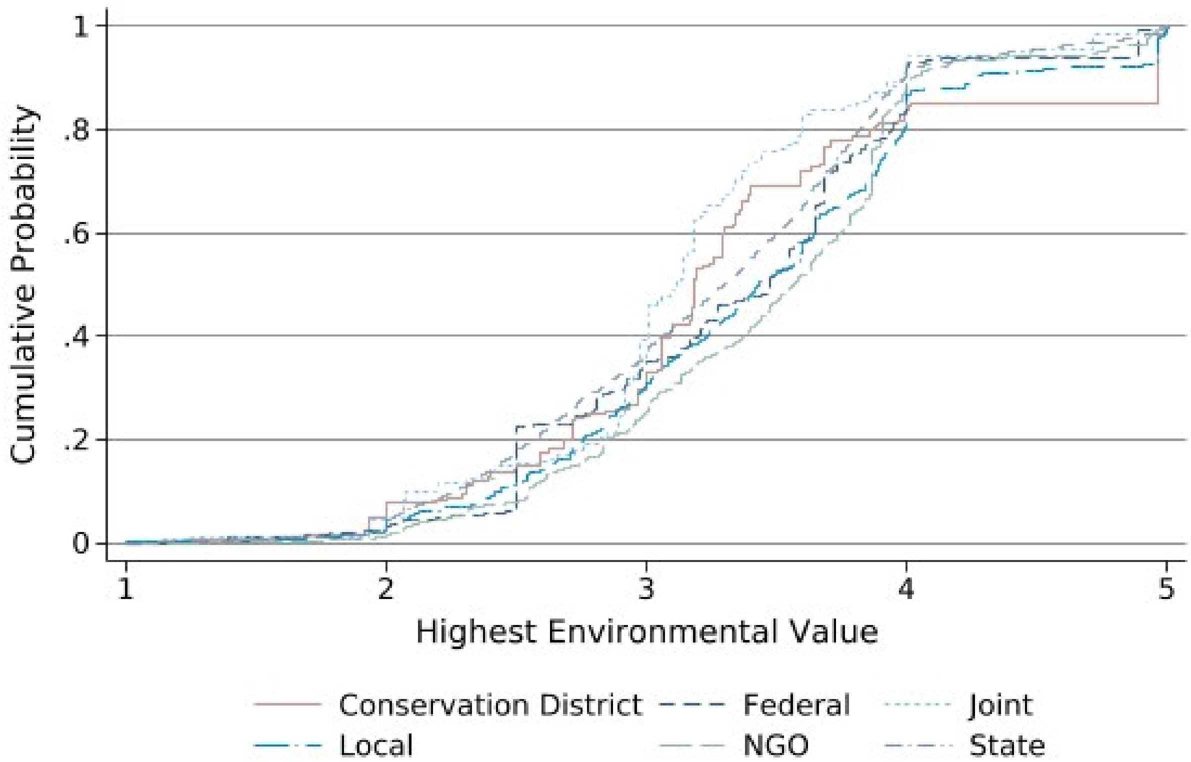


Figure 1.12: Land Quality Distribution by Land Trust Type



in donations to the land trusts with lower standards. The results in Figure 1.11 show that this is not the case. None of the coefficients change significantly with the addition of this full set of fixed controls. Shifts in donation quality within land trusts, rather than between them, must therefore drive the treatment effect.

In addition, no particular kind of land trust performs significantly better than another. Figure 1.12 traces the cdfs of the six types of land trusts with more than 100 easements in Virginia. Federal, state, local government, conservation district, NGO, and jointly held easements all trace similar distributions. Given how well government-owned conservation lands perform, it is perhaps surprising that the government land trusts, which could internalize the full public cost of the easements they accept, do not perform better. This may reflect a gap between the divisions of government that accept donations, often departments of conservation whose goal is to increase protected land, and the tax system that ultimately loses revenue from the credits.

1.8 Model Applications

A key policy question arises from this model and results: if the increase in tax subsidies for easements was targeted more specifically at high conservation priority parcels, how would the cost to taxpayers, conserved acreage, and environmental quality of that acreage have changed? Since the VaNLA rankings do not give us a direct monetary valuation of each rank, I instead look to identify the quality-acreage tradeoffs that same-cost quality-targeted policies could have produced. I particularly focus on the case of high development value parcels in the state of Virginia, which on average saw large increases in tax incentives, to illustrate the potential costs and benefits of quality targeting. To show the potential bias from assuming a constant quality distribution, I also compare estimates with and without differing elasticities by quality.

Using a simple model of easement supply, I calibrate the marginal quality response to tax incentives with my difference-in-difference results, and I estimate the total acreage of donations using my estimated elasticity. With this model I then explore the acreage and quality of land conserved under two policy alternatives that have the same taxpayer cost as the actual policy: a subsidy rate increase available only to parcels with a conservation priority of 3 or above, and a subsidy rate increase available only to parcels with a conservation priority of 4 or above.

To build my supply curve of easements, I need to quantify three dimensions that might respond to changes in easement incentives: acreage donated, environmental quality, and cost per acre. I use a constant elasticity supply curve for total acreage, $p = \alpha q^{1/\epsilon}$. I set the elasticity of total donations as $\epsilon = -1.79$ from Table 1. I calibrate the constant α using the acreage q and average estimated conservation price p for each quartile group of pre-reform conservation price, and I assume a constant acreage per donation and development value of easement in a quartile from the pre-reform mean.

For environmental quality, I use the Table 1.3 logit estimates of $P(e_i > k|s)$, the probability that, given a specific subsidy rate, an offered easement's maximum conservation priority e_i is greater than some floor k. This lets me estimate the share of donated land that will fall in each quality bin, as well as the share of offered land that will not be allowed to donate under the policy floor scenarios. I use the coefficient on Post * Price Shift as an instrument for the post-reform shift in s, and use that to calculate the change in environmental quality under alternative post-reform s.

1.8.1 The Effects of Targeting

Compared to a uniform increase in subsidy rates, how would a same-cost policy offering a higher increase only to easements that meet a certain quality threshold perform? Such a policy could be implemented through channels already used by government funds, similar to a competitive grant-offering system. To focus particularly on the effect of a rate increase, I narrow my analysis to one pool of easements: the quartile with the highest conservation price, and the highest development value parcels. This pool is also of particular interest because it both saw the largest effect of the reform and represents the largest acreage and government expenditures.This group faced an average pre-reform conservation price of $p = .856$ per dollar of development value, and received 13.3 donations per year that conserved 2,300 acres at a total cost of \$2 million. The 2006 reform decreased this group's price of conservation to $p = .638$ on average. The calibrated model predicts that this would have increased donations per year to 22.6 per year, conserving 3,880 acres per year at a cost of \$11 million.

Using the estimated supply curve of easements, I calculate the quality floor policies that would match the \$11 million annual cost while providing an increased subsidy only

Table 1.8: Actual and Alternative Policy Scenarios

	Policy Scenario		
	Actual	Floor 3	Floor 4
Donation Price if Priority < 3	.638	.856	.856
Donation Price if Priority ≥ 3 & < 4	.638	.563	.856
Donation Price if Priority ≥ 4	.638	.563	.149
Estimated Annual Cost for Top Quartile	\$11 million	\$11 million	\$11 million

to easements with a maximum conservation priority score above the "high quality" mark of three (Floor 3) or above the "very high quality" score of four (Floor 4). The resulting donation prices are listed in Table 1.8. The Floor 3 case keeps an $p_l = .856$ donation price for easements with quality below 3, and a $p_{h3} = .563$ price for easements with quality 3 or above. Setting the floor for high-quality easements at 4 allows for even greater subsidies to the most valuable land: in Floor 4, the donation price for high-quality land drops to $p_{h4} = .149$.

Figure 1.13 shows the acreage implications of these alternatives. In the actual policy with no floor, more than a third of private conservation acreage has a maximum conservation quality below 3, a quality level equivalent to the lowest 6% of public conservation lands. Putting in place a floor at 3 shifts the mass of easements from the [2, 3) bin to the [3, 4) bin. The cost in total acreage conserved is low. The state gives up some cheap acres, but the [3, 4) bin is highly responsive to the 5 percentage point increase in incentives over the Actual scenario. Since there is relatively little [4, 5] land and it is positively selected, the Floor 4 scenario must lower p_{h4} dramatically, creating a larger drop in total acreage than in the transition from the Uniform to Floor 3 scenario.

This illustrates the key tradeoff of introducing quality floors as subsidy rates increase. The supply of medium-quality donations is quite elastic, and medium- and low-quality donations become even more common as subsidy rates rise. Setting a quality floor reduces payments on these low-quality lands, but increased spending on high environmental value lands has lower returns in terms of acreage. The policymaker would prefer the Floor 3 policy over the Uniform policy if an acre of quality 3 or above land is worth at least 1.18 acres of land below quality 3. In addition, the policymaker would prefer setting a floor at four instead of three if an acre of quality 4 land is worth at least 1.61 acres of land below quality 4. The literature on environmental valuation suggests the acreage tradeoff for these higher floors is likely to be worthwhile: Phillips and McGee 2016 finds that high-value stretches of riverbank in the Chesapeake Bay watershed provide almost five times the environmental services per acre of low-value areas, and Ingraham and Foster 2008's estimates of environmental services from National Wildlife Refuge system lands in New England ranged from \$60 per acre per year from well-conserved grasslands to \$2,670 from wetlands. Gaining one high-quality acre per 1.2 or 1.6 low-quality acres lost would create a substantial gain even if the true value difference between VaNLA levels is much smaller than the range found in those papers.

The decline in marginal quality heavily shapes the expected benefits from any of these policies. Figure 1.14 shows the acreage conserved by these policies assuming the marginal quality of land $P(e_i \geq k|s)$ is constant, still allowing for time trends in quality distribution. With constant marginal quality, increasing the subsidy rate looks like a very effective policy

Figure 1.13: Private Conservation Acreage Under Alternative Policies

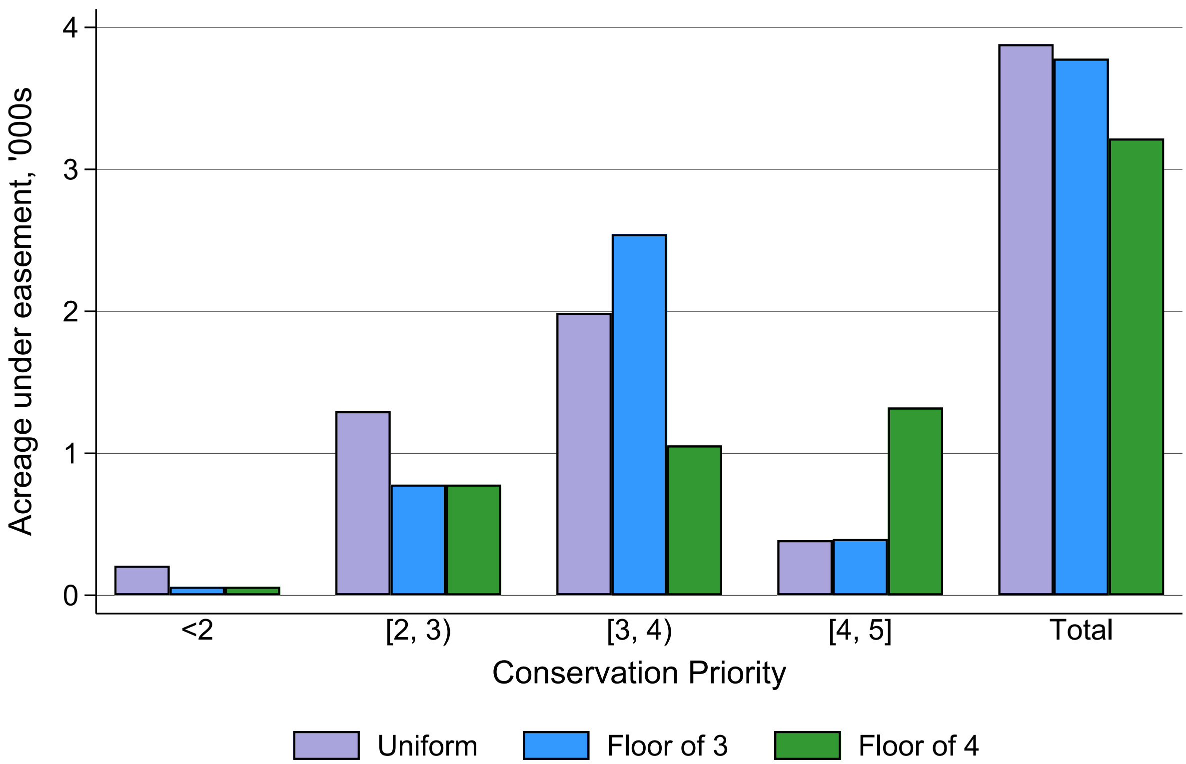
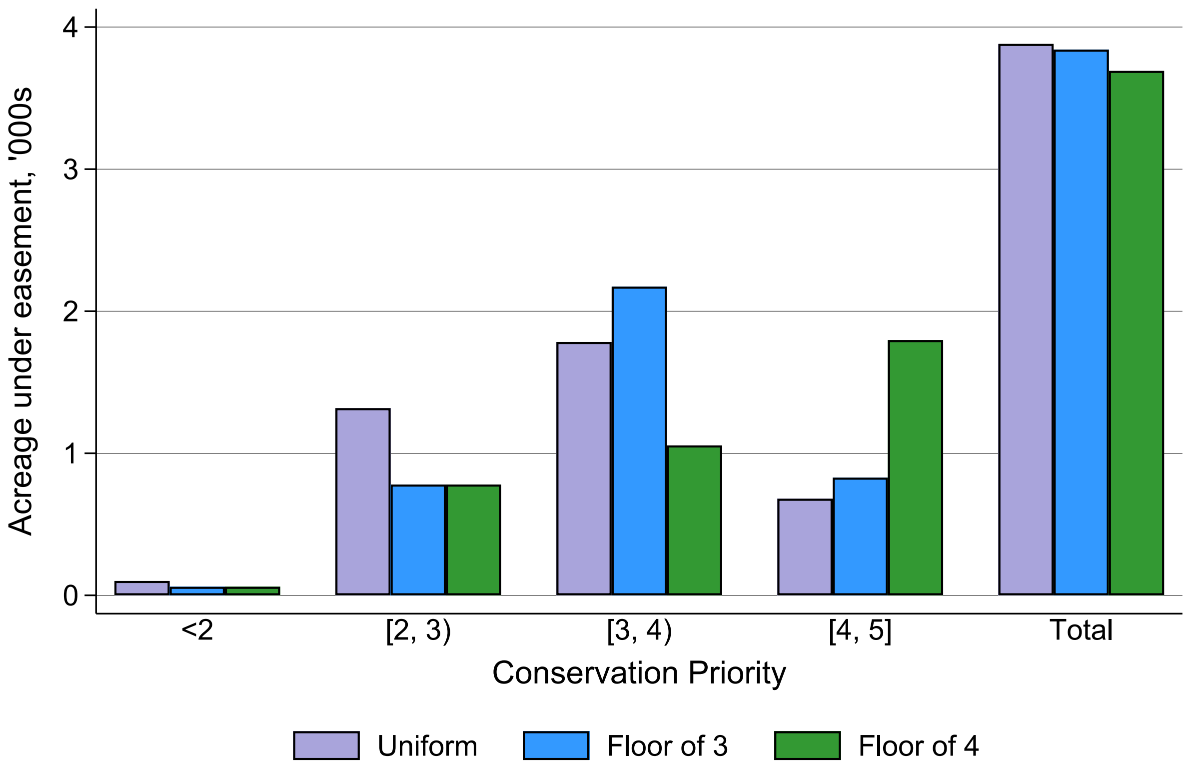


Figure 1.14: Private Conservation Acreage Under Alternative Policies, Price-independent Quality Distribution



for attracting high-quality land. The constant marginal quality model in Figure 1.14 expects that the uniform policy would conserve 175% of the very high-quality land that the model expects when marginal quality is allowed to vary. The constant marginal quality model is also overoptimistic about the benefits of offering targeted subsidy increases: it overestimates the very high-quality land conserved in the Floor 4 case by 35% and overestimates the total land conserved by 15%. The acreage trade-offs involved in increasing quality floors are smaller as well, since the constant marginal quality model ignores the relative inelasticity of the higher-quality land supply. Without marginal quality shifts, the policymaker can attract significant high-quality acreage with only modest increases in the subsidy rate. Together, this suggests that ignoring the marginal quality effect can make policymakers overoptimistic about the benefits of raising conservation easement subsidies. Very high-quality land in Virginia is not very elastic to these policy shifts, but lower-quality land certainly is.

1.9 Robustness checks

This section checks the robustness of the core results to three variations: changing the assumptions about the incomes of easement donors and classifying easements in a binary high- or low-value instead of assigning a continuous price shift effect.

1.9.1 Income Assumptions

In the main results, I calculate the post-tax easement donation prices using estimated donor incomes of four times the census tract median income. Since there is not publicly available data on the exact incomes of Virginia easement donors, this assumption matches the national trend in easement donors, who in 2005 had on average 4 times the median household income nationally. Here, I test the robustness of my results to varying the income estimation procedure: assuming two times the median income, six times the median income, or constant income of four times the statewide median.

The results in Table 1.9 show that varying the income assumption slightly shifts point estimates and in some cases changes significance levels for the quality results, but the core outcomes hold. Mean conservation priority has a significant and positive treatment coefficient for all income levels, and maximum priority also reaches significance at the 10% level for the 2x median income scenario and significance at the 1% level for the constant statewide income scenario. The effect on agricultural priority is consistently large and significant as well. The effects on biodiversity and watershed priority remain ambiguous.

The land use results in Table 1.10 also remain similar. Development threat level's relationship with the price shift post-2002 is still positive and significant in the 6x and 2x median income scenarios, as is the agricultural land share. They do lose their significance in the constant income scenario, though the coefficient remains positive. Since development pressure is higher in areas with higher incomes, assuming constant income statewide reduces the regression's ability to pick up that variation. Forest land share has a negative relationship with price shift * post in all scenarios, and the effect on natural land share remains negative although varies in significance.

1.9.2 Binary Treatment Effect

Given the uncertainties around precise donor incomes and development values, I also test whether my result is robust to comparing only the highest development value easements that almost certainly faced a lower donation price post-2002 to the low-donation value parcels that almost certainly faced a higher post-reform donation price. I define low development value parcels as those with development values below $120,000 and high development value parcels as those with development values above $600,000. The low parcels would have received smaller subsidies post-reform and the high parcels would have received larger subsidies over at least 95% of the national range of land donor incomes. I omit mid-valued easements from the regression.

The environmental quality results in Tables 1.11 are similar in direction and significance to my main results, since high value corresponds to a negative post-reform price shift. The high-value group has lower mean and maximum priority post-reform, particularly in agriculture. The land use results in Table 1.12 do change slightly: without the greater nuance in the main results, the shift away from agricultural land use is no longer detectable. However, Column 6 suggests that the share of wetland under easement may decline as prices fall, which would contribute to a decline in marginal quality given the high biodiversity and water regulation value of wetlands.

Table 1.9: DiD Effect on Environmental Value with Varying Income Assumptions

VARIABLES	(1) Mean Priority	(2) Maximum Priority	(3) Biodiversity Priority	(4) Watershed Priority	(5) Agricultural Priority	(6) Forestry Priority	(7) Recreational Priority
Panel A: Income 6x Median							
Post * Price Shift	0.700***	0.548	-0.287	0.229	1.417**	-0.791	0.0182
	(0.256)	(0.377)	(0.582)	(0.279)	(0.609)	(0.612)	(0.0679)
Post Reform	0.0225	0.0600	0.347***	0.154***	-0.158*	0.238***	0.00750
	(0.0361)	(0.0514)	(0.0781)	(0.0386)	(0.0836)	(0.0796)	(0.0102)
Price Shift	-0.864***	-0.357	0.954**	0.0271	-2.110***	1.176**	0.159***
	(0.221)	(0.330)	(0.484)	(0.241)	(0.530)	(0.537)	(0.0539)
Constant	1.879***	3.174***	0.785***	2.276***	2.114***	1.843***	0.0631***
	(0.0291)	(0.0423)	(0.0594)	(0.0314)	(0.0681)	(0.0646)	(0.00710)
Panel B: Income 2x Median							
Post * Price Shift	0.578**	0.551*	-0.130	-0.321	1.742***	-0.929*	0.0262
	(0.233)	(0.328)	(0.510)	(0.245)	(0.509)	(0.501)	(0.0619)
Post Reform	0.0864*	0.105	0.353***	0.102**	0.0442	0.133	0.00401
	(0.0489)	(0.0676)	(0.109)	(0.0512)	(0.104)	(0.101)	(0.0144)
Price Shift	-0.648***	-0.477*	1.309***	0.518**	-2.280***	1.267***	0.0990**
	(0.192)	(0.272)	(0.397)	(0.203)	(0.418)	(0.419)	(0.0465)
Constant	1.819***	3.124***	0.924***	2.338***	1.883***	1.972***	0.0717***
	(0.0390)	(0.0543)	(0.0831)	(0.0412)	(0.0815)	(0.0812)	(0.0103)
Panel C: Constant Income							
Post * Price Shift	0.748***	0.843**	0.0404	0.219	1.497***	-0.702	0.0211
	(0.232)	(0.332)	(0.520)	(0.252)	(0.533)	(0.531)	(0.0640)
Post Reform	0.0593	0.0947*	0.379***	0.179***	-0.111	0.216**	0.00557
	(0.0402)	(0.0567)	(0.0892)	(0.0425)	(0.0889)	(0.0858)	(0.0121)
Price Shift	-0.908***	-0.742***	1.180***	0.353*	-2.698***	1.286***	0.153***
	(0.192)	(0.278)	(0.413)	(0.211)	(0.445)	(0.450)	(0.0473)
Constant	1.838***	3.133***	0.841***	2.298***	1.984***	1.901***	0.0696***
	(0.0322)	(0.0464)	(0.0682)	(0.0346)	(0.0712)	(0.0693)	(0.00859)
Observations	1,767	1,780	1,780	1,780	1,780	1,780	1,780

Robust standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.1

Table 1.10: DiD Effect on Land Use with Varying Income Assumptions

VARIABLES	(1) Development Threat Level	(2) Agricultural Land Share	(3) Developed Land Share	(4) Natural Land Share	(5) Forest Land Share	(6) Wetland Land Share
Panel A: Income 6x Median						
Post * Price Shift	1.388***	0.244*	-0.0259	-0.218	-0.293*	0.0741
	(0.455)	(0.145)	(0.0630)	(0.151)	(0.154)	(0.0620)
Post Reform	-0.435***	-0.0750***	0.000345	0.0747***	0.0278	0.0363***
	(0.0646)	(0.0189)	(0.00947)	(0.0198)	(0.0204)	(0.00920)
Price Shift	-1.629***	-0.726***	0.204***	0.522***	0.427***	0.0590
	(0.323)	(0.129)	(0.0539)	(0.133)	(0.134)	(0.0446)
Constant	7.028***	0.388***	0.0656***	0.547***	0.496***	0.0335***
	(0.0447)	(0.0158)	(0.00755)	(0.0163)	(0.0165)	(0.00619)
Panel B: Income 2x Median						
Post * Price Shift	0.855**	0.297***	-0.0441	-0.253**	-0.345***	0.0841
	(0.431)	(0.114)	(0.0698)	(0.121)	(0.129)	(0.0628)
Post Reform	-0.388***	-0.0332	-0.00949	0.0427*	-0.0132	0.0439***
	(0.0900)	(0.0223)	(0.0160)	(0.0243)	(0.0262)	(0.0133)
Price Shift	-1.660***	-0.746***	0.192***	0.554***	0.428***	0.0759*
	(0.295)	(0.0975)	(0.0591)	(0.104)	(0.107)	(0.0457)
Constant	6.862***	0.313***	0.0845***	0.603***	0.539***	0.0414***
	(0.0602)	(0.0186)	(0.0133)	(0.0200)	(0.0211)	(0.00887)
Panel C: Constant Income						
Post * Price Shift	0.271	0.171	-0.0129	-0.158	-0.282**	0.109*
	(0.416)	(0.124)	(0.0620)	(0.130)	(0.136)	(0.0612)
Post Reform	-0.523***	-0.0659***	-0.00310	0.0690***	0.0142	0.0427***
	(0.0750)	(0.0197)	(0.0118)	(0.0210)	(0.0222)	(0.0108)
Price Shift	-1.832***	-0.723***	0.184***	0.539***	0.425***	0.0743*
	(0.289)	(0.108)	(0.0517)	(0.112)	(0.115)	(0.0420)
Constant	6.945***	0.356***	0.0731***	0.571***	0.514***	0.0370***
	(0.0524)	(0.0166)	(0.00965)	(0.0174)	(0.0180)	(0.00705)
Observations	1,780	1,780	1,780	1,780	1,780	1,780

Robust standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.1

Table 1.11: DiD Effect on Environmental Value, Binary Treatment

VARIABLES	(1) Mean Priority	(2) Maximum Priority	(3) Biodiversity Priority	(4) Watershed Priority	(5) Agricultural Priority	(6) Forestry Priority	(7) Recreational Priority
Post * High Value	-0.242***	-0.260**	-0.302	-0.0834	-0.361*	0.0853	-0.0130
	(0.0811)	(0.117)	(0.186)	(0.0893)	(0.190)	(0.190)	(0.0241)
Post Reform	0.124*	0.148	0.586***	0.112	-0.00892	0.168	0.0140
	(0.0699)	(0.0996)	(0.161)	(0.0748)	(0.155)	(0.152)	(0.0227)
High Development Value	0.245***	0.259***	-0.343**	-0.135*	0.846***	-0.275*	-0.0326*
	(0.0647)	(0.0946)	(0.143)	(0.0713)	(0.157)	(0.159)	(0.0173)
Constant	1.805***	3.063***	1.035***	2.442***	1.695***	2.017***	0.0709***
	(0.0526)	(0.0767)	(0.117)	(0.0562)	(0.122)	(0.121)	(0.0156)
Observations	1,286	1,298	1,298	1,298	1,298	1,298	1,298
R-squared	0.012	0.006	0.040	0.018	0.040	0.008	0.017

Robust standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.1

Table 1.12: DiD Effect on Easement Land Use, Binary Treatment

VARIABLES	(1) Development Threat Level	(2) Agricultural Land Share	(3) Developed Land Share	(4) Natural Land Share	(5) Forest Land Share	(6) Wetland Land Share
Post * High Value	-0.0197	-0.00368	0.0141	-0.0104	0.0445	-0.0599***
	(0.149)	(0.0432)	(0.0252)	(0.0460)	(0.0484)	(0.0221)
Post Reform	-0.501***	-0.0633*	-0.0159	0.0793**	-0.00251	0.0786***
	(0.136)	(0.0332)	(0.0250)	(0.0364)	(0.0393)	(0.0193)
High Development Value	0.635***	0.188***	-0.0649***	-0.123***	-0.0947**	-0.0105
	(0.102)	(0.0378)	(0.0210)	(0.0398)	(0.0405)	(0.0136)
Constant	6.660***	0.285***	0.0996***	0.615***	0.548***	0.0367***
	(0.0914)	(0.0281)	(0.0208)	(0.0302)	(0.0312)	(0.00964)
Observations	1,298	1,298	1,298	1,298	1,298	1,298
R-squared	0.062	0.062	0.049	0.032	0.008	0.030

Robust standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.1

1.10 Conclusion

While the 30x30 goal measures conservation achievement in acreage, the quality of the land preserved will decide whether society meets the ultimate goals of the Kunming-Montreal agreement and, indeed, of conservation more broadly: to preserve species and ecosystems, to provide clean air and water, to supply natural resources like wood and food, and to protect cultural systems like recreation and agriculture that are tied to the health of the land. Oliva and García Frapolli 2024 describes the tendency to set goals in area protected as a "surface bias" which risks blinding us to the actual value of conservation. This paper shows that in the voluntary market for private conservation, quality will not necessarily follow quantity. The conservation easement tax incentive program in Virginia has successfully attracted large volumes of land for conservation, with the state's protected acreage increasing 24% between 2002 and 2011 (O'Bannon 2012). However, that land varies widely in environmental value, and increasing subsidies for easements does not fix this. It is very concerning that the land under private conservation more closely resembles a random assortment of undeveloped parcels than it does the areas deliberately chosen for public conservation.

Going forward, policymakers who want to expand voluntary conservation should be wary of these quality concerns. If offering credits, programs can help check a potential increase in low-quality applicants by restricting eligibility to lands that meet a higher environmental standard. Conducting the VaNLA mapping assessment marked a step towards improving conservation targeting in Virginia; using this detailed map of priorities to create better standards for the large state credit program could mark another. In Virginia's case, this could have substantially boosted the amount of high-quality lands conserved at a very reasonable cost in terms of total acreage. In other states, raising the quality threshold for large incentives could involve competitive application processes where state grants would go only to parcels offering the highest environmental value. At the federal level, it is worth exploring whether the parcels that claim tax incentives based only on their "open space value," as opposed to historical or environmental value, truly provide enough public value to justify the incentive's costs. Simply leaving the quality screening role to land trusts is not necessarily enough, as the case of Virginia shows. Most land trusts accept parcels with a wide range of quality, which is a reasonable response when they only need to bear the cost of monitoring an easement and not of the full state, federal, and local tax losses.

The size of this quality problem likely varies across regions. Albers, Ando, and Chen 2008 offers some evidence that the environmental quality of private conservation varies considerably across states: private conservation acreage in Illinois is concentrated in high environmental value counties, while Massachusetts conservation acreage correlates more strongly with income. The decline in marginal quality estimated in this paper suggests that Virginia's very high subsidy rate, now approximately 60% from state and federal subsidies, may have attracted many low-quality easements that would not have happened without these subsidies. In addition, the correlation between high development value and low environmental value contributes to this problem. Further work should explore how widespread this problem is nationally.

Private conservation may also vary in distributional effects, both in terms of who receives tax benefits for easements and who benefits from the environmental values of conserved land.

The easement tax credit is extremely regressive, with more than 95% of federal deductions for land and easement donations in 2005 going to households with incomes over $100,000 and 22% going to households making over $10 million (Wilson 2005). Further, as tax policy changes where conservation happens, it will also shift who benefits from the conserved lands. Virginia's private conservation spending is heavily concentrated in the wealthiest and whitest counties in the state, as shown by Vogelsong 2021 and my work here. Conservationists far beyond Virginia are concerned about similar patterns. Within Virginia, at least, the easements attracted by lower subsidy rates are more likely to be in lower income areas, though they remain concentrated in the least racially diverse areas of the state. Future research on attracting conservation land to disadvantaged areas could help ensure that more of the population can share in the benefits of private land conservation.

Chapter 2

Additionality and Duration in Cover Cropping Incentives[1]

2.1 Introduction

Agricultural producers shape the land's environmental health through the practices they choose. Different methods of planting, tilling, and managing weeds and pest can store or release carbon, pollute or preserve waterways, and foster or hinder biodiversity. Environmentally beneficial agricultural practices may provide enough financial benefits to be profitable on some farms while not on others, depending on a farm's economic and environmental circumstances (Wittwer et al. 2017). As such, since the Soil Erosion Service formed in 1933 to prevent the recurrence of the Dust Bowl, government has played a role in encouraging American farmers to adopt soil-preserving practices (Turner et al. 2014). Today, the expanding market for carbon credits has also reached agriculture. These crediting programs pay for producers to adopt practices that increase the carbon sequestered in soil, such as reducing tillage or planting cover crops. Cover crops are planted during times when the ground would otherwise be left fallow, protecting the soil until the farmer kills the cover crop to make way for the cash crop. Cover cropping can control runoff and erosion (Laloy and Bielders 2010), reduce water pollution (Kladivko et al. 2014), allow farmers to reduce the use of artificial fertilizers and herbicides (Schipanski et al. 2014), and increase the level of carbon sequestered in the soil (Poeplau and Don 2015).

Conservation agriculture programs face some key challenges to creating long-term benefits, as improvements in soil quality can easily be reversed if a farmer reverts to their previous practices. Soil carbon includes multiple chemical forms, some of which remain unchanged for thousands of years and others that can degrade and release stored carbon within a decade if a farmer abandons their soil conservation practices (Dynarski, Bossio, and Scow 2020). This problem cannot be solved by simply lengthening the duration of carbon contracts. Farmers in the soil carbon credit market are uninterested in long-term contracts for practices, preferring the flexibility of shorter contracts, and would likely require unfeasibly high payments to make longer contracts (Drechsler, Johst, and Wätzold 2017). Most soil carbon credit standards

[1] The findings and conclusions in this paper are those of the author and should not be construed to represent any official USDA or US government determination or policy.

today offer farmers contracts lasting 5 to 30 years (Oldfield et al. 2022), and the USDA's long-running Environmental Quality Incentives Program (EQIP) provides incentives for 1 to 5 years after which a farmer cannot get another EQIP contract for the same practice on the same field (USDA NRCS 2018).

However, the designers of these incentive programs often expect that providing short-term incentives may lead to long-term practice change. First, the short-term incentives may help farmers overcome short-term adoption costs. Cover cropping introduces new seed costs immediately but takes several years to improve the soil enough to bolster yields, so the practice may need three or more years to become profitable (Myers, Weber, and Tellatin 2019). Short-term incentives may also help farmers learn how beneficial a practice is for them, and how best to implement the practice in their area: what cover crop species to plant, when to plant it, and how and when to kill it. Farmers cite uncertainty around the economic benefits of conservation agriculture practices as a key barrier to implementing them (J. G. Arbuckle and G. Roesch-McNally 2015; Reimer, Weinkauf, and Prokopy 2012; Conservation Technology Information Center 2023; Gonzalez-Ramirez, Kling, and J. Gordon Arbuckle 2015). Accordingly, access to information about cover crops plays a strong role in practice adoption (Baumgart-Getz, Prokopy, and Floress 2012).

To understand the social benefit of these carbon market contracts and short-term incentive programs, we therefore must understand if or when short-term contracts lead to long-term practice changes. Understanding the duration of practice change has important implications for the cost-efficacy of incentives and interventions. If farmers only need financial incentives to get through the initial costs of adoption and generally continue the practice afterwards on their own, then a few year's spending in support may create decades of benefits. On the other hand, where practices never become profitable to the point that farmers choose to sustain them on their own, a program may only provide benefits for as long as it continues payments. I introduce a model that provides insights into the relationship between short-term additionality and long-run impacts, driven by the underlying adoption costs and long-term costs that program participants face. Additionality is the degree to which a program creates behavior change compared to what participants would have done in the business-as-usual case. If all participants in a cover cropping program were already planning to cover crop, the program will have no additionality; if none of them would have done so, the program is completely additional. Change must be both additional and persistent to create substantial reductions in environmental damages, particularly in regards to nature-based solutions for climate change. A low additionality program creates little to no benefit in the short or long run unless it is cheap enough and widespread enough to compensate for the small individual impact. The level of additionality is estimated to vary widely across agricultural practices (Mezzatesta, D. Newburn, and Woodward 2013; Pannell and R. Claassen 2020). Low practice duration would mean a high risk of rerelease of carbon, increasing the optimal ex ante discounting rate of temporary storage (Murray, Sohngen, and Ross 2006; Lötjönen et al. 2024). In the literature, these two problems are often discussed as separate concerns.

In this model, I provide a framework suggesting that the two problems can be traced to the same economic fundamentals. Participants with higher adoption costs and/or annual costs are more likely to be additional, but they are more likely to persist with the practice post-contract when a practice is at least slightly profitable to maintain in the long run. Thus, if potential participants differ by adoption costs and share the same long-term costs, targeting the "most additional" participants will also target those with the greatest potential for long-term change. However, if potential participants differ mostly by long-term costs, the "least additional" participants will be always-adopters for whom the payment makes no difference, and the "most additional" participants will be those with high long-term costs who will be quick to drop the practice after payments end. In this case, targeting the "moderately additional" participants for whom the practice is near the edge of profitability will have the highest long-term impact, even if this group has lower levels of short-term additionality. This framework thus simplifies the difficult-to-estimate metrics of additionality and long-term impact into two concrete questions: how large is the adoption barrier to a practice? And how costly is the practice to maintain? Encouraging joint consideration of short-term and long-term impact can help programs better optimize their targeting of participants.

I empirically explore this question in the context of EQIP, focusing on EQIP contracts for cover cropping. Founded in 1996, EQIP is the largest and longest running program providing incentives for conservation on working lands nationally, contracting with at least thirty thousand farmers per year from 2014 to 2024 (NRCS 2024). With access to the USDA's ProTracts database of EQIP applications, I use a regression discontinuity around EQIP application scores to compare cover-cropping rates of barely-successful and barely-unsuccessful EQIP applicants.

I find that receiving an EQIP cover-cropping contract increased cover cropping rates among successful applicants by 95% during the contract period. This indicates that 95% of EQIP cover cropping contracts are additional, meaning that the farmer would not have planted a cover crop without the EQIP contract. Numerous papers have attempted to estimate the additionality of cover cropping payment programs, often using matching approaches (Mezzatesta, D. Newburn, and Woodward 2013; R. Claassen, E. N. Duquette, and Smith 2018; Sawadgo and Plastina 2021), modeling of adoption costs (Lichtenberg, H. Wang, and D. Newburn 2018; Fleming, Lichtenberg, and D. A. Newburn 2018), or county-level estimations (Park et al. 2023) that do not control for whether a producer has applied for or expressed interest in the conservation program. Rosenberg, Pratt, and Szmurlo 2024 provides an instrument that is clearly exogenous to producer interest, using a regression discontinuity across areas eligible and ineligible for expanded EQIP cover crop funding through the National Water Quality Initiative (NWQI). They find that most of the impact of the NWQI on cover cropping comes from increased cover cropping on additional lands under EQIP contracts. However, they cannot control for the other channels through which the NWQI can impact cover cropping, such as technical education support and increased EQIP funding for practices such as reduced tillage that are complementary to cover cropping. Accessing the EQIP application database allows this paper to provide a unique regression discontinuity estimate that minimizes room for omitted variable bias. The resulting 95% additionality estimate for cover crops is similar to the 80% and higher cover crop additionality estimates found in the bulk of the literature (Rosenberg, Pratt, and Szmurlo 2024; R. Claassen, E. N. Duquette, and Smith 2018; Mezzatesta, D. Newburn, and Woodward 2013; Fleming, Lichtenberg, and D. A. Newburn 2018), though it is higher than Sawadgo and Plastina 2021's estimated 54% additionality for cover cropping in Iowa.

This paper also finds that EQIP seems to substantially increase long-term cover-cropping, though estimates are imprecise. Since the literature on cover crop additionality to date has largely focused on the during-contract effect, this long-term effect is a unique contribution to the cover cropping literature, and joins a larger conversation about hysteresis effects in environmental programs. After the expiration of an CRP contract, FIX CITE Rosenberg, Pratt, and Arnold 2022 finds that most fields revert to previous practices at high rates. Environmental programs seem able to create substantially durable change in many cases: Wallander, Bowman, et al. 2017 find that EQIP-induced changes in tillage do persist well beyond the contract period, and 66% of land retired through the Conservation Reserve Program remains as grassland in the 6 years after the program expiration (Barnes et al. 2020).

I test how this relationship between additionality and long-term impact appears in the case of cover cropping. Breaking out the population by region and operation size, I find that that a group's additionality is inversely correlated to its long-term impact. This suggests that targeting the most additional populations for cover cropping contracts would decrease the long-term benefits from the program.

In this paper, I first discuss the EQIP program and the data used in this paper. I then provide the model of the farmer's practice adoption and persistence decisions. I then explain the regression discontinuity methodology, and the present the results in the following section.

2.2 Data and Setting

2.2.1 The EQIP Program

This paper's analysis focuses on the USDA's Environmental Quality Incentives Program (EQIP). Established in the 1996 Farm Bill, the program offers educational, technical, and cost-share incentives to agricultural producers adopting environmentally beneficial practices on working lands. It was first authorized to spend $1.2 billion over 7 years, and has been renewed and expanded in every Farm Bill since.

Producers can apply for assistance with adopting practices that reduce water or air pollution, conserve water, control soil erosion, or protect habitat for at-risk species (Stubbs 2011). EQIP has historically funded many key greenhouse-gas-emission-reducing practices, such as reducing tillage and planting cover crops, for their value in reducing soil erosion as well as air and water pollution. The Inflation Reduction Act of 2022 substantially increased funding specifically for emission-reducing practices. Cover cropping is EQIP's most commonly funded practice, drawing $504 million in payments made between 2017 and 2022 (Environmental Working Group 2023).

In an EQIP contract, a producer receives a payment for performing certain agreed-upon conservation actions. The state-level Natural Resources Conservation Service (NRCS) office chooses the size of the incentive for each eligible practice. The payment amount is set to cover a maximum of 75% of the expected direct costs of implementing a practice in that state and up to 100% of the expected revenue to be lost through reduced output, when applicable (USDA NRCS 2018). The producer receives the payment once the USDA certifies they have implemented the practice. For practices like cover crops or reduced tillage that must be repeated annually, the EQIP contract requires farmers to continue the practice for one to five years[2], and the producer is paid a partial payment each year once the USDA has confirmed

[2] The 2018 Farm Bill, which covers some contracts that would begin but not end in my study period,

they completed the practice that year.

To participate in EQIP, producers must go through an application process. Acceptance rates have varied widely across years from 25% to 45%, reflecting year-to-year variation in EQIP funding and in application numbers (Happ 2021). Upon receiving an application, the NRCS first checks for application completeness and program eligibility. Applications are submitted into pools, and eligible applications are ranked and funded within these pools. The State Conservationist's office in each state defines the pools, and often group applicants by environmental concerns addressed, geographic area, beginning and socially disadvantaged producers, or type of crop or livestock produced (USDA NRCS 2018). The State Conservationist also chooses how much of the year's EQIP funding to place into each pool to best meet identified state-level needs while also following federal USDA requirements on elements like the share allocated to crop versus livestock producers.

Within the pools, applications are funded in order of their ranking score. Applications receive more points for practices with higher environmental impact, whether they address the NRCS's key program priorities, and also if the land has characteristics like highly erodible soil that make addressing a concern particularly urgent. Applications are also scored on cost-effectiveness (USDA NRCS 2018). Once applications are scored and ranked, the USDA funds each application in a pool from highest scoring to lowest until they run out of funds. The remaining applicants are deferred for consideration until the next year, when they can choose to resubmit their application as is, or they may cancel or modify the application. These deferred applications must then compete against the next year's pool, and again may or may not be funded.

2.2.2 Contract and Applicant Data

Data on successful and unsuccessful EQIP applicants comes from the USDA's ProTracts database, which tracks contracts for NRCS programs including EQIP and CSP from the initial application through the final payments. For each application, ProTracts records the practices covered, the application's funding pool assignment and score, and the contract's status as it moves through the application dataset. ProTracts also includes some geographic data. ProTracts collects a standardized USDA tract number, farm number, and planning land unit number to facilitate matching to individual farms and fields.

Combining the data from 2004 to 2022 produces information on 1.49 million unique applications, with summary statistics reported in Table 2.1. 53% of those have reported pool affiliations and scores, and 18% of these are unsuccessful applications. Cover crops are one of the most common practices, and are part of 10% of applications. The mean application expects $30,000 in payments and impacts 335 acres of land. 55% of applications include information on which practices the contract would cover, which is not always imputed into ProTracts until an application becomes an active contract. Among those, 10% of contracts include cover cropping.

The dataset for this analysis is compiled from annual system pulls from 2004 to 2022, providing a snapshot of contracts that are active or in the application process that year. The exact implementation of these data pulls varied across years. Unsuccessful applications

expanded the maximum contract length to 10 years (USDA NRCS 2018)

Table 2.1: ProTracts Applications Summary Statistics

VARIABLES	(1) N	(2) mean	(3) sd
Contract accepted	1,490,199	0.590	0.492
Share of contract practices certified	1,490,199	0.373	0.461
Ranking score	796,205	523	5,057
Estimated payment ($)	823,423	30,038	1,104,996
Acres treated	1,061,707	335	6,039
Conservation cropping in contract	832,059	0.039	0.192
Cover cropping in contract	835,454	0.100	0.300
Reduced tillage in contract	831,103	0.011	0.105
No-till in contract	832,181	0.044	0.205

were omitted from some years of the data pulls, and the share of unsuccessful applications included may vary within years as well. As Figure 2.1 shows, this results in the number of deferred or canceled applications caught in the pull varying widely across years, particularly in 2011-2013 and 2019-2020, which I therefore omit from my analysis.

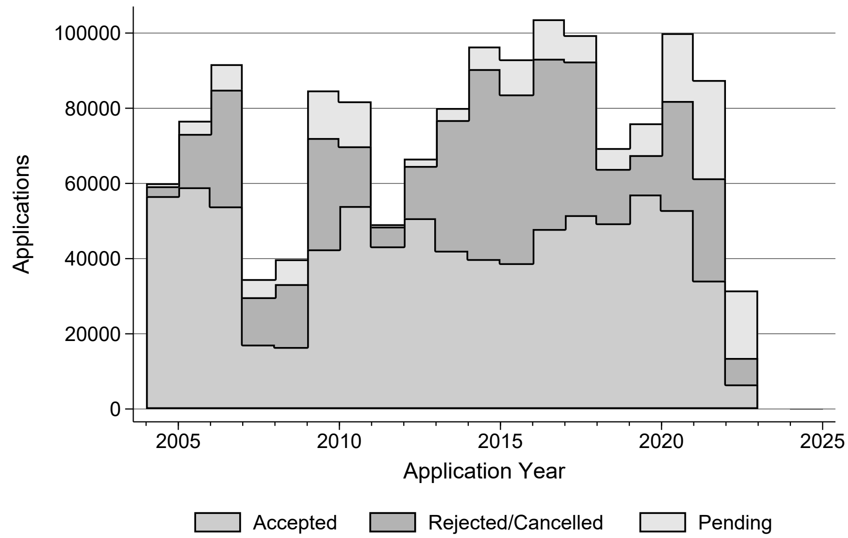
To manage the differential missingness created by these omissions, I identified the years where the data pull included a smoother distribution of successful and unsuccessful applications across the score margin, since this discontinuity may be representative of data issues. Figure 2.2 shows the sharp discontinuity in application counts across the score cutoff, a result of these missing applications. The results in Figure 2.2b show that after omitting applications originally submitted in the low-quality years of 2011-2013 and 2019-2020 and the similarly identified states of most concern, this discontinuity still exists but is somewhat smoothed.

2.2.3 Practice Data

To track the practices implemented on individual parcels, I use two USDA datasets: the Crop Acreage Reporting Database (CARD) from the FSA and the Agricultural Resource Management Survey (ARMS) from the ERS/NASS. CARD compiles data from Form 578, which all agricultural producers must file annually to participate in USDA programs, including insurance or subsidies. Farmers must report what crops they are planting and whether the crop is for harvest, grazing, or cover only. The dataset provided for analysis includes data on 23.6 million fields annually from 2013 to 2019, 855 thousand of which I match to ProTracts cover cropping applications. CARD-reported cover cropping rates stayed fairly even from 2013 to 2019, with 3.5 to 4.5% of fields using a cover crop annually as depicted in Figure 2.3.

From this dataset I construct a measure of cover cropping in line with the broad definition used in current USDA analysis of CARD (Pratt et al. 2023). I classify a field as cover cropping if it reports a cover only planting or if it reports a planting that is not for grain or silage as a second planting after a cash crop. Since farmers may only report one purpose for each crop, this broad definition captures plantings that are serving the role of cover crops

Figure 2.1: ProTracts Applications From 2004 to 2023



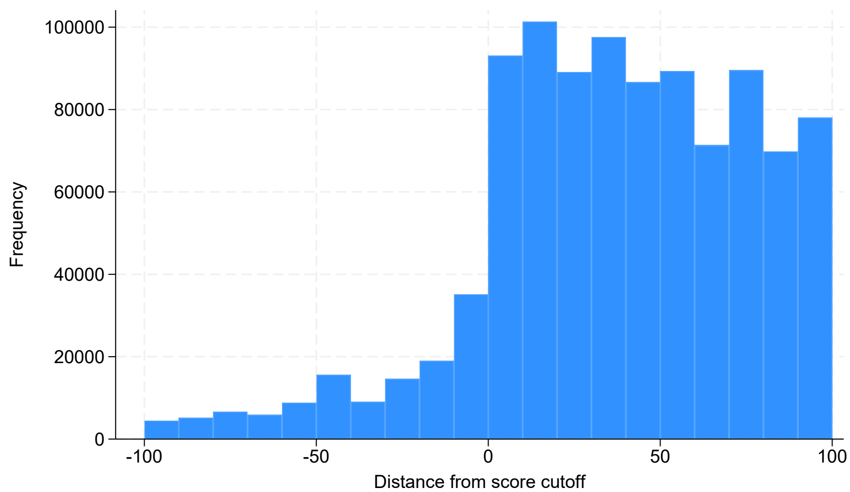
while also fulfilling other uses, such as grazing. In addition, in my main measure, I classify fields with an active EQIP cover cropping contract are implementing cover cropping. This corrects for underreporting through Form 578 among EQIP participants. Very few active EQIP cover crop participants report a cover only crop through Form 578, possibly because they have already reported the cover crop to the USDA through the separate channel used for EQIP reporting. Since 95% of farmers under active contract complete the practices and get payment, this assumption will not substantially overestimate the cover cropping rate for this group.

A key concern with this measure is that farmers other than EQIP participants have also historically underreported non-cash crops on Form 578. The USDA has long asked that farmers report all plantings through Form 578, but farmers only need to report their cash crop plantings to receive full insurance eligibility. Before 2021, they received no incentives or disincentives for accurately reporting cover crops. As such, CARD estimated much lower rates of cover cropping than other data sources such as windshield surveys (Pratt et al. 2023). Therefore, this paper's estimates may be an upper bound on the additionality of EQIP cover cropping and a lower bound on EQIP's long-term effect.

Going forward, I plan to correct this by examining effects in the years 2021 and beyond, when reporting improved after the Pandemic Cover Crop Program introduced a $5 per acre insurance premium discount for farmers that planted and reported a cover crop. This $5 incentive is fairly modest compared to the $30-60 EQIP payments (Myers, Weber, and Tellatin 2019), and was intended primarily to provide additional pandemic-era support to farms using this socially beneficial practice. However, it dramatically increased reporting of cover cropping into line with cover cropping estimates obtained through other methods (Pratt et al. 2023), and the habit of reporting cover crops through form 578 seems to have sustained itself even after the incentives ended in 2022. This analysis will be possible once the CARD data for 2020 and beyond is panelized for analysis, enabling data linkages.

Figure 2.2: Density of Applications Near the Cutoff

(a) Scores near the Threshold: All Merged Data



(b) Scores near the Threshold: Good States/Years

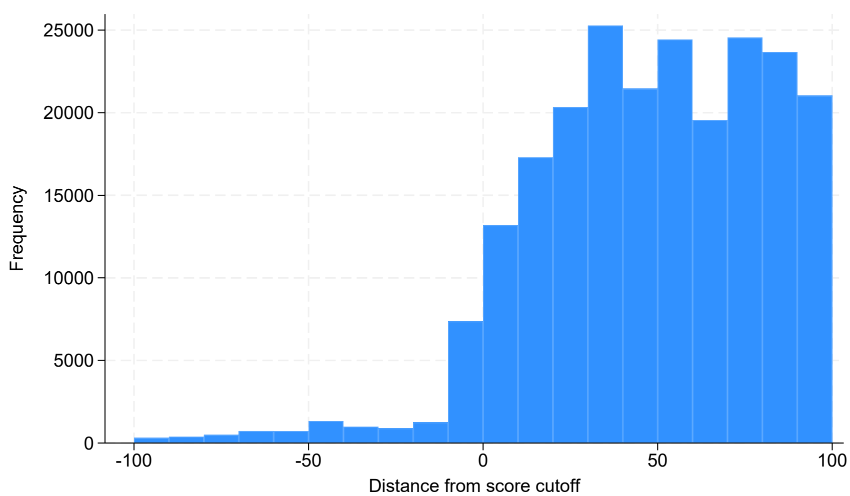
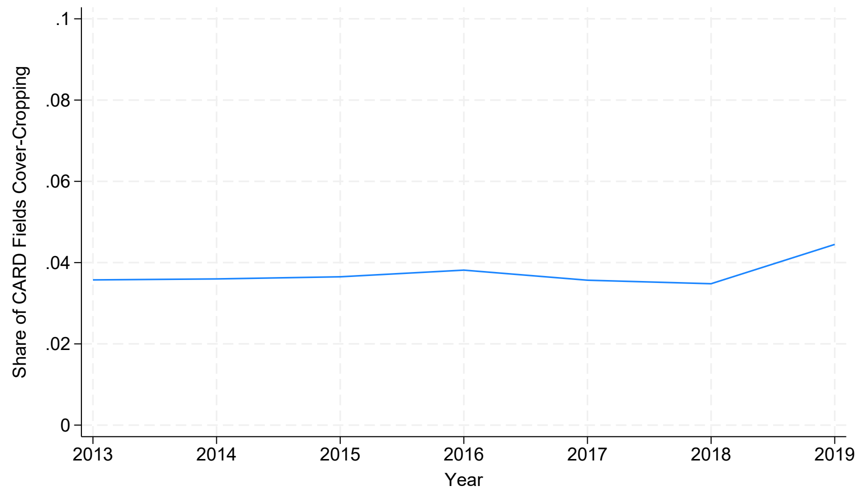


Figure 2.3: Cover Cropping in CARD



2.2.4 Linking CARD and ProTracts

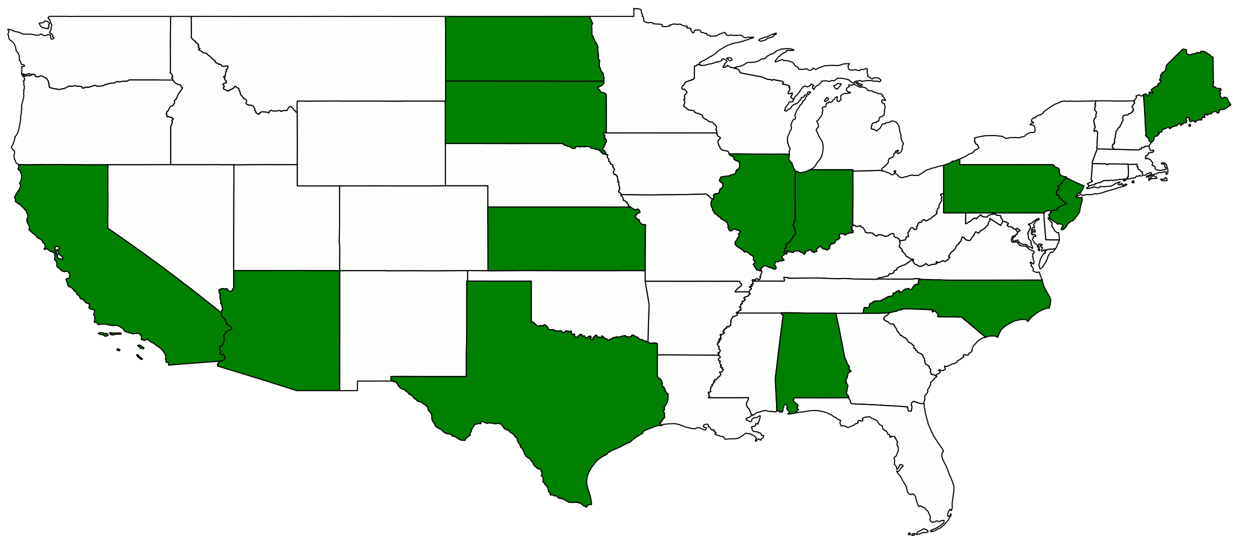
To follow the outcomes of these ProTracts applications over time, I use USDA Planning Land Units (PLU) and Common Land Unit (CLU) identifiers of the field to merge ProTracts to the CARD crop planting data. As part of the application process, the USDA requires applicants to include the PLU information that will let the USDA identify the relevant fields. Most commonly, the PLU field, county, and tract is the same as the USDA CLU that identifies fields in CARD. However, this use of CLUs for PLUs varies across states. 27% of applications have clearly unmergeable field identifiers without numbers, and 60% of applications have PLU numbers that do not actually match to a USDA CLU.

To manage this missingness, I focus my analysis on 13 states that most commonly use CLUs in ProTracts and thus best merge to CARD, depicted in Figure 2.4, and that merge significant numbers of both successful and unsuccessful applications. For the states and years of focus, 72% of ProTracts fields have location data, and 52% of those merge to a parcel in CARD, resulting in 38% of entries matching. After narrowing down this matched sample to those with scores near the cutoff, I use 140,000 matched fields in my main regressions.

2.3 Model

In designing EQIP, the USDA faces a policy problem common to many government programs. Through EQIP, the USDA uses a short-term, voluntary incentive program to induce a behavior change that will provide social benefits. This behavior continues to provide a stream of benefits as long as it is sustained, and stops if the practice reverts. Calculating the total benefit of the program requires the answers to three questions. The first is additionality: during the program's duration, how much does it change participants' behavior from what they would have done otherwise? The second is hysteresis: after the program ends, how much of this behavior change continues? The final element is the translation from behavior change into environmental benefits: among the people who change their behavior, how much

Figure 2.4: States Included in Analysis



environmental impact in terms of carbon stored or pollution averted will their actions create?

A policymaker developing such a program must consider these factors as they decide which populations they wish to target, how much they will offer in payments, and whether they will allow previous participants to re-enroll. In this model, I link a participant's additionality and hysteresis to a pair of underlying factors: the short-term cost of adoption, and the long-term opportunity cost (or benefit) of continuing the practice. Potential participants may vary widely in these short-term and long-term costs, and those costs impact both the additionality and duration components of a contract's effects.

In this section, I first model the farmer's practice adoption decision as a function of adoption costs, long-term practice costs, and a shock to practice profitability. Next, I discuss how the hysteresis/additionality relationship differs depending on whether farmers vary by adoption cost or by long-term costs. I then explore how this relates to the program's cost of contracting. Finally, I discuss some extensions of the model.

2.3.1 The Farmer's Adoption Decision

This section establishes how a farmer will behave with or without a contract, which allows estimation of a their additionality under contract. In this model, a farmer who is not under contract must decide in each time period whether to take a socially beneficial action $x_t = 1$ that produces a social benefit e, or to use a conventional practice $x_t = 0$ that produces no social benefit. If under contract, the farmer must set $x_1 = 1$, but may then freely choose their practice in subsequent periods. In this paper's case, the beneficial action is cover cropping. The farmer's profit in a period t is

$$\begin{aligned} \pi_{kt}(X_{kt}, \epsilon_t) =& b_{0k} \text{ if } x_{kt} = 0 \\ & b_{1k} - a_k D(x_t > x_{t-1}) + \epsilon_t \text{ if } x_{kt} = 1 \end{aligned} \tag{2.1}$$

X_{kt} is the history of practices on field k through time t, b_{xk} is the constant average profit for practice x, $a_k \geq 0$ is the adoption cost of cover cropping. ϵ_t is a time-variant shock in the profitability of cover cropping, with $E[\epsilon_t] = 0$. The farmer learns ϵ_t before they choose period t's practice x_t. I assume that the farmer is risk-neutral. They therefore choose their practices to maximize $E[\sum_t (1-r)^{t-1} \pi_{kt}(X_{kt}, \epsilon_t)]$, the risk-neutral expected profits discounted at rate r. Here, I explore a simple two-period setup, so the farmer chooses x_1 to maximize $\pi_{k1}(X_{k1}, \epsilon_1) + (1-r)E[\pi_{k2}(X_{k1}, \epsilon_2)|x-1]$. The farmer has not cover cropped before, so $x_0 = 0$ and the farmer has not yet paid the adoption cost. In this case, the farmer's period 1 cover cropping decision will depend on three things: the adoption cost a_k, the long-run cost difference $\Delta b_k = b_{0k} - b_{1k}$, and the profitability shock ϵ_k. The farmer will choose $x_1 = 1$ if

$$\epsilon_1 > \Delta b_k + a_k - (1-r)\Delta E[\pi_{k2}] \tag{2.2}$$

where $\Delta E[\pi_{k2}] = E[\pi_{k2}|x_1 = 1] - E[\pi_{k2}|x_1 = 0]$, the expected increase in period 2 profits from having cover cropped and paid the adoption cost in period 1. When $x_1 = 1$, the farmer chooses $x_2 = 1$ so long as $\epsilon_2 > \Delta b$. When $x_1 = 0$, the farmer cover crops only if $\epsilon_2 > \Delta b_k + a_k$. The farmer's total change in expected profits from adopting in period 1 is therefore

$$\begin{aligned}\Delta[\pi_{k2}] =& P(\Delta b_k < \epsilon_2 < \Delta b_k + a_k)(\Delta b_k + E[\epsilon_2|\Delta b_k < \epsilon_2 < \Delta b_k + a_k]) \\ &+ a_k P(\epsilon_2 > \Delta b_k + a_k)\end{aligned} \tag{2.3}$$

Two terms drive this change in expected period 2 profits. The first term is the difference in profits created by the farmer choosing to cover crop in period 2 when $x_1 = 1$ when they would not have if $x_1 = 0$. The second is the farmer's higher profits when the positive shock to cover cropping ϵ_2 is large enough that they choose to cover crop whether $x_1 = 1$ or 0.

The expected probability of additionality of Period 1 cover cropping is therefore

$$Additionality(a_k, \Delta b_k) = P(\epsilon_1 < \Delta b_k + a_k - (1-r)\Delta E[\pi_{k2}]) \tag{2.4}$$

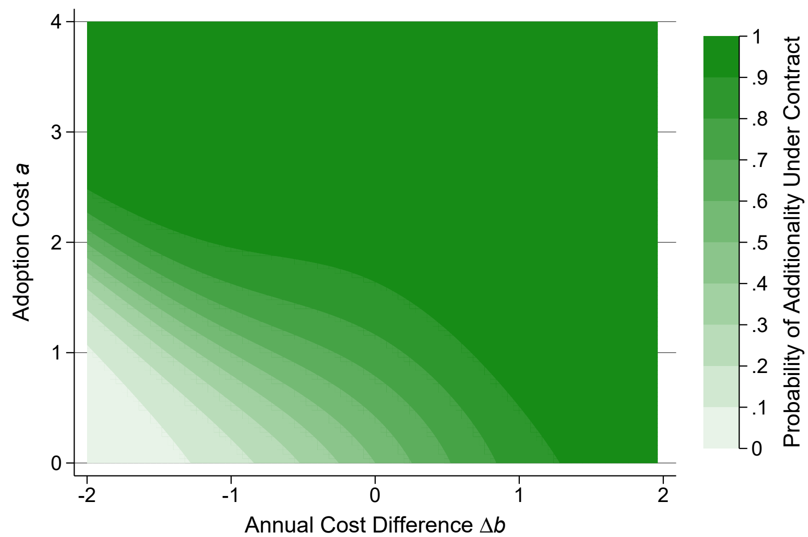
Figure 2.5a models additionality of cover cropping with $\epsilon \sim N(0,1)$. Cover cropping is less likely when adoption costs are higher and when cover cropping provides lower annual profits, so expected additionality is increasing in both a and Δb. If a farmer faces high adoption costs and does not expect to see a long-term annual profit, they are very unlikely to adopt a practice on their own, so an incentive should substantially change their behavior while under contract.

Behavior change under contract is not guaranteed to translate into continued behavior change post-contract. If a farmer cover crops in period 1 and pays the adoption cost, cover cropping will be profitable in period 2 whenever $\epsilon_2 > \Delta b_k$, while if the farmer did not cover crop in period 1, they will cover crop only if $\epsilon_2 > \Delta b_k + a_k$. So

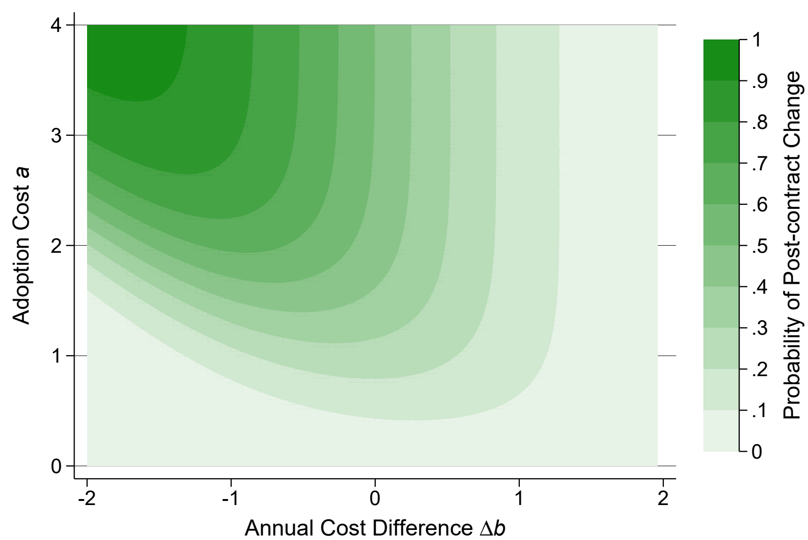
$$LongTerm(a_k, \Delta b_k) = Additionality(a_k, \Delta b_k) * P(\Delta b_k < \epsilon_2 < \Delta b_k + a_k) \tag{2.5}$$

Therefore, the period 2 impact from a period 1 contract will be the first-period additionality times the probability that a period 2 cover crop would be profitable only if the farmer already invested. This equation shows that for a contract to have long-term effects, two conditions must hold. First, the contract must have changed their during-contract behavior, so $Additionality(a_k, \Delta b_k) > 0$. Equation 2.4 shows that this is more likely when the adoption

Figure 2.5: Contract Impacts on Cover Cropping Over Time



(a) During-Contract Additionality of Cover Cropping



(b) Post-Contract Change in Cover Cropping cost or annual costs of the practice are higher. Second, having paid the adoption cost must meaningfully impact whether cover cropping is profitable in some states, so $P(\Delta b_k < \epsilon_2 < \Delta b_k + a_k) > 0$. This term is strictly increasing in the adoption cost: if $a = 0$, then past cover cropping experience has no impact on its future profitability, and a short-term contract does not impact later choices. However, its relationship to the annual cost Δb is more ambiguous. If Δb is so large that farmers would almost always choose to cover crop or so small that they would never wish to cover crop, the adoption cost would not weigh as heavily in their decision. It's the farmers for whom cover cropping teeters on the edge of profitability, the farmers for whom Δb is slightly below 0, for whom $P(\Delta b_k < \epsilon_2 < \Delta b_k + a_k)$ will be the largest. Figure 2.5b shows how this varies across Δb and a.

Figure 2.5b demonstrates that long-term behavior change increases in a similarly to additionality, but differs in its response to annual costs. When adoption costs increase and annual costs remain fixed, additionality and long-term benefits both increase, as Figure 2.6b depicts. On the other hand, long-term impacts are increasing and then decreasing in the annual cost of cover cropping. Figure 2.6b traces a curve of additionality when long-term benefits vary and adoption cost a is a positive constant. When cover cropping is particularly profitable in the long run, farmers are likely to adopt cover cropping in period 1 even in absence of the contract, as the low additionality on the left side of Figure 2.6b demonstrates. Moving right on the graph as the the annual cost of cover cropping increases, rising first-period additionality increases the impact on second-period behavior.

However, as we approach $\Delta b = 0$, the tipping point of long-term profitability, increasing Δb becomes associated with lower long-term impacts. As additionality approaches 1, $P(\Delta b_k < \epsilon_2 < \Delta b_k + a_k)$ begins decreasing because it becomes increasingly unlikely that a farmer would ever find cover cropping profitable without support. Getting a farmer past the adoption cost hurdle no longer matters when the annual profit losses are large enough to discourage cover cropping on its own.

This means that if a policymaker targets participants or evaluates programs based only on their short-term additionality, whether this maximizes total environmental benefits will depend on whether potential participants vary more by long-term or adoption cost. If adoption cost is the primary driver, the higher additionality participants will also have the greatest long-run benefit (or, if the practice is unprofitable for most to sustain in the long run, will at least be no worse.) On the other hand, if long-term costs vary, pushing for the "most additional" short-run participants may lead a policymaker to target the farmers who simply will not sustain a practice after the end of payments.

2.3.2 Contract Cost

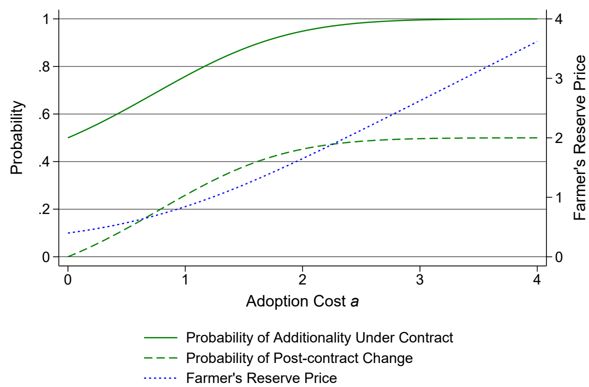
Farmers with different values of a and Δb will also differ in the size of contract payment p needed to induce participation. Combining the payout structure from Equation 2.1 with the cover cropping choices in Equation 2.2, the farmer will accept a contract if

$$p \geq P(\mathit{Additional})(a + \Delta b - E[\epsilon_1|\text{Equation 2.2}] - (1-r)\Delta E[\pi_2]) \tag{2.6}$$

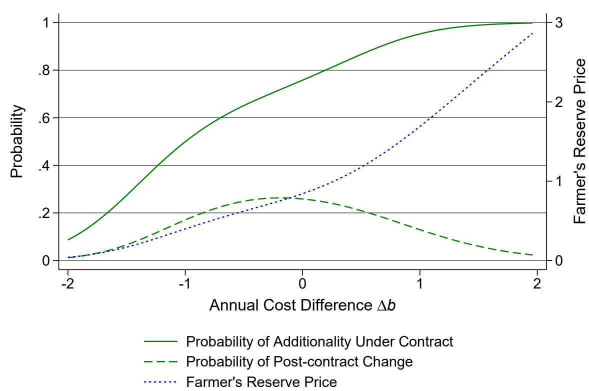
More additional farmers will tend to a have a higher minimum reserve price, since the contract will alter their expected profits only when they expect to change their behavior. Similarly,

Figure 2.6: Exploring The Effect of Cost Variation

(a) The Effects of Varying Adoption Cost



(b) The Effects of Varying Long-term Cost



expected period 1 profit losses increase the farmer's reserve price. However, the needed payment is decreasing in $\Delta E[\pi_2]$, the farmer's expected change in long-term profits from having already paid the adoption cost. If a farmer thinks they will wish to cover crop in the future, they are more willing to pay the adoption cost today.

Graphing farmer reserve prices in Figure 2.7a, reserve prices move in the opposite direction of additionality. Higher adoption costs and higher annual costs increase the chance of additionality, but they drive the farmer's reserve cost upwards.

Given that additionality, cost, and long-term impact may not move together, who should the policymaker target? Assume that land provides some fixed environmental value e each year that a farm cover crops, and the planner discounts the environmental value at the interest rate r. If the policymaker knows the farmer's value of a_k and Δb_k, they can choose to offer farmers contracts with their reserve price p_k. The policymaker's expected value per dollar for offering a farmer this contract is therefore

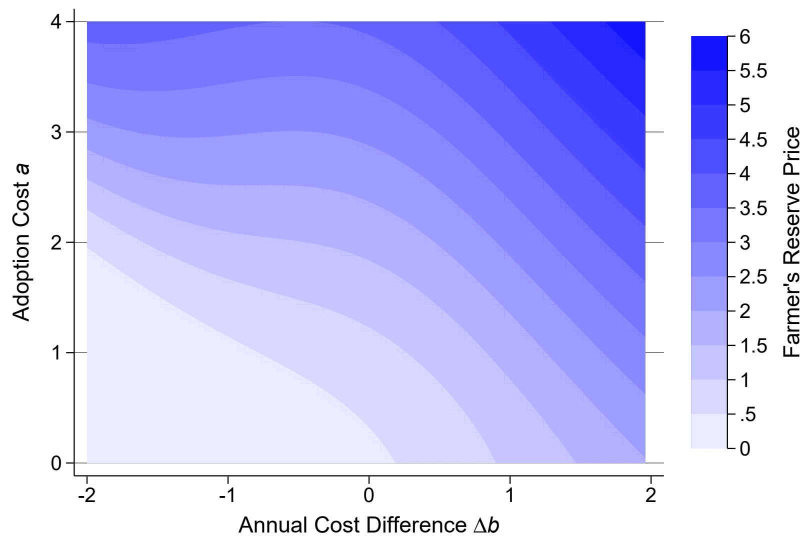
$$Benefit_k/Cost_k = e\frac{Additional_k + (1-r)Longterm_k}{p_k} \tag{2.7}$$

Figure 2.7b plots the results assuming $e = 1$. In this model, it is actually more efficient for the policymaker to target the farmers with $\Delta b < 0$ for whom cover cropping is expected to be beneficial in the long run, since they both demand lower subsidies and have a more persistent effect when they are additional. The exact shape of the relationship between additionality and optimal targeting will vary across situations, as adoption costs, long-term costs, and error structures vary. However, it will rarely be the most cost-effective to target purely the "highest-additionality" populations, since they will require a high cost to implement a practice they would never find profitable on their own, and they are not likely to sustain the practice in the long term.

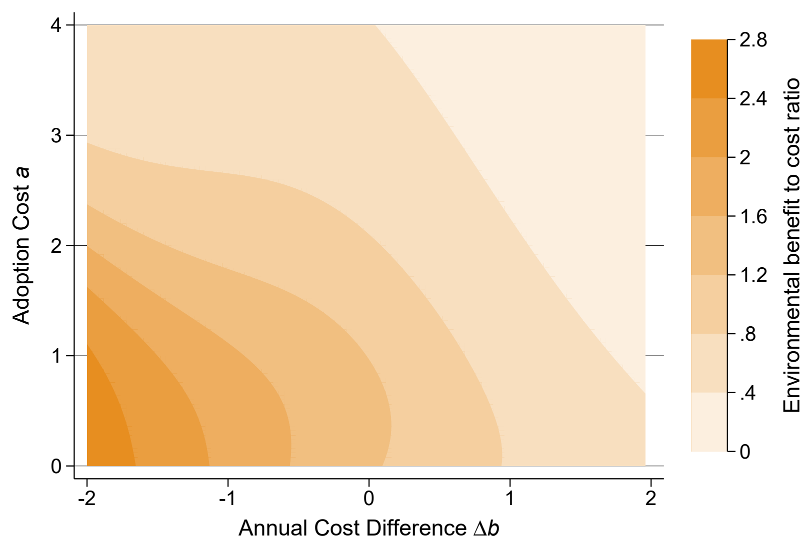
Comparing the farmers' reserve price of a contract also offers insight into who will select into a program. EQIP offers a price per acre p^{EQIP} for cover cropping that is fixed at the state level. Within a state, any eligible farmer whose reserve price is $\leq p^{EQIP}$ should therefore want to apply for EQIP. This means the pool of applicants will look like the sample in Figure 2.7a with $p <= p^{EQIP}$, represented by the lighter share of the graph. These applicants would disproportionately include farmers with low adoption prices and low annual costs, who are therefore particularly willing to adopt Figure 2.5 demonstrates that these farmers are also the ones with the highest chance of adopting a cover crop on their own and thus have lower short-term additionality.

This applicant selection effect explains why papers that compare successful applicants to nonapplicants may overestimate EQIP's additionality. The matching papers do control for a wide set of covariants, hoping to avoid omitting any variables that might determine a farmer's odds of adopting a practice on their own. However, a farmer's decision whether to apply to EQIP or not is a particularly strong signal of willingness to adopt, and one that is omitted from other analysis. A farmer who would never apply to EQIP also would have little or no interest in cover cropping without support, while the applicant pool will include a mix of always-adopters and sometimes-adopters. Since this paper limits analysis to the applicant pool, it limits comparisons to this more similar group.

Figure 2.7: Contract Costs and Net Benefits



(a) Farmer Reserve Prices for Contract



(b) Contract Efficiency at Reserve Price

2.3.3 Applications to Cover Cropping

Where does cover cropping actually fall in this diagram of adoption and annual costs? Scientists and economists have created a considerable literature on the adoption costs and ongoing costs and benefits of cover cropping. Annual costs and benefits are extremely heterogenous: as a farmer surveyed by the Conservation Technology Information Center 2023 noted, "recipes... don't work in a living biological system." Both regional and farm-specific agroecological and economic conditions can greatly impact what cover cropping techniques are most appropriate, and can vary the costs and benefits of that optimal practice. In this section, I discuss the annual and adoption costs and benefits of cover cropping, then discuss what this means for optimal policy.

The annual cost of planting and killing the cover crop is highly heterogeneous based on a farmer's choice of cover crops and management methods. Nationally, total seed, planting, and termination costs may range from \$15-\$78 per acre (Myers, Weber, and Tellatin 2019). In Kansas alone, cover cropping requires between \$42 to \$119 per acre for seed, planting, fertilizer, application, and termination (J. S. Bergtold et al. 2019). Planting costs are more consistent, about \$17.70 per acre in 2019, but seed and fertilizer costs are highly variable: from \$24.50 per acre for crimson clover to \$91.50 for densely planted and fertilized rye (ibid.).

Different choices of cover crop provide different benefits that may allow farmers to reduce their costs through a variety of methods. Cover crops can fix nitrogen, an important nutrient for crops, out of the air, allowing farmers to reduce fertilizer applications (Blanco-Canqui, M. M. Claassen, and Presley 2012). 21% of corn growers surveyed by Conservation Technology Information Center 2023 said that they reduced fertilizer costs by \$20 or more per acre, though half reported spending the same on fertilizer after integrating cover cropping. Cover crops can also reduce herbicide costs in no-till systems, and can help with pest management (Snapp et al. 2005), though the ability of cover crops to help depends on what weed and pest pressure the field previously faced.

The changes that cover crops work on the soil can also lead to increased yield, though this effect is again highly variable. Cover crops can improve soil quality through improving moisture management, reducing soil compaction and erosion, and increasing soil organic matter (J. S. Bergtold et al. 2019). Together, these can help cash crops use nutrients more efficiently and produce larger harvests. In Corn Belt corn and soy systems, the areas that benefit the most from cover cropping may see yield increases of 15%, while the areas that are least suited for cover cropping may see a 5% decrease in yields (Deines, S. Wang, and Lobell 2019). Cover crops can also help with water management, improving yields in drought years in regions of the Corn Belt (O'Connor 2013), though cover crops in arid regions of the Great Plains may harm yields by depriving cash crops of needed moisture (Robinson and Nielsen 2015).

Altogether, the annual profit impact of cover crops varies widely. While many farmers can implement cover crops profitably after an adjustment period, the magnitude of those expected returns for corn may range from \$17 to \$110 per acre after 5 years based on agroecological and economic characteristics including whether the farmers can graze livestock on the cover crops, if they practice no-till, or the year's weather (Myers, Weber, and Tellatin 2019). In Kansas, a cover crop may have a net return of \$7 per year on irrigated land and a net cost of \$28 per year on dryland systems (J. S. Bergtold et al. 2019), while more than half of

South Dakota cover crop adopters believe that cover cropping has had little effect on the profitability of their operation (T. Wang et al. 2021).

This variability in the long-term optimal choice for cover cropping drives much of the adoption cost for cover cropping, which comes in the form of learning costs. Some farmers do need to pay physical adoption costs, such as no-till drills for planting or mechanical crimpers for killing the cover crop (J. S. Bergtold et al. 2019). However, farmers far more frequently cite the challenge of learning how to cover crop as the biggest barrier to adoption. When learning how to cover crop, a farmer risks making some costly mistakes, such as choosing the incorrect timing to kill the cover crop. Using herbicide for termination requires careful timing. Herbicide may persist in the soil and damage the cash crop if the farmer uses the herbicide too close to planting or uses too much for their particular soil and water conditions (Curran 2016). However, waiting longer to kill the cover crop may give additional nitrogen benefits (Sainju and Singh 2001) and help build the biomass needed to derive the full benefits of a cover crop (Morton, J. Bergtold, and Price 2006). Farmers may also take several years to decide how to adjust their fertilizer usage in response to a cover crop. Cover crops can fix N in soils, reducing fertilizer needs, but the rate at which cover crops will make N available to cash crops can vary across locations and situations (Snapp et al. 2005).

Farmers' knowledge of cover cropping and their confidence in their ability to get these choices correct immediately may therefore be a substantial source of adoption cost heterogeneity. Even within a narrow geographic band of South Dakota, farmers who have not practiced cover cropping have widely dispersed beliefs about the impact of cover cropping on profitability (T. Wang et al. 2021). It may be possible to close some of this gap with education and supporting access to farmer networks, which farmers describe as key to getting the information they needed to adopt cover crops (G. E. Roesch-McNally et al. 2018). These farmer networks can provide the hyperlocal knowledge that farmers need, helping them learn from closely comparable farms.

Finally, it often takes farmers several years to realize the full benefits of cover croppng, even when they find the optimal way to implement cover cropping on their land. Cover cropping increases yields primarily through soil quality improvements, such as increased soil organic matter and improved moisture management. These changes accumulate slowly over several years of cover cropping, so many farmers will not see substantial benefits until the soil has improved for 3-5 years (Myers, Weber, and Tellatin 2019). Yield benefits for maize and soybeans increase slowly over the first several years since cover crop adoption, with yield benefits for fields that have used cover crops for a decade or more estimated to be 10 times as large as the effect in the first year (Deines, S. Wang, and Lobell 2019).

Together, this tells us that farmers face some variable adoption costs to cover cropping, and their annual net costs of cover cropping range from the positive to the negative. With the high variation of annual net costs from profitability to prohibitive expense, participants could have any combination of short-term and long-term impact discussed above. Higher short-term additionality does not guarantee a greater long-term impact for a group, since there are farmers who would discontinue cover cropping without incentives.

In the empirical section, I test whether targeting on observables, such as farm size, crop type, or region, would allow policymakers to jointly maximize long-term and short-term impacts, or whether some groups show lower short-term but higher long-term impacts than others. While this data does not let me identify the underlying annual and adoption costs that farmers face, we can get an idea of how this variation matters.

2.3.4 Possible Extensions

This model is simplified to illustrate the roles of short-term and long-term costs of a practice in determining additionality and long-term impact, but it could be extended to accommodate a range of real-world considerations.

First, the model can be extended to more time periods. As the post-contract period grows longer, the long-term opportunities and costs of cover cropping will weigh more heavily for both farmers and the policymaker. As such, adoption costs will become less important to farmers than long-run profitability, and the policymaker will more strongly prefer the farmers with a high expected long-term effect over those with certain short-term additionality.

Also, changing risk structures could create new opportunities for efficient contracting. I currently assume a risk-neutral farmer, but this model could include a risk-adverse farmer. Risks could increase the short-term cost if farmers are uncertain about the adoption costs, or they could reduce long-term costs given that cover cropping can reduce damages from drought or irregular moisture patterns (Myers, Weber, and Tellatin 2019). The risk premium may alter farmers' reserve price for cover cropping, and it will do so differently across farmers if their actual or perceived levels of risk differ. Cover cropping is risk-reducing for farmers in some systems, such as Tennessee cotton (Boyer et al. 2018). If a practice increases risk, risk-neutral policymakers may be able to find more cost-efficient incentive solutions by offering insurance policies as well as fixed payments.

In addition, this model could include a transaction cost for contracting. High transaction costs would make the policymaker shift focus towards contracts with higher additionality and long-term impact, since it becomes expensive to pay the transaction costs for large numbers of cheap but lower-impact low additionality contracts. Introducing transaction costs would also mean that participants could be additional to the program but not to the practice, since some people for whom the practice is profitable would still not join the program unless the benefits are higher than their transaction costs. EQIP applicants report spending a modest 8.4 hours ex ante on planning and applications and 1.9 hours ex post on acceptance and compliance paperwork (McCann and R. Claassen 2016). In this case, perceived transaction costs may be larger than the true transaction costs: 29% of nonapplicant producers surveyed about their reasons for not applying cited the application process as too complicated and time-consuming, and 31% gave the same concerns about documenting compliance (ibid).

2.4 Methodology

2.4.1 Regression Discontinuity Design

This paper instruments for receiving an EQIP contract with a regression discontinuity around the application score. The central regression specification is:

$$
\begin{aligned}
Y_{kgt} = \alpha_{gt} + \beta_1 Score_k + \beta_2 ScoreAcc_{kg} + \beta_3 ScoreAcc_{kg} * Score_k \\
+ \beta_4 ScoreAcc_{kg} * During_{gt} + \beta_5 ScoreAcc_{kg} * After_{gt} + \\
\beta_6 During_{gt} + \beta_7 After_{gt} + \epsilon_k gt
\end{aligned} \quad (2.8)
$$

where Y_{kgt} is the outcome variable for field k in application pool g in year t and α_{gt} is a fixed effect for application pool g. $Score_k$ is the application score for field k relative to the group's threshold variable, and $ScoreAcc_k$ is an indicator variable that equals one if $Score_k$ is greater than or equal to the score acceptance threshold for that pool in that year. $During_{gt}$ and $After_{gt}$ are indicator variables that reference the year relative to the application year: $During_{gt}$ marks the three years after the application when successful applications would be under contract, and $After_{gt}$ indicates years four to nine after an application pool's contract term would end. β_4 and β_5 are the key variables of interest, since they estimate the differential effect of meeting the score threshold on applicant's behavior during the contract period and after the contract period respectively.

Scores are not a perfect predictor of treatment since some participants accepted based on scores later drop out of their contracts and some participants are accepted in later years. Therefore, I use the fuzzy regression discontinuity estimator of treatment effects. For each variable of interest Y, I derive the short term effect of contracting as

$$
TE^Y = \frac{\hat{\beta}_4^Y}{\hat{\beta}_2^P + \hat{\beta}_4^P} \quad (2.9)
$$

where $\hat{\beta}_4^Y$ is the $\hat{\beta}_4$ estimated using Equation 2.8 with variable Y as the dependent variable, and $\hat{\beta}_2^P + \hat{\beta}_4^P$ are the coefficients estimated using Equation 2.8 with the probability of receiving an EQIP contract as the dependent variable. I similarly derive the long-term effect as $TE^Y = \frac{\hat{\beta}_5^Y}{\hat{\beta}_2^P + \hat{\beta}_5^P}$.

2.4.2 Identifying the Acceptance Threshold

To use this technique successfully first requires identifying what the threshold score for acceptance is within each funding pool and year. While accurately tracking the score of each individual application is one of the ProTract dataset's key administrative tasks, it does not explicitly track score cutoffs. Using ProTracts' annual snapshot of accepted and rejected applications, I estimate the threshold for each score through a two-step process. First, I identify pools using the group listed in ProTracts and application batch dates from USDA listings. Second, I find the lowest score of an application accepted in the first reporting year from that batch and use that as the cutoff score.

The first task is identifying the pools. In the EQIP funding process, the state-level USDA office first screens applicants for eligibility, then sorts applications into bins of its choosing. They often define these pools by geographic area within the state, by crop or animal production, or by resource concern such as water quality or wildlife habitat. Within ProTracts, these are listed by fund code. The fund code is reported for 99.8% of applications that have completed the scoring process, and applications without a fund code are omitted from the sample. The median state has 50 pools in a given year.

The USDA then evaluates these pools in batches. All applications received before a certain cutoff date are evaluated for funding at once. Applications received after that date will be rolled into the next evaluation and funding group. State USDA offices may choose their own evaluation dates, and they may perform these evaluations one to four times per year (USDA NRCS 2018).

To track these dates, I cross-reference the ProTracts signup date of the application with the USDA's listed EQIP deadlines for a state. The signup date is the date at which the USDA receives a complete application that is ready for scoring and evaluation. The USDA listed EQIP deadlines come from a centralized page maintained from 2022 to 2024 that linked producers to their state's filing deadlines and application websites. Using the WayBack machine, I recorded all application dates from those years. Older application dates are not systematically recorded. Application dates for a state wavered somewhat over time, but stayed relatively constant: 78% were within the same month, allowing the date to vary to keep the day of the week constant. To accommodate these variations, I assume an actual batch cutoff three weeks after the later date within a month. Application levels decrease immediately after a cutoff, so this method is unlikely to misclassify many applications that belong in the next pool. Also, some offices may have changed the number of cutoff dates within a year over time. Between 2022 and 2024, 15 states had a different number of listed batch dates within at least one year. In those cases I use the highest listed number of batch dates since the Wayback Machine may have failed to capture some of the repeat dates. Since this may split some batches into multiple batches for analysis, it may weaken the power of my analysis, but should not introduce bias.

Within pools, applications are funded in order of scores until the category has allocated all available funds. Pools therefore vary widely by the acceptance rate. 11% of applications in included years are in pools where all applications that meet the minimum eligibility requirements are funded. These fully funded pools are generally smaller, receiving a mean of 2.8 applications per fully funded pool compared to 26.1 per competitive pool. I omit the fully funded pools from my regression discontinuity analysis because they do not have a score discontinuity. Applications that cannot be funded in a year are deferred, and are eligible to subsequently resubmit the same application in another year.

2.4.3 Threats to Identification

In this section I discuss two key threats to identification: manipulation of the score around the threshold, and differential nonrandom data loss. First, this identification relies on the assumption that whether a farmer's application score falls just above or below the threshold is essentially random. The barely successful applicants must not be systematically different from the barely unsuccessful applicants. This can happen if some applicants are aware of their proximity to the threshold and are able to alter their applications to push them over the edge.

Overall, this kind of precision manipulation is highly unlikely in this setting. Applicants need two levers to take advantage of the threshold: they must have some power to manipulate their scores, and they must have an idea of their proximity to the cutoff. EQIP does give producers the first since applicants will receive different scores based partially on the suite of practices they choose to offer. However, anticipating the score cutoff would be quite difficult. Since the state USDA funds applications in a pool until all the pool's allocated funds are committed, the cutoff score depends on both who applies to a pool in a given year and on how much funding the state allocates to the pool. Given fluctuations in the number of applications and funding, acceptance scores can swing substantially across years. Nationwide acceptance rates varied from 15 to 67% between 2000 and 2011, swinging back and forth based on EQIP funding and on application numbers (Stubbs 2011). In addition, the exact formula for the application score is not publicized. In conversation with USDA employees and with other producers in their area, an applicant might glean a rough idea of whether their pool will be particularly competitive in a given year. However, even those USDA employees would find it nearly impossible to predict the exact acceptance score of a competitive pool. The graph in Figure 2.8 of matched scores near the discontinuity bears this out. While there is differential missingness of data across the discontinuity, there is no particular bunching of scores just above or missing mass just below.

The difference in counts of successful and unsuccessful applications in my data is instead due to differential missingness. As discussed in the data section, there are two ways that applications may drop out of my sample. The first is that rejected contracts and deferred contracts that the applicant does not wish to resubmit are purged from the ProTracts system once annually. To manage this problem, I remove some years where the data purge and data pull may have happened in close proximity. The second channel is that some data is lost in the match from ProTracts to CARD, and unsuccessful applications have poorer match rates. While 79% of initially successful ProTracts applications in my target states have numeric geographic data, only 27% of deferred applications have this data. Ultimately, 42% of successful applications and 15% of unsuccessful applications in ProTracts can be matched to CARD.

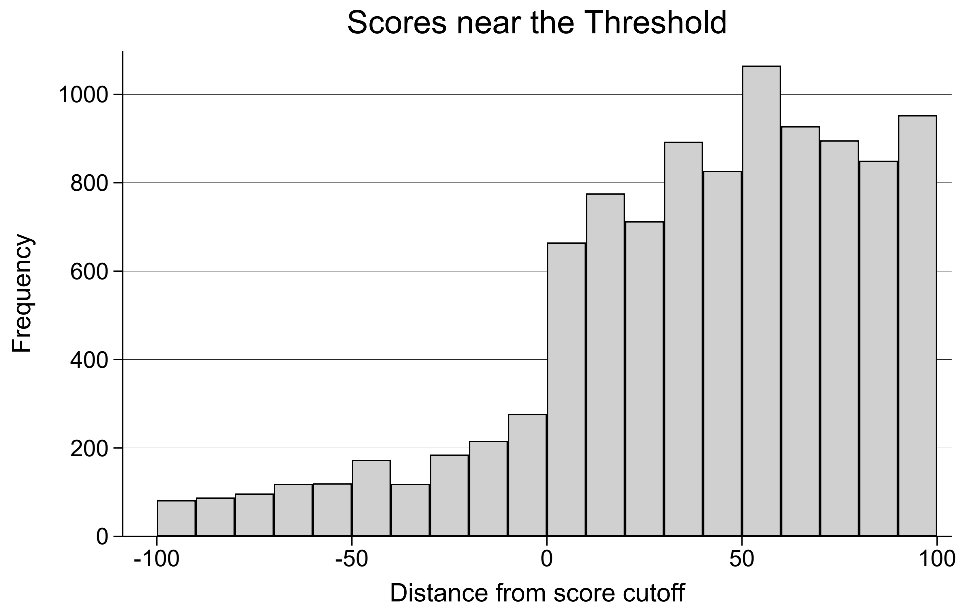
I also check for the frequency of repeat applications and find that while many applicants will resubmit their application until accepted, submitting a new repeat application is relatively uncommon. I find that only 21% of deferred applicants in ProTracts are later accepted. However, since never-accepted applications are less likely to have matchable CLU data, 85% of CARD-matched deferred applicants used in my sample are later accepted. I also find that only 5% of farm fields appear in more than one contract or application over time, suggesting that few farmers who are rejected once consider it worth their time to apply again.

2.5 Results

2.5.1 Difference-in-difference results

First, I find that having a score above the acceptance threshold for the initial application round increases the probability of having the contract ever accepted and the probability of completing a contracted practice by 20%. Figure 2.9 shows the discontinuity. Almost all applications that are deemed eligible accept the contract, as we would expect given that a farmer must put effort into producing their bid. 78% of applicants just below the threshold ultimately end up receiving a contract in subsequent years as their initially deferred application proves successful

Figure 2.8: Scores Near the Threshold



in another year's less competitive application pool. This probability of a later successful application stays quite constant across the hundred-point bandwidth, suggesting there is not much substantial variation in applicant traits across this range. Contract completion by 2020 takes a similar path to contract acceptance, since only 14.5% of farmers with cover cropping contracts do not complete their practices (Wallander, Claasen, et al. 2019). The lower level of contract completion overall reflects that many contracts in this dataset had not yet reached the end of their planned duration by 2020.

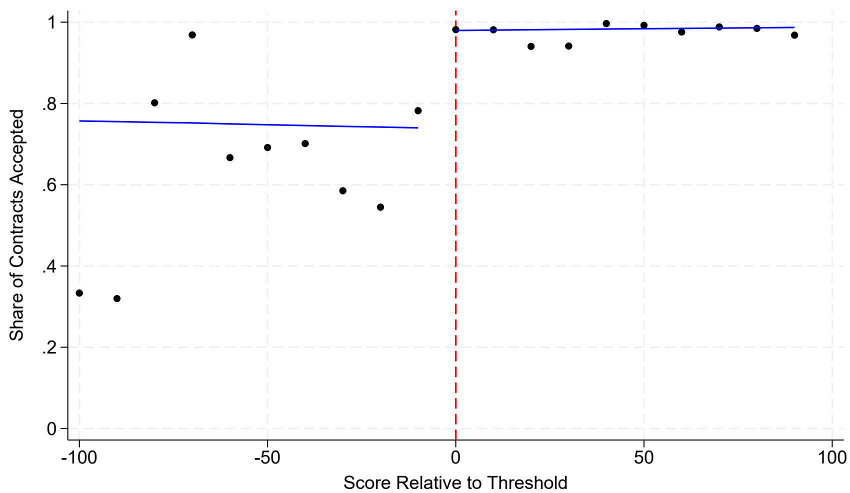
Figure 2.10 shows that cover cropping also increases at the discontinuity, even before controlling for pool fixed effects as in the full regression. By the broad definition, cover cropping increases by 25% at the discontinuity during the contract period. The post effects are difficult to detect without the appropriate controls, but do show a slight increase in post-contract cover cropping.

The results of the full regression discontinuity, shown in Table 2.2, estimate that being above the score threshold increases the chance of contract acceptance for the during-contract group by 47% and increases during-contract cover cropping by 46%. These effects are combined to estimate the contract effect after and during contract at the bottom of Table 2.2. The coefficients are calculated according to Equation 2.9, using the Table 2.2 Column 3 results for $\hat{\beta}^P$. I estimate standard errors for these effects using the delta method. The resulting analysis finds that receiving an EQIP contract increases during-contract cover cropping by 96% during the contract and continues to increase cover cropping after. This high level of short-term additionality is consistent with estimates from elsewhere in the literature (R. Claassen, E. N. Duquette, and Smith 2018, Mezzatesta, D. Newburn, and Woodward 2013, Fleming, Lichtenberg, and D. A. Newburn 2018).

The persistence of change is difficult to detect in the current sample, but potentially quite

Figure 2.9: Discontinuity in Contract Acceptance and Completion

(a) Contract Acceptance



(b) Contract Completion

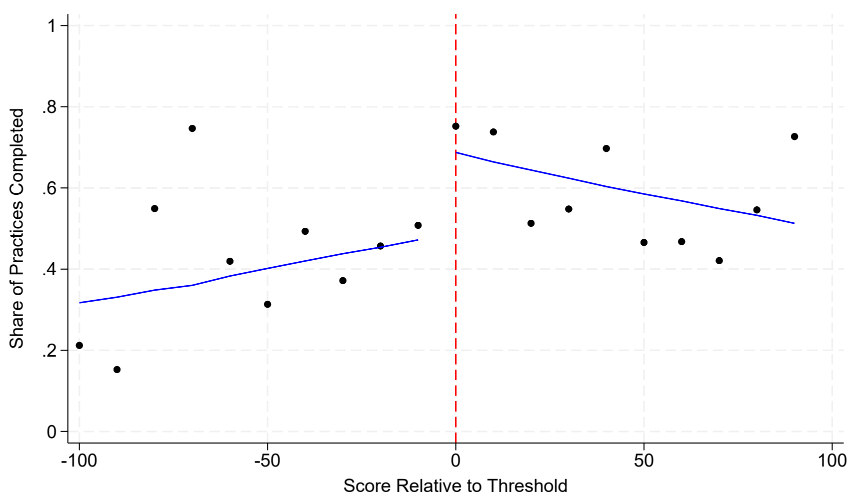
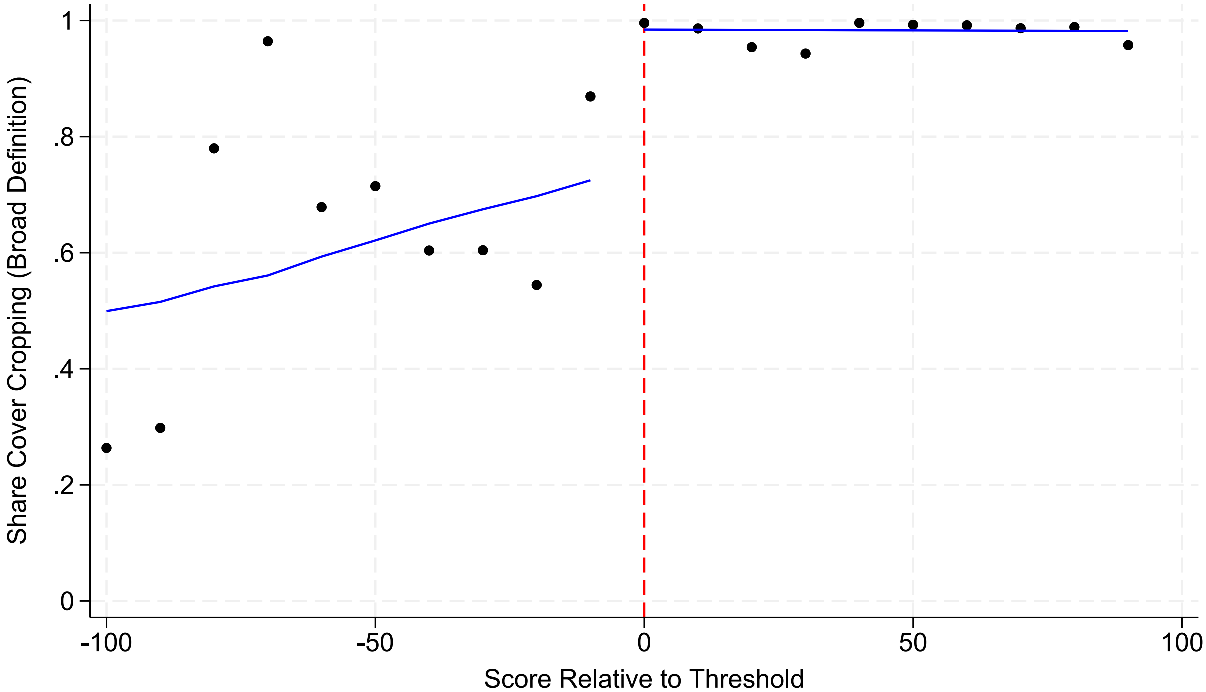


Figure 2.10: Discontinuity in Short- and Long-Term Cover Cropping

(a) Cover Cropping During Contract Period



(b) Cover Cropping After Contract Period

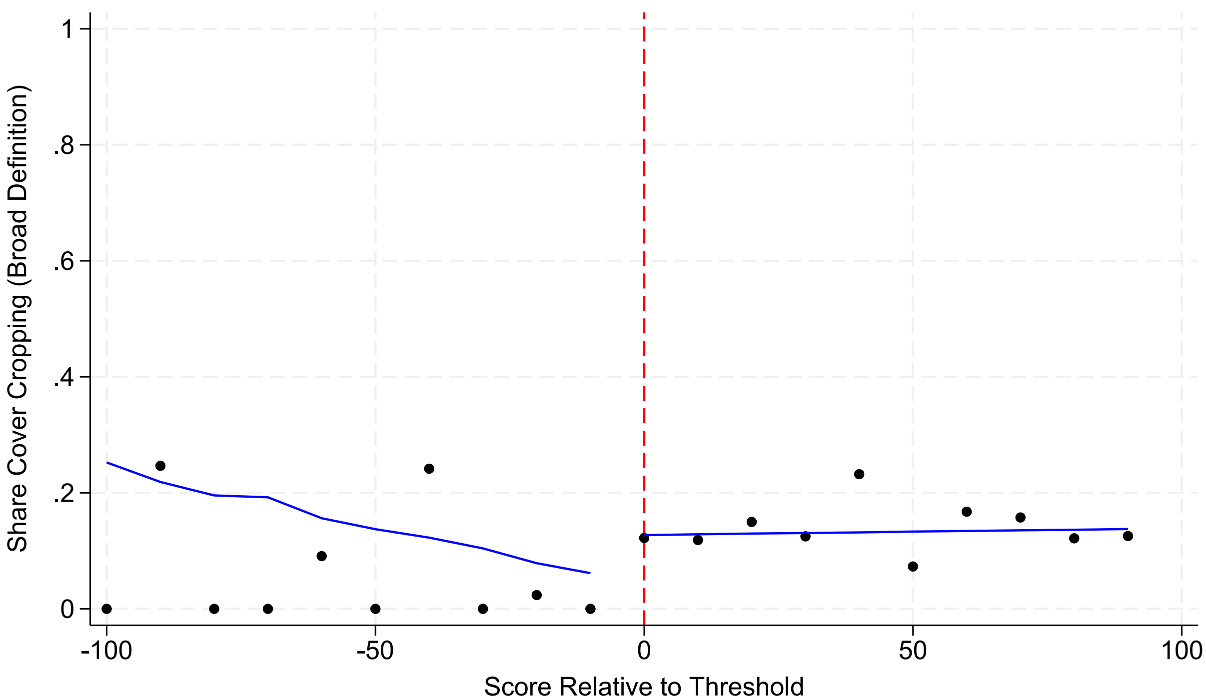


Table 2.2: Regression Discontinuity Results

VARIABLES	(1) Cover Cropping (CARD/EQIP Max)	(2) Cover Cropping (CARD only)	(3) Contract Accepted	(4) Practice Certified
Application Score	0.00109***	0.000512***	0.00317***	0.000716***
	(0.000119)	(0.000136)	(0.000110)	(0.000120)
Score # Score Above Threshold	-0.00157***	-0.000999***	-0.00298***	-0.000677***
	(0.000120)	(0.000138)	(0.000111)	(0.000121)
During Contract	0.459***	-0.0871***	-0.374***	-0.771***
	(0.0131)	(0.0148)	(0.0453)	(0.0575)
After Contract	0.111***	-0.0557***	0.764***	1.450***
	(0.0141)	(0.0132)	(0.0336)	(0.0432)
Score Above Threshold	-0.0351***	-0.0350***	-0.00200	0.00891*
	(0.00507)	(0.00589)	(0.00380)	(0.00500)
Score Above Threshold # During Contract	0.465***	0.101***	0.475***	0.933***
	(0.0128)	(0.0146)	(0.0148)	(0.0184)
Score Above Threshold # After Contract	0.0203**	0.0540***	0.00534	-0.0422***
	(0.00823)	(0.00945)	(0.00621)	(0.00799)
Pool Fixed Effects	Yes	Yes	Yes	Yes
Contract Effect During Contract	0.968***	0.210***		
	(0.0403)	(0.0265)		
Contract Effect After Contract	4.135	10.42		
	(5.331)	(14.05)		
Observations	630,917	614,992	711,903	718,341
R^2	0.695	0.256	0.488	0.903
Number of id	114,312	114,274	113,388	114,312

Standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.1

large. Since the bulk of application pools with comprehensive data on unsuccessful contracts were made between 2014 and 2017 and the cover-cropping panel tracks fields from 2013 to 2019, the post-contract acceptance discontinuity is smaller and based on data with higher missingness. Accordingly, the contract effect after contract as estimated with Equation 2.9 and shown in Table 2.2 has a wide confidence interval, with a coefficient estimate of 4.135 and a standard error of 5.331. As further research tracks the outcomes of the 2014 to 2017 application groups, more precise estimates of the long term effect should become possible.

2.5.2 Effects on Subgroups

This section explores the differential effects on subgroups by region and by acreage under contract. Both tracked in the application database, these variables are used for decisions on targeting funding and awarding contracts, so understanding where EQIP cover cropping is particularly effective and ineffective is valuable for targeting. This section also tests the relationship between duration and additionality across subgroups. As discussed in the model section, additionality could and duration could either covary or oppose one another depending on the distribution of adoption and long-term costs. In this context, I find that subgroups with higher additionality tend to have lower long-term effects, so targeting exclusively on additionality could lead to poorer long-term outcomes.

First I calculate the treatment effect by the acreage included in a cover cropping contract application. I divide contracts into three acreage groups with approximately equal numbers of applications: small contracts with less than 100 acres, large contracts with more than 1000 acres, and the medium contracts in between. I then regress with interactions for size, and calculate each group's treatment coefficient using Equation 2.9. The resulting treatment effects are plotted in Figure 2.11a.

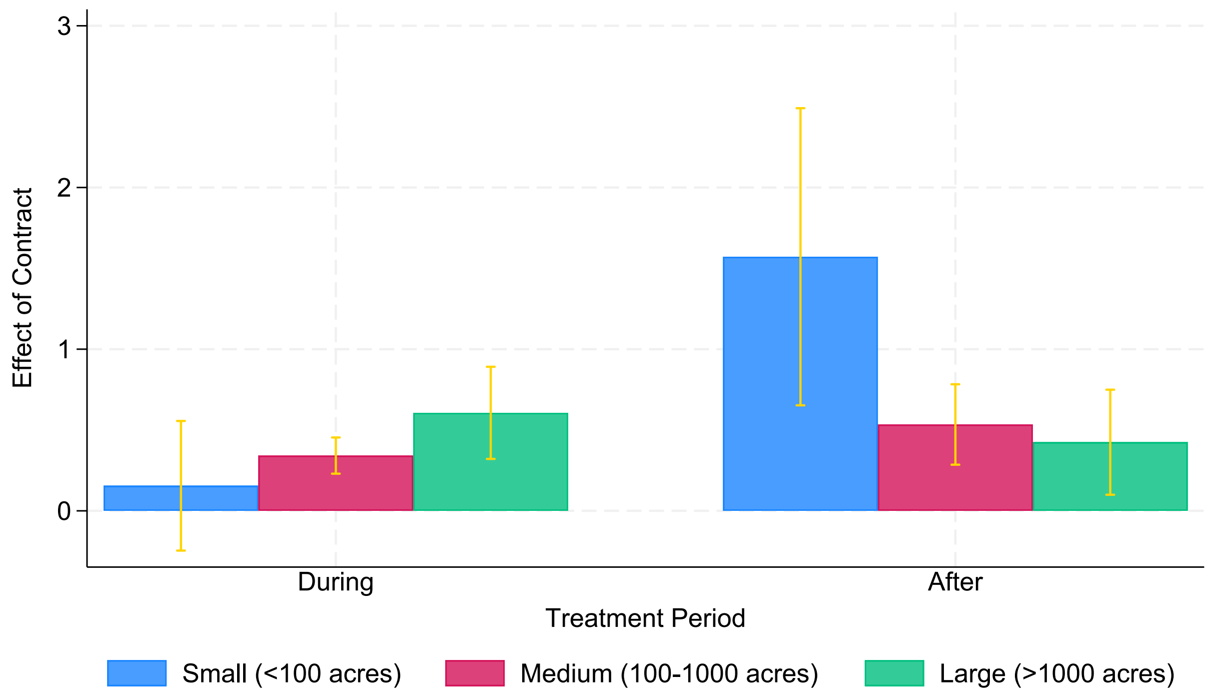
This analysis finds that short-term additionality is increasing but long-term impact is decreasing in contract size. Small contracts covering less than 100 acres have an estimated additionality of 20%, while the largest contracts have 60% estimated additionality. Long-term effect estimates have large confidence intervals, but trend downwards with size: the small contracts have an estimated treatment effect of 1.5, and large contracts have an estimated treatment effect of .6. The 95% confidence intervals for all groups overlap. However, the confidence intervals do show that additionality is either increasing or approximately constant with acreage, and long-term effects are either constant or decreasing with acreage.

I also analyze heterogeneity by USDA production region, with results in Figure 2.11b. While these estimates are often noisy, most regions do not show substantial effects in the short or the long term, including the key Corn Belt region of substantial interest to many private sector cover crop programs. The key exceptions are Appalachia and the Northern Plains. Contracts in Appalachia show the lowest additionality and highest long-term effect of any region, and Northern Plains contracts conversely have the highest additionality and second smallest long-term impact estimated.

Both sets of heterogeneous treatment effects share a trend: additionality and long-term impact of subpopulations pull in opposite directions. To better visualize this, Figure 2.12 plots long-term and short-term effects of contracts together, with treatment effects winsorized to a maximum value of 1 and minimum value of 1 for clearer visualization. For each set of effects, point estimates slope downward across the graph, indicating a tradeoff between

Figure 2.11: Treatment Effects by Subgroups

(a) Differential Effects by Contract Acreage



(b) Treatment Effects by Region

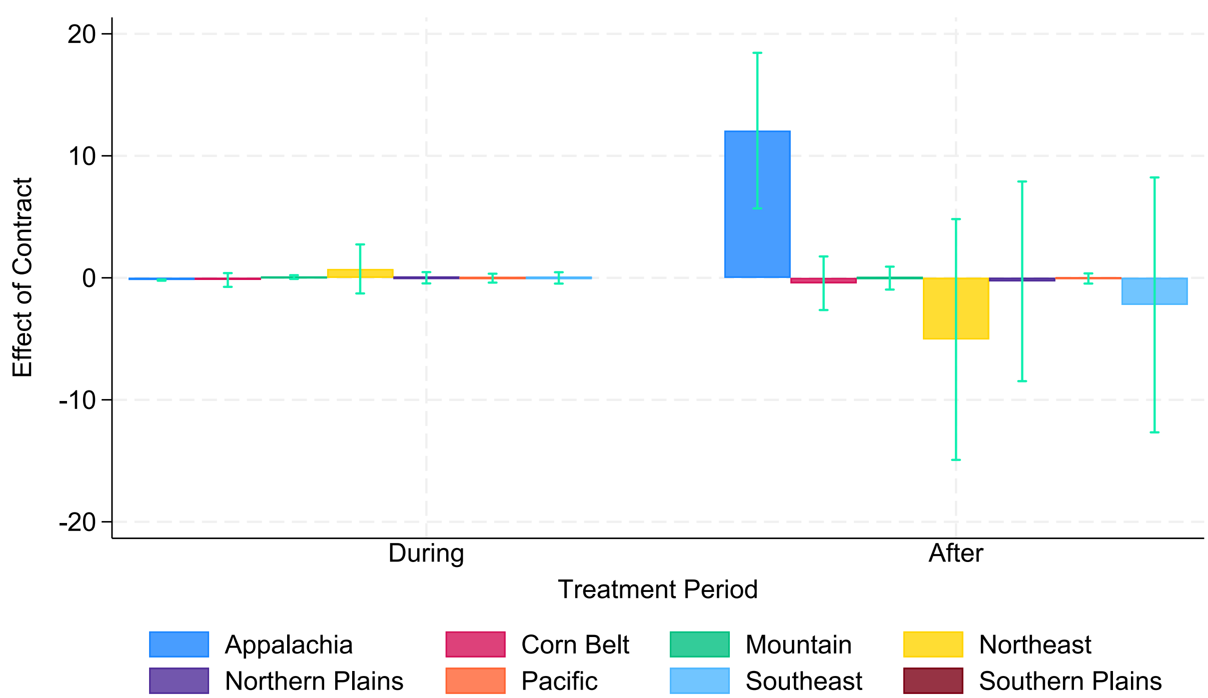
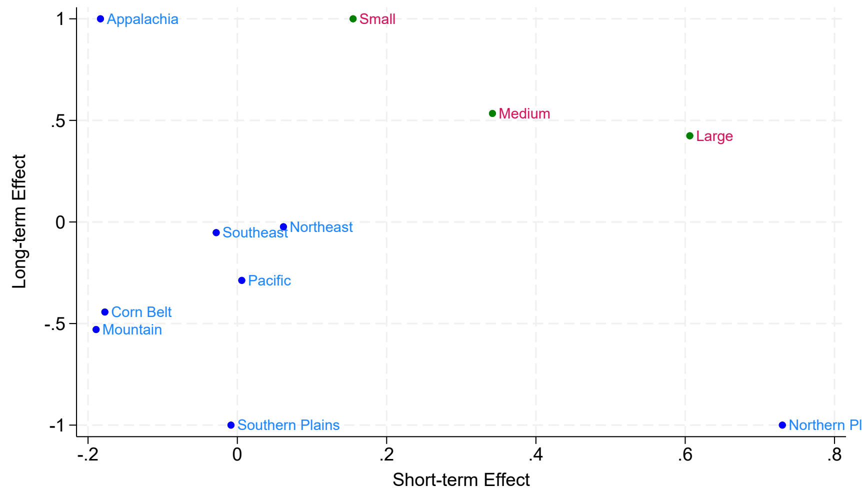


Figure 2.12: Long-term and Short-term Treatment Effects on Subgroups



Effect size winsorized at [-1, 1]

additionality and long-term impact.

This paper's model predicts that this will occur when subgroups primarily vary by the long-term cost of cover cropping rather than the adoption cost. When cover cropping is profitable in the long run, we expect low additionality and higher long-term effects. When it is unprofitable, we expect substantial additionality but high rates of discontinuation after incentives end. This makes improving program targeting difficult. If the NRCS used this paper's additionality estimates to target high-additionality groups like the Northern Plains and large contracts, this would shift funding away from groups where the effect of EQIP will persist more strongly in the long run.

2.6 Conclusion

Like many environmentally beneficial practices, cover cropping requires effort both to adopt and to maintain. The perceived size of the adoption costs and the the long-run costs or benefits can determine whether a farmer will adopt the practice on their own, whether they'll apply for programs like EQIP that offer transition funding, and whether they'll continue the practice after incentives end. This paper finds that the EQIP cover cropping program participants are largely additional, with program acceptance increasing the chance of cover cropping by 95%. The translation into long-term practice change is uncertain.

However, subgroup effects suggest that long-term and short-term effects cannot easily be jointly targeted for cover cropping. As this paper's model explains, long-term effects of short-term incentive programs depend on both the practice's additionality and on farmers' willingness to persist with it after incentives end. Farmers for whom cover cropping would be somewhat profitable in the long run are more likely to adopt on their own, but they are also more likely to keep using the practice. Subgroups such as small farms and those in Appalachia seem to fall into that category where low short-term effects are paired with higher long-term effects. Conversely, groups for whom the practice is less profitable to sustain will be more additional but have a lower long-term impact, which fits the results for larger parcels and the Northern Plains region. Adoption costs may be similar across these subpopulations with some random variation, but these results are not consistent with a strong positive or negative correlation between adoption costs and long-term costs.

This indicates that in cover cropping, targeting exclusively based on additionality may lead to long-term losses. In this context and others, evaluators measure additionality more frequently than long-term impact, and policymakers may be tempted to focus on improving additionality through adding new eligibility restrictions. That instinct could increase a program's impact if participants mostly vary by adoption costs. However, if that variation in additionality is due to long-term costs, policymakers risk cutting into their program's long-term impact. Cover cropping is known to have highly variable long-term benefits, and the subgroup results in this paper suggest that the variation in long-term benefits dominates any effect of variation in adoption costs.

These findings also highlight potential challenges for voluntary carbon credit markets for cover cropping. Given the mixed additionality and persistence found among subgroups in this study, a carbon crediting program would struggle to guarantee both that the carbon they store is additional and that it would persist. To manage the additionality problem, the credit program might choose to discount credits based on the estimated overall level of additionality. To deal with the rerelease problem, the credit market might design some program to replace credits after carbon is released or switch to a credit-year system, which sets a ton of temporary storage as worth a fraction of a permanent ton emitted (Fearnside, Lashof, and Moura-Costa 2000; Brandão and Levasseur 2011). Any of these solutions will decrease the size of the market payment available to farmers and thus may limit the program's ability to create large-scale change.

Short-term payment programs like EQIP can avoid some of these difficulties. Mismeasuring the additionality and long-term benefits of government payments may lead to inefficient uses of funds, but it will not lead to excess greenhouse gas release as nonadditional private market credits might. In addition, government incentive programs can take into account the full range of externalities provided by a practice: cover cropping incentives have significantly improved water quality in the Chesapeake Bay (Fleming, Lichtenberg, and D. A. Newburn 2018), and the first cover cropping incentives were designed only in response to concerns around soil and water quality (Turner et al. 2014).

In addition, policymakers can use this framework of short-term and long-term costs to design the structure of repeat payment programs. EQIP's one-time contract structure can provide all needed support for changes that need adoption costs but face few if any long-term costs, such as building water management structures. However, depending on the farm, some carbon-storing practices like cover cropping and reduced tillage may produce long-term environmental benefits but not enough financial benefits for the producer to continue them on their own. In those cases, repeat contracts with smaller payments that cover these ongoing costs could have substantial additionality. This highlights a role for programs like the Conservation Stewardship Program (CSP), a USDA program that pays past conservation practice adopters to enhance or add new conservation practices while sustaining previous ones.

This paper also illustrates the need for more work in this area that controls for the application decision and that follows practices over longer periods of time. There have historically been few data sources that reliably track farmers over time, and connecting farmers to past program participation has typically relied on imperfect recall questions. Modern satellite imagery databases may make tracking practices in individual fields over time more achievable. The ProTracts database used in this project was designed primarily to track successful applications and not unsuccessful ones, which led to the differential missingness of unsuccessful applications in this paper's sample. To improve the precision of the long-term effects estimated here, I will continue to incorporate more years of outcomes as they become available. Future programs should put care into tracking unsuccessful applicants, since they are a subgroup that are likely to be more similar in terms of unobservables than any other group. In addition, they are particularly relevant when exploring the likely effects of marginal increases or decreases in program funding.

In future work, researchers can also explore how a range of nature-based solutions fit into this framework of adoption and long-term costs. Nature-based solutions could provide up to 30% of the the emissions reductions needed to meet global goals (Miles et al. 2021). They include a wide range of practice changes, including improved forestry management, reduced tillage and other agricultural changes, regenerative rangeland management, agroforestry, and shifts in nutrient management. Achieving the full potential of these changes will require tailoring incentives to a wide variety of economic and biological contexts. The adoption cost/long-term cost framework of this paper can provide a starting point for that work, ideally through estimating average adoption and long-term costs and through how much they vary within and between populations.

Chapter 3

Hold Your Horses: Market Integration, Livestock Insurance, and Fire Sales among Mongolian Herders

3.1 Introduction

The world's 217 million (Thornton et al. 2002) nomadic pastoralists dwell in some of the harshest climates on the planet, and Mongolia is no exception. Mongolian herders often face killer winters called *dzuds*, disasters where some combination of extreme cold, heavy or irregular snows, or poorly timed frost-thaw cycles lead to mass livestock death. In 2000 and 2010, *dzuds* killed more than 20% of the country's livestock, and the 2024 *dzud*'s final death toll included at least 12% of the country's livestock (Associated Press 2024). Since nomadic pastoralists store most of their wealth in and get most of their food and income from their animals, the economic consequences can be devastating. The 2010 *dzud* caused $345 million in damages (Nandintsetseg et al. 2018), and the 2024 *dzud* has destroyed more than 70% of some families' herds, a loss that may force many of them to leave herding and move to the slum districts around Ulaanbaatar for work (Associated Press 2024).

Fluctuations in the livestock market add a second disaster to *dzud* years. When winter conditions worsen, herders need more fuel, food, and fodder to keep themselves and their animals alive. All of these cost money, and few banks or individuals want to lend to herders on the edge of a winter disaster, so herders often must sell off some of their animals to save the rest of them. Unfortunately, when a large share of herders all sell at the same time, the price of livestock plummets. This low price forces herders to sell even more animals to buy their necessary supplies, compounding the wealth destruction from livestock mortality (Ikegami 2016).

In this paper I study the dynamics of these livestock fire sales, and the effect of Mongolia's index-based livestock insurance (IBLI) program on smoothing them. Under IBLI, herders can purchase protection for their animals that pays out when winter mortality for a specific livestock species in a particular district rises above 6%. The World Bank and the Mongolian government chose the index-based approach to avoid the moral hazard, selection, and monitoring cost concerns associated with implementing individual livestock insurance in one of the world's least densely populated countries (Mahul and Skees 2007). While the insurance does not send payments until late summer, it does improve access to credit earlier in the year. Limited access to credit is a common problem in herder communities, and in settings including Ethiopia loans may only be available to herders with a large enough herd size to serve as collateral (Santos and Christopher B. Barrett 2011). From the program's inception, banks agreed to offer larger loans and lower interest rates to insured herders (ibid.), and insured herders in the 2010 *dzud* were twice as likely to report receiving a loan as similar uninsured herders (Bertram-Huemmer and Kraehnert 2018). As such, insured herders were 10-15 percentage points less likely to report selling animals as a *dzud* coping mechanism (ibid).

Through reducing the glut of livestock sales and smoothing the price of livestock in disaster years, this reduction in distress sales by insured herders may provide spillover benefits to uninsured herders. This potential spillover could be particularly important because a willingness-to-pay study found that the mid- to low-wealth herders who face the greatest risk of falling into a livestock poverty trap are less likely to buy insurance (Chantarat, Andrew G Mude, and Christopher B Barrett 2009). If IBLI does increase community-wide *dzud* resilience by smoothing price fluctuations, governments and development agencies may wish to subsidize livestock insurance programs to help stabilize vulnerable communities in a changing climate.

I begin this paper by discussing the dynamics that create fire sales in Mongolia, as herders selling into an imperfectly elastic livestock market seek to make asset-protecting investments while liquidity constrained. I then empirically measure the fluctuations in the Mongolian livestock market and trace its connection to livestock mortality and market integration, key components of the fire sale dynamic. Higher mortality is associated with a large dip in livestock prices, both for the species most directly impacted by mortality and for all livestock species. I then attempt to estimate the effect of IBLI on price fluctuations in Mongolia with a difference-in-difference design around the gradual province-by-province rollout of the livestock insurance between 2006 and 2012, which produces noisy null estimates.

This speaks to several important strands of the economics literature. First, I add to the research on the impacts of index insurance on risk management and disaster recovery. Theory suggests that index-based insurance for herders may help them avoid falling below the well-documented herd size poverty trap seen in this population (Chantarat, Andrew G. Mude, et al. 2017; Lybbert, Christopher B. Barrett, et al. 2004; Christopher B. Barrett, Marenya, et al. 2006). In Kenya, IBLI for pastoralists reduced distress sales during shocks but increased sales in years without shocks, consistent with a model where uninsured herders increase herd size as insurance against risk (Jensen, Christopher B. Barrett, and Andrew G. Mude 2017). In addition, the RCT found that IBLI increased milk income per animal and decreased livestock mortality, suggesting the reduced precautionary savings were channeled into productivity-enhancing investments. Janzen and M. R. Carter 2019's study of IBLI in northern Kenya finds that IBLI reduced sales among the wealthy and smoothed consumption and slightly reduced asset sales among the poor. In the Mongolian context, Bertram-Huemmer and Kraehnert 2018's matching study finds that insured herders recovered in herd size more quickly than uninsured herders. I explore the mechanisms for a price externality from distress livestock sales in the Mongolian context, though the empirical analysis does not profice a precise estimate.

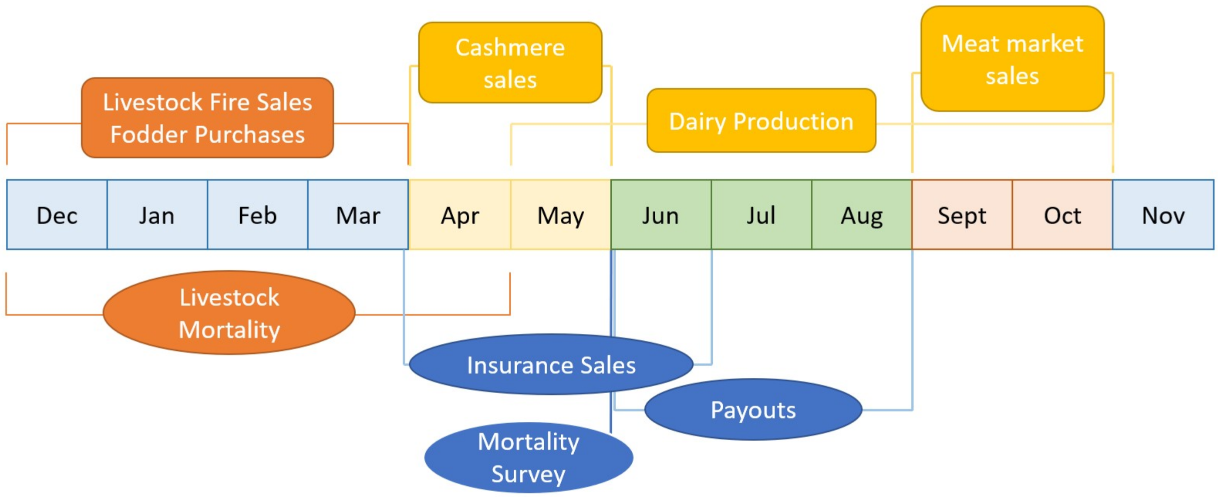
I also add to the literature exploring the role of general equilibrium price effects in amplifying or reducing the impact of poverty interventions. Cash transfers in Mexico marginally increased the price of goods for others in their villages, while in-kind transfers reduced prices by 4%, with the strongest effects in areas where the markets were most fragmented (Cunha, De Giorgi, and Jayachandran 2019). Giving Kenyan farmers credit, allowing them to delay crop sales until prices rose, produced returns of 29% and smoothed marketwide prices (Burke, Bergquist, and Miguel 2018). An asset grant in Bangladesh enabled some poor women to shift from day labor to livestock production, boosting day labor wages for women who did not receive the asset (Bandiera et al. 2017). In India, where low-income laborers see wages dip when landowners change crops in drought years (Jayachandran 2006), giving insurance to laborers decreases the volatility of wages in response to rainfall, while giving insurance to producers increases wage volatility (Mobarak and Rosenzweig 2014). This paper discusses the role that price risk plays in increasing the harm of natural disasters.

This paper also contributes to the body of knowledge exploring the degree to which livestock can serve as a "buffer stock" that households can use to smooth consumption in periods of distress. Development economists have long hypothesized that households in West Africa and other regions with mixed crop/livestock systems keep livestock primarily to sell or consume in poor rainfall years. However, most households facing a severe drought in Burkina Faso only used livestock sales to offset 30 percent or less of the income shock (Fafchamps, Udry, and Czukas 1998), and households used grain stores more than livestock to smooth consumption (Kazianga and Udry 2006). Among herders, livestock sales may be used more to smooth assets than to smooth comsumption (Lybbert and McPeak 2012), perhaps to avoid falling into an asset-based poverty trap (Lybbert, Christopher B. Barrett, et al. 2004; M. R. Carter and Lybbert 2012; Christopher B. Barrett, Marenya, et al. 2006). This paper explores how price risk and, in particular, livestock fire sales reduce the ability of households to effectively smooth consumptions through selling livestock.

This paper also offers an additional piece of the solution to the index-based insurance demand puzzle. The demand for index insurance, particularly in crop settings where insurance is tied to rainfall, often remains far lower than economic returns would lead us to expect (M. Carter et al. 2017). Basis risk has been studied extensively as a possible source for low index insurance demand (Clarke 2016), and factors like lack of understanding of insurance (Johnson et al. 2019), liquidity constraints (S. Cole et al. 2013), and behavioral salience of risk (Shawn Cole, Stein, and Tobacman 2014) may also play a role. I suggest a further factor in the correlation between price risk and index basis risk. In the herding setting, fire sales mean that price risk and base risk move in opposite directions: if one herder has a bad year while their neighbors have a good one, the herder will not be eligible for an IBLI payout, but they will at least receive the higher good-year prices for any animals they sell. For standard agricultural products like grain, the relationship moves the opposite direction: prices are likely to increase in low-production years and fall in high-production years. Therefore, price risk worsens basis risk, since someone who has a low-productivity year when other farmers do well will both get no insurance payout and face a low price for their limited output.

In this paper, I first describe the Mongolian setting, with attention to the economic course of a typical year or a *dzud* year for a Mongolian herder, and how index-based insurance fits into that pattern. I then discuss the data used in this paper in the Data section. In Section 4, I explore the market dynamics of Mongolian livestock markets, exploring the correlation of

Figure 3.1: A Mongolian Herder's Year



prices across space and their covariance with mortality shocks. Next in Section 5, I explain the difference-in-difference design used to measure the effect of IBLI on price shocks. I discuss the results in Section 6, and conclude in Section 7.

3.2 Setting

3.2.1 Mongolia's Herding Economy

Nomadic herding has been a key part of Mongolia's economy throughout its history and remains so today. Out of Mongolia's population of 3.4 million, 50% live in and around the capital of Ulaanbaatar, 20% in Mongolia's towns, and 30% in rural areas (National Statistics Office of Mongolia 2024). Since Mongolia's short growing season and cold dry climate confine agriculture to less than 1% of the country's land area (Rasmussen and Dorlig 2011), the vast majority of rural residents are nomadic herders who raise a mix of goats, sheep, cattle, yaks, horses, and camels (Shagdar 2002).

Mongolia's rural population is scattered across a vast area, with the 1.8 million people living outside of Ulaanbaatar spread across six hundred thousand square miles. Given the low population density, most herders mostly work exclusively with their family's herd and have little outside employment: herder households earn an average of 12.6% of household income from employment or self-employment outside of herding (Meurs, Amartuvshin, and Banzragch 2017). Rural herders migrate two to twelve times a year, usually within the bounds of one of Mongolia's 330 districts. Most herders return yearly to the same established pasturing sites, with occasional longer trips elsewhere in years with localized poor pasture.

A Mongolian herder's life follows an annual economic cycle depicted in Figure 3.1. The relief of spring comes in April or May as new grass starts to grow and the bitter winter cold and snow fade. Around this time, herders also harvest cashmere from their goats as they shed their winter undercoats. These cashmere sales were responsible for more than half of market income for herders in 2009 (Meurs, Amartuvshin, and Banzragch 2017) and feed into well-integrated national markets for processing and export (Kusano and Saizen 2013). By May, herders also begin to milk their animals, producing dairy products that are a key component of herder diets, though more rarely sold (Meurs, Amartuvshin, and Banzragch 2017). Animals bulk up over the summer, building a layer of fat to help sustain them through the winter months.

In autumn when animals are in peak condition, herders sell livestock into regional and national markets. After cashmere, these sales are the second most important source of money for herders (Meurs, Amartuvshin, and Banzragch 2017). Herders often sell their livestock to intermediaries in district or subdistrict centers, who then either sell the animals to national processors or sell them for local consumption (Kusano and Saizen 2013). Historically, most of this meat stayed in domestic markets, though recent investments in meat processing infrastructure has allowed exports to increase in recent years (Ooluun 2024).

3.2.2 The Structure of Winter Disasters

Mongolian winters are long and harsh. From November through April, Mongolian livestock face bitter cold and heavy snows. During this time, animals slowly lose the fat they accumulated over the summer months, since their bodies must use more energy to maintain warmth and there is little grass to eat. A winter turns into a disaster known as a *dzud* when weather conditions cause livestock mortality. A *dzud* can be triggered by a range of different conditions, such as severe cold, heavy snow, frost-thaw cycles that create a layer of ice over grass, or too little snow in the desert regions that rely on snow for water. These conditions limit an animal's ability to eat or increase the energy its body demands to stay alive. Many *dzud* years see several of those problems strike over the course of the season, and mortality often peaks in February through April when animals reach the end of their fat reserves. A *dzud* can be localized, regional, or national, depending on how the pattern of weather conditions occur.

Herders have several practices that they can use to ameliorate the effects of a *dzud*. One is to provide animals with fodder. In many regions, herders will make some hay of their own from summer grass. Since the 1999-2001 *dzud*, Mongolia's national government has maintained a State Emergency Fodder Fund, and province and district governments established their own reserves following the 2009 *dzud* (Rasmussen and Dorlig 2011). The government reserve provides fodder for purchase at the government's price of procurement, subsidizing transport costs. However, fodder production has fallen sharply after the end of the Soviet era (ibid.), and poor years for Mongolian livestock also often correspond with lower fodder productivity, making it difficult to produce adequate levels of fodder. During a more localized *dzud*, herders may also feed their animals by taking their long-legged animals (horses, cows, yaks, and camels) long distances to reserve pastures while the short-legged sheep and goats remain in their usual winter pasture.

Both of these coping strategies require finances, and formal financial access can be limited. Herders will often receive some money from preemptively selling off the weaker animals that are unlikely to make it through the winter. Mongolia does have a reasonable density of banks: by 2003, there was 1.17 bank branches per district center and 4.6 banks per province center (Brophy and Cuddy 2007). However, this proximity of bank access does not guarantee access to loans, particularly during severe winters. A 2007 analysis found that "the quality of the available collateral is often considered the most important indicator of the worth of a loan application," and the largest agricultural bank, Ag Bank, requires herders to have at least 200 animals to be eligible for even a microloan: a threshold that only the wealthiest 15% of herding households could meet (Brophy and Cuddy 2007). Further, these collateral requirements often increase during winter, especially if the weather is showing signs of dzud. Given this gap, the IBLI program can play an important role in providing financial access.

3.2.3 Index-based Livestock Insurance Program

Mongolia's index-based livestock insurance program exists to insure large covariate livestock mortality shocks. From spring through the end of June, herders may purchase insurance contracts through local banks to insure a number of animals they choose. This insurance policy will then cover animals for the following winter. In the following June a Mongolian livestock census measures livestock mortality, and in July through September herders receive payouts based on the subdistrict-level mortality of a species. If mortality for a species is above 6%, herders who insured that species get a payout equal to

$$Payout_{sdti} = Inscount_{sti} * Value_{sd} * (Mortrate_{sdt} - .06) \tag{3.1}$$

where $Inscount_{sti}$ is the number of heads of livestock of species s that herder i insured in year t, $Value_{sd}$ is the start-of-year estimated value of one head of species s in subdistrict d, and $Mortrate_{sdt}$ is the mortality rate of the species in the subdistrict in year t.

This livestock insurance program was established in 2006, in the wake of the disasterous 1999-2001 *dzud* that killed more than 20% of the country's livestock. Implemented with funding and guidance from the World Bank, the pilot program started in 2006 in three provinces chosen to test how the program would perform in Mongolia's three major biomes: steppe, mountain, and desert. In 2009, a *dzud* year that caused economic losses worth $345 billion (Nandintsetseg et al. 2018), 15-21% of herders in these three provinces had purchased livestock insurance. Sukhbaatar province also joined the program that year as planners prepared for an expansion, and 4% of herders in that province purchased 2009-10 insurance. The program was quickly rolled out in all other provinces by 2012 after the 2009 dzud refocused policymaker attention on the importance of dzud protection.

Today, the program is largely self-funding, with some support from the Mongolian government. Private banks sell the insurance policies to herders and provide the first tier of funds for payouts. These banks purchase reinsurance through MongolianRe, a national reinsurance company that then purchases reinsurance from international banks and governments. The Mongolian government significantly regulates the policy's structure and pricing, and runs the biannual livestock surveys that are used to determine policy payouts.

While this insurance program does not offer payouts until summer, the insurance policies still help smooth access to credit during the winter months when herders need to purchase fodder and fuel. Bertram-Huemmer and Kraehnert 2018 found that insurance holders in the 2010 *dzud* were twice as likely to have borrowed money as uninsured herders. This is backed up by 2020 discussions with the insuring banks: many of MongolianRe's partners will only loan money to herders in winter if they have particularly large herds, or if they have an insurance policy as collateral. The summer insurance payouts also help herders to rebuild herd numbers, letting them slaughter fewer animals by purchasing food instead. (Bertram-Huemmer and Kraehnert 2018).

3.3 Data

3.3.1 Livestock Data

The statistics on Mongolian livestock comes predominantly from the Mongolian National Statistics Office, both through the datasets available on their website 1212.mn and through special requests. The NSO provides monthly data on the prices of meat and 16 age-sex-species categories of livestock for each of Mongolia's 330 districts and 22 provinces. District-level officials collect district prices through surveying middlemen in the district center, the small town that holds a district's schools, government buildings, and shops. The province level statistical offices compile this data and collect similar data in the province center, which is generally the largest town in the province, has a range of businesses and markets, and has the best-connected roads to the capital of Ulaanbaatar. The NSO also collects annual data on livestock mortality, sales, reproduction, and population annually at the district and province level, with data avaiable from 1990 onwards.

3.3.2 Insurance Data

To track insurance, I use district and province level data from MongolianRe, the national Mongolian reinsurance company that partners with local banks to run the program. MongolianRe provided insurance purchase rates, prices, premium totals, and payouts by species and province from 2006 to 2021, and by species and district from 2009 to 2021.

3.4 The Dynamics of Mongolian Livestock Markets

How much IBLI will impact prices depends on how strongly mortality shocks covary and how well the domestic and international markets can absorb the market shocks. In this section, I first explore the covariance of mortality shocks across subdistricts, districts, and provinces. I find that prices fall in winter in years with severe mortality shocks, and these shocks covary closely among districts within provinces and somewhat across provinces. I then decompose the price risk that herders face into monthly, national, provincial, and district-level variation. I find that national variation accounts for half of the fluctuation in district livestock prices. in addition, regressions show that national livestock mortality has an effect on province prices four times as large as province mortality. Together, these indicate that my estimates of the effect of IBLI on price smoothing are a lower bound for the national equilibrium effect, since there are substantial spillovers across provinces.

Figure 3.2: Livestock Mortality Rates

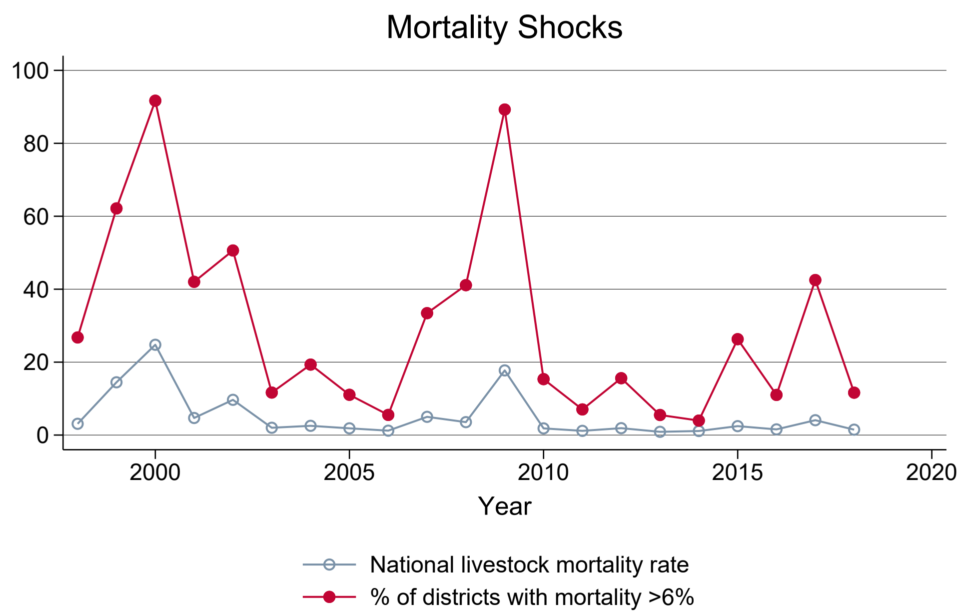


Table 3.1: Livestock Mortality Covariance by Species

Correlation	Sheep	Goat	Cattle	Horse	Camel
District Mortality/Province Mortality	.706	.707	.806	.677	.462
Province Mortality/National Mortality	.754	.754	.772	.754	.745
District Mortality /National Mortality	.501	.499	.633	.452	.344

3.4.1 Winter Mortality Shocks

Although almost every year some regions experience more localized severe winters, Mongolia has experienced 3 large national *dzuds* between 2000 and 2020. Figure 3.2 shows the overall mortality rate of livestock and the share of districts that had 6% or higher mortality for at least one species. The spike in 1999 to 2001 shows the *dzud* that sparked the establishment of the IBLI insurance program: if the program had been in place, 90% of districts would have been eligible for payouts in 2001. The 2009-10 *dzud* then killed 23% of the country's livestock. 2019 also qualified as a regional *dzud*, with 40% of districts above 6% mortality. However, at least 3% of districts experience high mortality for any year in the sample, with 17% of districts reaching the payout threshold in the median year. Livestock mortality shocks covary strongly within provinces. Table 3.1 shows that district mortality and province mortality correlate by coefficients of approximately .7 for all species except camels. Province mortality correlates with national mortality by approximately .75. Together, these indicate that many livestock mortality shocks are national, but districts and provinces do also experience smaller regional shocks.

3.4.2 Market Structure and Integration

This section explores the level of integration in Mongolian livestock markets. Imperfect market integration is the reason why herders experience such local fluctuations in livestock prices in response to local mortality. Mongolia makes up a tiny share of the global market for meat. If Mongolian herders could easily export their excess animals during a fire sale and receive the global price, even the peak volume of Mongolian distress sales could not move the global price in any significant fashion. Similarly, if markets were perfectly integrated nationally, the national market could at least slightly absorb and smooth the price effect of distress sales in the case of a regional *dzud*. Strong national integration in the livestock market would also shrink my estimated effect of insurance, since I compare the magnitude of the price shocks across more and less insured provinces.

There is not data available on the magnitude of sales across provinces, so I focus instead on identifying integration through price shocks. Following a variation on the method in Christopher B. Barrett and Luseno 2004, I decompose the livestock price in district d, province c, time t, and month of year m into five components:

$$p_{dt} = \bar{p} + (\bar{p}_m - \bar{p}) + (p_t - \bar{p}_m) + (p_{ct} - p_t) + (p_{dt} - p_{ct}) \tag{3.2}$$

$$p_{dt} = \bar{p} + M + N + C + D \tag{3.3}$$

where $\bar{p}$ is the average price in the central Ulaanbaatar market. $M = \bar{p}_m - \bar{p}$ is the monthly average price's variation from the annual average price, and $N = p_t - \bar{p}_m$ is the national market shock experienced in a given month. $C = p_{ct} - p_t$ is the province center's price difference relative to the national market, and $D = p_{dt} - p_{ct}$ is the district-specific price difference.

Plotting the mean values of C and D reveals that there are substantial markups for livestock in the capital compared to the rural districts. Shown in Figure 3.3, district prices for goats are on average 8% higher in provinces than in districts, and 14% higher in Ulaanbaatar than in districts. The Ulaanbaatar premium is higher than 11% for all species other than horses, which are less useful for transportation in the capital than in rural areas. This markup reveals some of the transport frictions in moving livestock to market, causing higher prices in the centers where they are consumed compared to the rural centers where they are produced.

To turn Equation 3.3 into a decomposition of risk, we can then take the variance. This produces

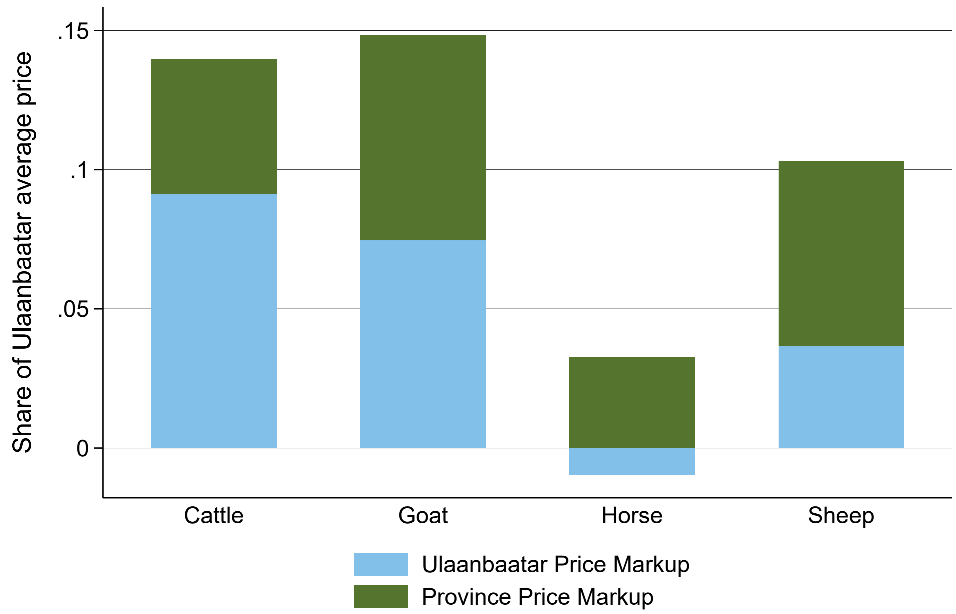
$$\begin{aligned} Var(p_{dt}) =& Var(M) + Var(N) + Var(C) + Var(D) + 2(Cov(M,N) + Cov(M,C) \\ &+ Cov(M,D) + Cov(N,C) + Cov(N,D) + Cov(C,D)) \end{aligned} \tag{3.4}$$

which can be broken down into four components: the variation from annual monthly fluctuations $MRisk$, the share from national shocks $NRisk$, the effect of province centers $CRisk$, and the variation of district-level shocks $DRisk$. For each of these components,

$$XRisk = Cov(X,M) + Cov(X,N) + Cov(X,C) + Cov(X,D) \tag{3.5}$$

To calculate the share of risk from each source, we can plot $XRisk/Var(p_{dt})$. This decomposition of risk is graphed in Figure 3.4. National and province-level price fluctuations drive

Figure 3.3: District Price Differentials



80% of variation in prices, while 20% of price risk comes from district variations. Monthly fixed effects play almost no role in the risk exposure of Mongolian herders.

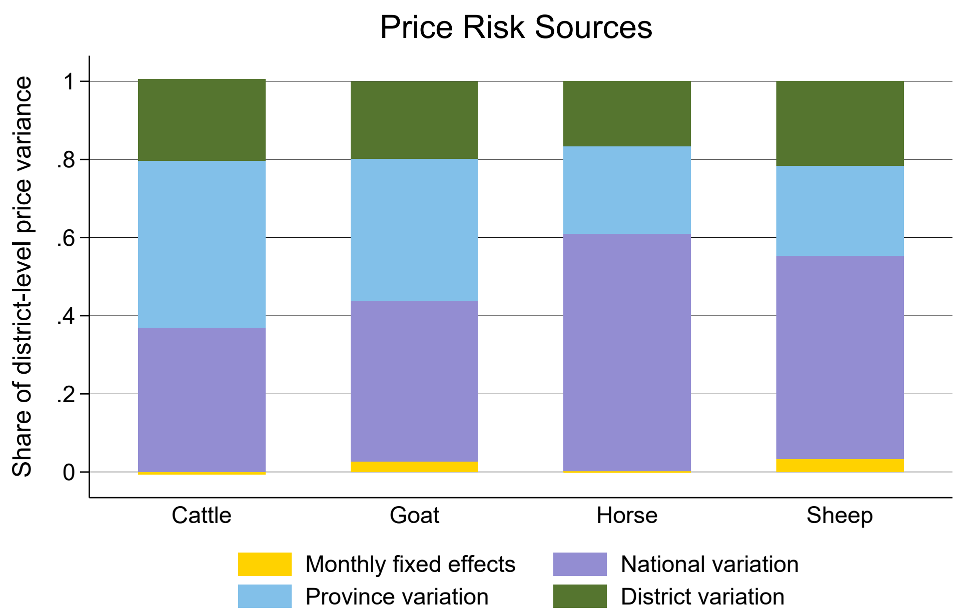
The large share of variation tied to national fluctuations stems from two sources: integration between national and local markets, and correlation in shocks between national and province markets. The correlations in Table 3.1 show that the correlation between district and national mortality rates is approximately .5, similar to the share of district price risk stemming from national variation. Similarly, the correlation between district and province mortality is approximately .7, and 80% of district price risk comes from district and national sources. This suggests that correlation in environmental shocks explains most of the correlation in price shocks.

3.4.3 Prices and Winter

Prices fluctuate across districts, provinces, and time. I next explore how much of this fluctuation is tied to winter livestock mortality. Both species-specific and overall livestock mortality depress the price of livestock for all species aside from horses.

Table 3.2 shows the result of a fixed-effect regression testing the effect of winter, livestock mortality, and their interaction on livestock prices. Winter snow and cold increases the transportation costs of getting livestock to market, lowering livestock prices in rural districts. Winter prices decrease further with both species-specific and overall mortality. The effect of overall mortality indicates the presence of a fire sales dynamic: the lowered price of cattle in a difficult year for cattle might only reflect the reduced value of an animal that isn't expected to survive, but selling across species in a difficult year for cattle is consistent with a "fire sale" mechanic where herders increase all sales to make necessary investments. Goats are

Figure 3.4: A Decomposition of Sources of Risk



the odd exception: their prices increase with total livestock mortality, though they decrease as normal with goat mortality. This may reflect that goats provide a large spring influx of money through cashmere sales, well-timed to help herders begin to rebuild from a difficult winter. Regional and national mortality largely overshadow the effect of local mortality. Part of this is because district mortality covaries strongly with mortality across the province and the country. District mortality and province mortality have a covariance of approximately .7 for most species, as shown in Table 3.3. Provinces similarly covary by approximately .75 with national mortality, so districts and national mortality covary by approximately .5.

This spatial covariance in mortality means that the coefficients in Table 3.2 reflect both the effect of local and regional mortality. In Table 3.4, including the province and national mortality rates sharply diminishes the effect of local mortality. To avoid problems of collinearity, the effects of overall and species-specific mortality shocks are measured through two separate regressions, with overall shocks in Panel A and species shocks in Panel B. The effect of local mortality in both cases remains negative but is only significant for species-specific sheep mortality. Province mortality has twice as large an effect as local mortality for most species, reflecting substantial province-level market integration.

The price effects of national mortality in Table 3.4 are larger still, showing the pull of the national Ulaanbaatar market. High national mortality sharply reduces livestock prices, even controlling for local mortality. This reflects that fire sale animals flood the Ulaanbaatar final market. The strength of this national effect means that successfully slowing fire sales in any region will help smooth the national price of livestock. Therefore, the empirical method presented in the following section, which compares effects across districts and provinces, will estimate a lower bound for the impact of insurance, with the magnitude reduced by spillovers in the somewhat integrated national market.

Table 3.2: Price Fluctuations in Response to Local Mortality Shocks

VARIABLES	(1) Sheep Log Price	(2) Goat Log Price	(3) Cattle Log Price	(4) Horse Log Price	(5) Camel Log Price
Total Livestock Mortality	-0.875***	-0.257***	0.658***	-1.326***	-0.986***
	(0.0439)	(0.0444)	(0.0682)	(0.0460)	(0.0243)
Winter	-0.0308***	-0.0162***	-0.00377**	0.000803	0.00148
	(0.00177)	(0.00183)	(0.00178)	(0.00181)	(0.00234)
Winter * Total Livestock Mortality	0.0489	0.121**	-0.214**	-0.151**	-0.115***
	(0.0601)	(0.0611)	(0.0872)	(0.0616)	(0.0318)
Species Mortality	-0.627***	-1.052***	-1.503***	0.0802	0.000699
	(0.0407)	(0.0388)	(0.0532)	(0.0490)	(0.0629)
Winter * Species Mortality	-0.574***	-0.613***	-0.0213	-0.0601	-0.158*
	(0.0556)	(0.0536)	(0.0686)	(0.0661)	(0.0829)
Date	-0.00236***	-0.00227***	-0.000428***	0.00118***	0.000441***
	(2.53e-05)	(2.60e-05)	(2.53e-05)	(2.58e-05)	(3.47e-05)
Constant	11.18***	10.79***	12.67***	12.27***	12.63***
	(0.00476)	(0.00489)	(0.00475)	(0.00483)	(0.00631)
Observations	163,025	150,760	138,932	136,078	71,026
R^2	0.142	0.133	0.102	0.132	0.088
Number of xtid	2,239	2,230	2,149	2,137	1,745

Standard errors in parentheses

*** p<0.01, ** p<0.05, * p<0.1

Table 3.3: Regional and National Correlation in Species Mortality

Correlation	Sheep	Goat	Cattle	Horse	Camel
District Mortality/Province Mortality	.706	.707	.806	.677	.462
Province Mortality/National Mortality	.754	.754	.772	.754	.745
District Mortality /National Mortality	.501	.499	.633	.452	.344

Table 3.4: Effects of Local, Province, and National Livestock Mortality Rates

Panel A: Overall Mortality Shocks

VARIABLES	(1) Sheep Log Price	(2) Goat Log Price	(3) Cattle Log Price	(4) Horse Log Price	(5) Camel Log Price
Local Mortality	-0.184*	-0.128	-0.122	-0.185*	-0.128
	(0.103)	(0.103)	(0.104)	(0.102)	(0.135)
Province Mortality	-0.436***	-0.123	-0.639***	-0.403***	-0.288
	(0.152)	(0.170)	(0.161)	(0.146)	(0.201)
National Mortality	-3.611***	-3.636***	-2.285***	-2.457***	-1.877***
	(0.154)	(0.173)	(0.156)	(0.160)	(0.209)
Date	-0.00360***	-0.00368***	-0.00118***	0.000210	-0.000365
	(0.000155)	(0.000189)	(0.000180)	(0.000174)	(0.000291)
Constant	11.27***	10.85***	12.80***	12.40***	12.71***
	(0.0290)	(0.0347)	(0.0328)	(0.0320)	(0.0526)
Observations	25,706	22,871	23,031	22,374	12,709
R^2	0.260	0.228	0.189	0.221	0.140
Number of xtid	319	319	315	316	265

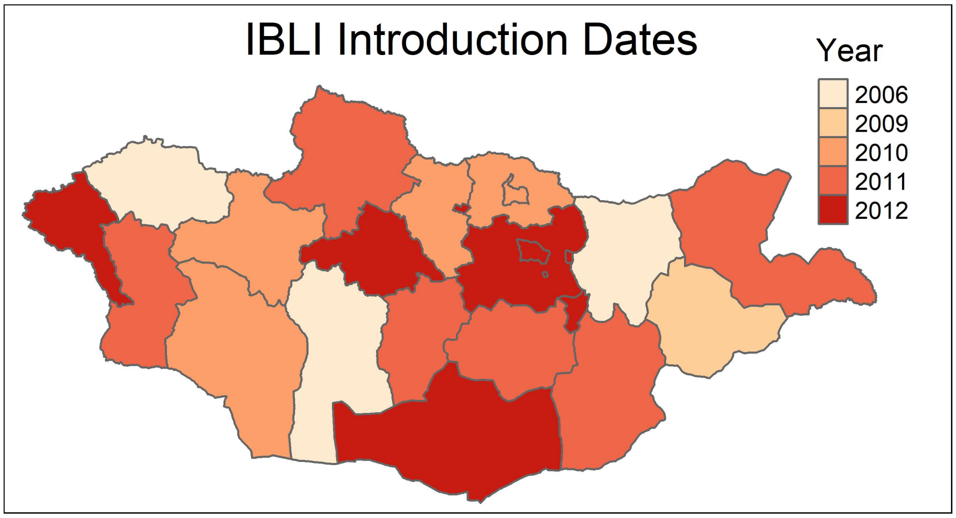
Panel B: Species-Specific Mortality Shocks

VARIABLES	(1) Sheep Log Price	(2) Goat Log Price	(3) Cattle Log Price	(4) Horse Log Price	(5) Camel Log Price
Local Mortality	-0.236***	-0.135	-0.145	-0.228**	0.162
	(0.0900)	(0.0893)	(0.0887)	(0.104)	(0.274)
Province Mortality	-0.444***	-0.330**	-0.545***	-0.402***	-2.098***
	(0.129)	(0.150)	(0.119)	(0.142)	(0.668)
National Mortality	-2.586***	-2.210***	-1.644***	-2.508***	-4.375***
	(0.121)	(0.128)	(0.108)	(0.143)	(0.593)
Date	-0.00380***	-0.00378***	-0.00118***	0.000332*	1.13e-05
	(0.000158)	(0.000187)	(0.000183)	(0.000175)	(0.000329)
Constant	11.28***	10.85***	12.80***	12.37***	12.64***
	(0.0293)	(0.0342)	(0.0334)	(0.0320)	(0.0583)
Observations	25,511	22,666	22,778	22,045	10,694
R^2	0.243	0.222	0.190	0.216	0.140
Number of xtid	319	319	315	316	256

Robust standard errors in parentheses

*** p<0.01, ** p<0.05, * p<0.1

Figure 3.5: IBLI Rollout Dates



3.5 Methodology

To investigate the effect of IBLI on fire sale price shocks, I implement an IV approach instrumenting for province-level insurance takeup with the date of insurance introduction to a province.

The two-stage least squares estimator I use is

$$p_{at} = \beta_a + \beta_t + \beta_1 \hat{Ins}_{ay} + \beta_2 Mort_{ay} + \beta_3 Mort_{ay} * \hat{Ins}_{ay} + e_{at} \tag{3.6}$$

where p_{at} is the price of an adult female animal in province a in month t, Ins_{ay} is the share of households purchasing insurance in year y, and $Mort_{ay}$ is the herd-share-weighted annual mortality rate. I use the price of adult female animals as the outcome because they are the main asset of most herds, and have the most completeness of any price series in the HSES dataset. They are also more easily valued as a standard unit than adult male animals, since a high-quality male can be used for stud on dozens of female animals and therefore has a more quality-variant price. For mortality, I use total livestock mortality as a share of herd value. Standard errors are clustered at the province level. I instrument for Ins_{ay} with the first-stage regression

$$\hat{Ins}_{ay} = \alpha_a + \alpha_y + \alpha_1 InsYears_{ay} + \alpha_{IntroYear} InsAvail_{ay} * IntroYear_a + \epsilon_a y \tag{3.7}$$

where $InsYears_{ay}$ is the number of years since IBLI's introduction in province a, and $InsAvail_{ay} * IntroYear_a$ is a fixed post-introduction effect for each wave of introductions. I structure the instrument as such because the staggered IBLI rollout was made in a balanced series of waves and the early waves had a differential effect on insurance uptake compared to later waves. The pilot IBLI program was tested in three provinces in 2006, and then rolled out in four further waves between 2009 and 2012 as depicted in Figure 3.5.

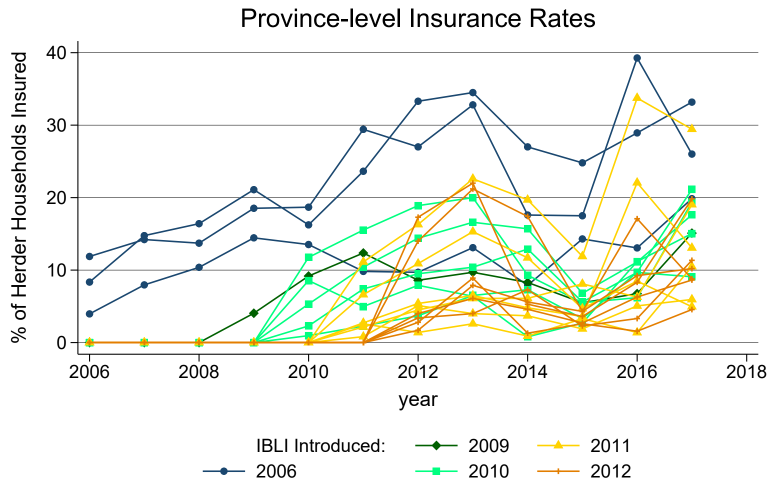
Table 3.5: Summary Statistics of Pilot and Non-pilot Provinces

Variable	(1) Pilot	(2) Non-pilot	(3) Difference
share of population in rural areas, 2006	0.692	0.668	-0.024
	(0.088)	(0.136)	(0.054)
GDP per capita, 2006	1,145.200	794.507	-350.693
	(890.705)	(200.706)	(411.164)
Distance from province to capital, km	601.370	542.742	-58.628
	(357.477)	(356.140)	(188.680)
Average herd size 2006, STUs	303.965	265.320	-38.646
	(92.058)	(55.099)	(44.576)
Average livestock losses, 2000 - 08	0.089	0.083	-0.006
	(0.034)	(0.030)	(0.017)
sheep share of livestock, 2006 SFUs	0.246	0.255	0.009
	(0.065)	(0.043)	(0.032)
goat share of livestock, 2006 SFUs	0.235	0.258	0.024
	(0.096)	(0.094)	(0.050)
cattle share of livestock, 2006 SFUs	0.233	0.197	-0.036
	(0.022)	(0.099)	(0.028)
horse share of livestock, 2006	0.069	0.062	-0.007
	(0.037)	(0.026)	(0.018)
Observations	4	15	19

The initial provinces were not selected at random, but they were selected to be representative of the country. The three provinces include one from the desert region, one from the steppe, and one in the mountains, representing the three major biomes of Mongolia. This ensured the program could be tested for suitability in Mongolia's full range of herding conditions, and it also reduced the risk of a simultaneous shock across all pilot sites (Mahul and Skees 2007). These pilot provinces look much like the country as a whole. Table 3.5 compares pilot and non-pilot provinces. The pilots have a similar share of the population in rural areas to other provinces, are slightly but not significantly further from Ulaanbaatar, and have a similar herd size, historic loss rate, and mix of livestock species to the rest of the country. GDP per capita is somewhat but not significantly higher in the pilot provinces. While selection into the pilot wave is not random, it seems largely uncorrelated with variables that could independently effect the magnitude of winter price shocks.

I include the separate effects of introduction for each wave because the earlier waves have had sustained higher insurance rates. According to program officials, those early provinces received heavier investment in advertising, from radio to presentations to local government collaborations (Mahul and Skees 2007). This early push encouraged a quick expansion of the insurance program in these districts, depicted in Figure 3.6. In addition, exposure to the 2010 *dzud* payouts increased trust in and popularity of the insurance in early-rollout provinces. The 2010 *dzud* is the worst disaster to strike Mongolia since the introduction of the IBLI program. Shown in figure 3.2, the 2009-10 *dzud* killed more than 10 million animals,

Figure 3.6: Province Insurance Rates Over Time



more than 20% of the national herd value, and affected every area of the country. 90% of districts lost more than 6% of at least one species, enough to trigger payouts under IBLI. This triggered substantial payouts for insured herders. According to Mongolian Re, herders received 6.6 times as much in payouts in 2010 as they spent on premiums, with over 1.8 billion MNT (1.2 million 2009 $USD) in total payouts to the 5,628 insured herders. Most herders who received insurance payouts considered themselves very satisfied, and continued to buy insurance at high rates in the following years (Bertram-Huemmer and Kraehnert 2018). As such, places with pre-2010 IBLI have continued to purchase IBLI at higher rates than other areas later introduced to the program.

This regression's central limitation is that there are market spillovers across space in Mongolia, as discussed in the previous section. If markets were perfectly integrated, a change in behavior by insured herders in one province would equally impact the price in every province, and so my regression would detect no effect at all. Table 4 suggests that province shocks impact district markets by about 10 to 25% as much as national shocks, suggesting that markets are reasonably but imperfectly integrated. I compensate for this through a couple methods: first, I run winter-only versions of my regressions, focusing on the period where Mongolia's winter weather creates larger barriers to national sales. I also test for effects specifically in more remote districts that face higher trading costs, and I compare the effects pre-2015 to post-2015 to account for the country's improved roads and transit systems after heavy investments in the early 2010s. However, my estimates still should be thought of as a lower bound, since I cannot account for the nationwide smoothing effect created by IBLI.

3.6 The Effect of Livestock Insurance on Prices

To begin, I discuss the first stage results of the effect of IBLI introduction waves on insurance rates. Shown in Table 3.6, the introduction of IBLI immediately led to substantial insurance takeup across all introduction waves. Column 1 shows the effects of each wave without

Table 3.6: First Stage Effects of Insurance Introduction on Insurance Takeup

VARIABLES	(1) Insurance Uptake	(2) Insurance Uptake (Time-Variant)
IBLI Introduced	0.0788***	0.0429**
	(0.0153)	(0.01831)
IBLI Introduced # 2006 Wave	0.112***	0.012**
	(0.0357)	(0.0324)
IBLI Introduced # 2009 Wave	0.00954	-0.00585
	(0.0153)	(0.0152)
IBLI Introduced # 2010 Wave	0.0171	0.00687
	(0.0197)	(0.0200)
IBLI Introduced # 2011 Wave	0.00857	0.00344
	(0.0288)	(0.0286)
Years Available		0.0103***
		(0.00225)
Observations	163	163
R^2	0.280	0.375

Robust standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.1

including a time-variant "years of exposure" variable, while Column 2 shows the preferred specification including a linear time effect. While the 2009-2012 waves induced similar uptake patterns to one another with 8-10% insurance coverage, the 2006 pilot wave had more than double the takeup effect of the other insurance waves. This group both received the largest advertising campaign and had widespread coverage at the time of the 2009-10 dzud, so this higher uptake among this group is expected. Insurance uptake also trends upward over time with a highly significant 1 percentage point increase per year of exposure to insurance as consumers and local banks gain familiarity with the product, so I include this trend in my regressions.

The full instrumental variables regression finds that winter dramatically reduces livestock prices, but leaves a wide interval of possible effects of IBLI on price shocks. Shown in Table 3.7, mortality is significantly associated with decreased winter prices for all outcome species[1]. In absence of insurance, a percentage point increase in mortality is associated with a 1.3 to 1.8% decline in livestock prices. Since mortality in a *dzud* year is often 10 percentage points

[1]Camels are omitted because of insufficient sample size, since they are raised in significant numbers only in a handful of provinces in the Gobi Desert.

Table 3.7: Price Response to Mortality and Insurance: Winter Only

VARIABLES	(1) (1) Sheep Log Price	(2) (2) Goat Log Price	(3) (3) Cattle Log Price	(4) (4) Horse Log Price
Insurance Uptake	1.204***	1.169***	-0.627	-0.880
	(0.451)	(0.453)	(0.399)	(0.592)
Insurance * Mortality	-1.365	-0.738	1.225	2.862*
	(1.742)	(1.857)	(1.460)	(1.566)
Mortality Rate	-1.534***	-1.309***	-1.493***	-1.773***
	(0.269)	(0.278)	(0.215)	(0.256)
Constant	10.48***	10.06***	12.58***	12.44***
	(0.0273)	(0.0274)	(0.0234)	(0.0343)
Observations	24,933	22,153	22,339	21,693
Number of xtid	323	323	318	320
Mean Price	39240	25943	285510	242633

Robust standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.1

or more above average, this suggests winter prices fall by at least 13 to 18% in a *dzud*. While these shocks are real, insurance has no statistically significant effect on smoothing them. For all species, the 95% confidence interval for the insurance and mortality interaction includes the null, a substantial negative effect, and a positive effect at least as large as the mortality coefficient. This analysis cannot confirm or reject that the insurance is effective at smoothing prices.

Mortality price shocks affect both winter and summer prices, though insurance does not seem to smooth either substantially. Table 3.8 shows the results of including seasonal interactions, with summer as the omitted baseline. The mortality coefficient is negative and substantial for all species, showing that mortality reduces year-round prices as well as winter prices. Insurance does even less to smooth price shocks in the summer. The seasonal results in Table 3.7 show that in summer, the price effects of mortality are weaker and not statistically signficant except the negative effect for cattle, and the effect of IBLI on summer price volatility is correspondingly weaker. This fits with the cycle of Mongolian livestock sales. Distress sales typically happen in fall and winter when the first signs of a winter disaster appear, so the sales that may be influenced by insurance happen most in those times. In addition, transporting and selling animals is easier when winter ends and snow melts, so the market can more easily absorb any sales shocks that do happen in spring or summer.

Insurance does seem to reduce livestock offtake in high-mortality years, though the effect is not statistically significant. Total livestock offtake is the combination of animals sold

Table 3.8: Heterogeneous Treatment Effects: Time of Year

	(1)	(2)	(3)	(4)
	(1)	(2)	(3)	(4)
	Sheep	Goat	Cattle	Horse
VARIABLES	Log Price	Log Price	Log Price	Log Price
Insurance Rate	1.225***	1.199***	-0.586	-0.829
	(0.452)	(0.453)	(0.383)	(0.581)
Winter * Insurance * Mortality	-1.916	-2.669*	-1.853	-1.410
	(1.435)	(1.582)	(1.206)	(1.533)
Insurance * Mortality	-1.339	-0.882	1.197	2.623*
	(1.393)	(1.440)	(1.149)	(1.530)
Mortality Rate	-1.308***	-1.065***	-1.316***	-1.632***
	(0.234)	(0.230)	(0.218)	(0.280)
Winter	-0.0344***	-0.0159**	0.00233	0.00209
	(0.00947)	(0.00785)	(0.00570)	(0.00440)
Winter * Mortality	-0.376**	-0.358**	-0.279***	-0.204*
	(0.182)	(0.169)	(0.0553)	(0.105)
Constant	10.50***	10.07***	12.58***	12.44***
	(0.0276)	(0.0262)	(0.0235)	(0.0349)
Observations	24,933	22,153	22,339	21,693
Number of xtid	323	323	318	320
Mean Price	39240	25943	285510	242633

Robust standard errors in parentheses

*** $p<0.01$, ** $p<0.05$, * $p<0.1$

Table 3.9: Livestock Offtake Response to Mortality and Insurance

	(1)	(2)	(3)	(4)
	(1)	(2)	(3)	(4)
VARIABLES	Sheep Offtake	Goat Offtake	Cattle Offtake	Horse Offtake
Insurance Uptake	-0.0296	-0.0324	-0.0731	-0.129**
	(0.103)	(0.111)	(0.118)	(0.0593)
Insurance * Mortality	-0.813	-1.239	-0.988	-0.279
	(0.822)	(0.883)	(0.953)	(0.477)
Mortality Rate	-0.0388	-0.258***	0.0839	0.136***
	(0.0641)	(0.0688)	(0.0875)	(0.0438)
Constant	0.259***	0.254***	0.178***	0.101***
	(0.00979)	(0.0105)	(0.0113)	(0.00568)
Observations	190	190	189	189
Number of t	10	10	10	10
Mean	0.253	0.236	0.172	0.0947

Standard errors in parentheses
*** $p<0.01$, ** $p<0.05$, * $p<0.1$

and animals slaughtered for household consumption. This is available annually and at the province level, so the sample size for the regression is comparatively small, and the estimates are noisy. However, the coefficient for the interaction of insurance and mortality in Table 3.9 is negative for all species, meaning that insured provinces see lower offtake rates in high mortality years.

To test for robustness, I also run the regressions with using share of households purchasing insurance as the measure of insurance takeup. Using this measure of insurance takeup, I find similarly ambiguous results on the effect of insurance on price shocks, with results in Table 3.10.

3.7 Conclusion

Today, weather shocks are worsening around the globe as climate change makes wild deviations from past precipitation and temperature patterns more common (IPCC 2023). As communities look to minimize the damage from these natural disasters, market effects can either help to smooth these shocks or make weathering them more difficult. In the case of Mongolia, livestock fire sales corresponding with mortality shocks add another source of pain in already difficult years.

Since IBLI cannot smooth these shocks on its own, a range of supports could assist in

Table 3.10: Price Response to Mortality and Insurance: Share of Households Insured

VARIABLES	(1) Sheep Log Price	(2) Goat Log Price	(3) Cattle Log Price	(4) Horse Log Price
Insurance Uptake	-3.466***	-3.632***	-0.752	0.943**
	(0.637)	(0.742)	(0.576)	(0.429)
Insurance * Mortality	1.040	1.955	-5.334**	-3.359
	(2.870)	(3.232)	(2.483)	(3.541)
Mortality	-2.000***	-1.795***	-1.438***	-1.489***
	(0.246)	(0.227)	(0.170)	(0.191)
Constant	10.76***	10.34***	12.60***	12.34***
	(0.0360)	(0.0394)	(0.0316)	(0.0224)
Observations	24,694	22,003	22,133	21,505
Number of xtid	314	314	309	311
Mean Price	39207	25995	285604	242575

Robust standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.1

smoothing this kind of fire sale. Improved market connections can help broader national and international markets absorb local supply shocks. Access to finance lets herders make the investments they need to endure a difficult winter without selling healthy animals. This paper does not find a statistically significant effect of insurance on winter price shocks, substantially due to the strength of market integration across provinces and due to the limited number of province units. In future work on insurance, I hope to explore new instruments for district-level insurance introduction, such as the presence of branches of banks that are more or less aggressive in marketing insurance. Also, research could compare the effects of insurance as infrastructure is developed. When new roads or new export markets open, localized markets should become less responsive to local conditions and more tied to international prices, changing the sources of price risk. To what extent does this change the value of insurance?

A major limitation of this work is that I could not track livestock flows, and that provides an opportunity for future work. With rising access to smartphones and the extensive work Mongolia has put into modernizing their meat supply chain, the country has recently set up a livestock tracking system that all animals must be entered into to be sold outside of their province. As this system gathers data, it may later provide an opportunity to understand exactly to what degree Mongolia's national livestock market absorbs supply shocks from rural areas, and to better identify the effects that IBLI has on Mongolia's livestock markets.

Appendix A

appendix

Appendices

A.1 Land Characteristic Data

A.1.1 Land Characteristic Data

Use Value: Estimates of agricultural land use value come from Virginia's Use-Value Assessment Program. The government-sponsored program housed at Virginia Tech provides estimates of the per-acre present daily value of agricultural land, using data from USDA farm surveys to estimate per-acre costs and profits. I use the annual estimates of the average use value by county. This serves as a proxy for the value of land with an easement, since agricultural land with an easement typically cannot be developed and thus is only valuable as an agricultural input.

Local Income: Local median incomes and racial demographic information is from the 2000 census, and is calculated at the census tract level.

Land Use: Land use data comes from the National Land Cover Database (NLCD). A satellite-based measure of land use, the NLCD's 30-meter resolution level allows detection of even small changes in development or land cover. NLCD sorts land into more than 30 kinds of habitat and human land use. For this analysis, I simplify these categories into developed land (buildings and impervious surfaces), agricultural land (agricultural fields and pasture), and natural land (all natural habitats). I also examine forests and wetlands as subsets of natural land.

Weather: For weather data I use Wolfram Schlenker's Daily Weather Data for Contiguous United States, calculating heating and cooling degree days from 65 degrees and days over and below each 10 degree bin. Estimates of daily high and low temperatures and daily precipitation for each day from 1970 to 2000 are made at a 2.5 mile resolution, interpolating data from the nearest PRISM weather stations. [1]

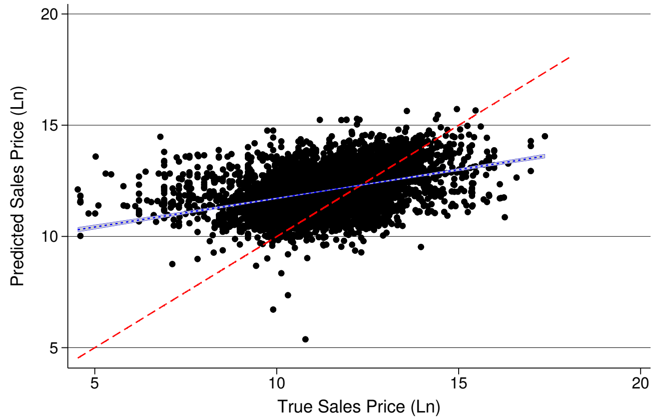
Soil quality: I use soil quality data from the USDA's Digital General Soil Map of the United States, or STATSGO2, which maps soil classification units across the United States. I use the soil survey's land capability classification system for soils, which offers a 0-100 scale of the land's suitability for agriculture. The state of Virginia has 67 soil classification units.

A.2 Estimating Sales Value

Since the sales value of a parcel is its fair market price, I use the ZTRANS and ZASMT real estate datasets from Zillow to estimate the sales value of land in Virginia. The ZTRANS dataset compiles data on land transactions from county records nationwide, including a parcel's location, size, and transfer type. The ZASMT dataset of real estate assessments provides details on what kind of improvements or buildings exist on each land parcel. It also offers some information on zoning restrictions. To build a dataset of land similar to what might be placed under an easement, I identify 25,099 transactions between 1998 and 2006 of undeveloped Virginia land outside of commercial and high-density housing zoning areas. I omit parcels with buildings to ensure my model values undeveloped land, and I omit areas zoned for high-density housing or commercial usage since easements are extremely rare in

[1]Data is available at http://www.columbia.edu/ ws2162/links.html

Figure A.1: Sales Value Prediction Accuracy



those areas. This leaves me with a sample of 24,237 land sales, of which I set a quarter aside as a validation set.

I then use a ridge regression on the remaining 17,866 parcels to estimate the price per acre of a parcel. The linear elastic net immproves the predictive accuracy of the model by reducing overfitting bias by imposing penalty terms for larger coefficients. The λ coefficient is chosen by cross-validating across the grid and choosing the coefficients that minimize prediction error. I then calculate the price per acre of a parcel as $\hat{p}_i = \hat{\beta} X_i$, where

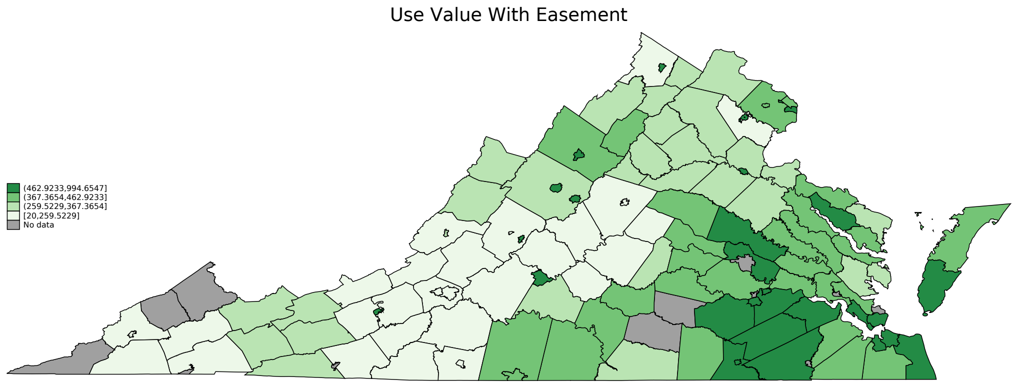
$$\hat{\beta} = argmin_{\beta} ||p - \beta X||^2 + \alpha ||\beta||^2$$

The variables in this regression cover a range of physical, economic, and environmental variables. They were derived from the ZTRAX data and from using the spatial coordinates given in the ZTRAX dataset to other data sources. These countrols are lot size, county, year, land use, flood risk, DCR-estimated development pressure, current land use, agricultural soil productivity, and weather. The weather variables are annual precipitation, heating degree days, and cooling degree days. The resulting model's performance on the ZTRAX test set is shown in figure A.1.

A.3 Estimating Use Value

To estimate the use value of parcels, I draw on the use value estimates for agricultural and forested land created by Virginia's Use-Value Assessment Program. Many of Virginia's counties tax undeveloped land based on its use value, essentially treating land as though the land was under easement. The Use-Value Assessment Program, implemented by Virginia Tech economists with funding and oversight from the state government, creates annual estimates of the per-acre use value of forestal and agricultural land by county. These estimates are then used as guidelines by local property tax assesssors. As such, they make a reasonable estimate for the way an assessor might determine the post-easement value of agricultural or forested land.

Figure A.2: Estimated Use Values Per Acre



The program creates these estimates by estimating the capitalized present day value of the future revenue stream that could be expected from a land use. For the agricultural estimates, they use farm-level agricultural production data from the USDA's Census of Agriculture and state-wide data on the prices of agricultural inputs and outputs to estimate the per-acre profits of agricultural land in each county, and they use similar data on timber production to estimate forest profits. They accommodate differences in climate and transportation costs across the state by producing separate estimates for each county, and they produce estimates for fair, good, and high quality land. These estimates are redone annually to reflect changes in prices of agricultural products and inputs. 76 of 95 Virginia's 95 counties have use value taxation and thus have estimates through the Use-Value Assessment Program.

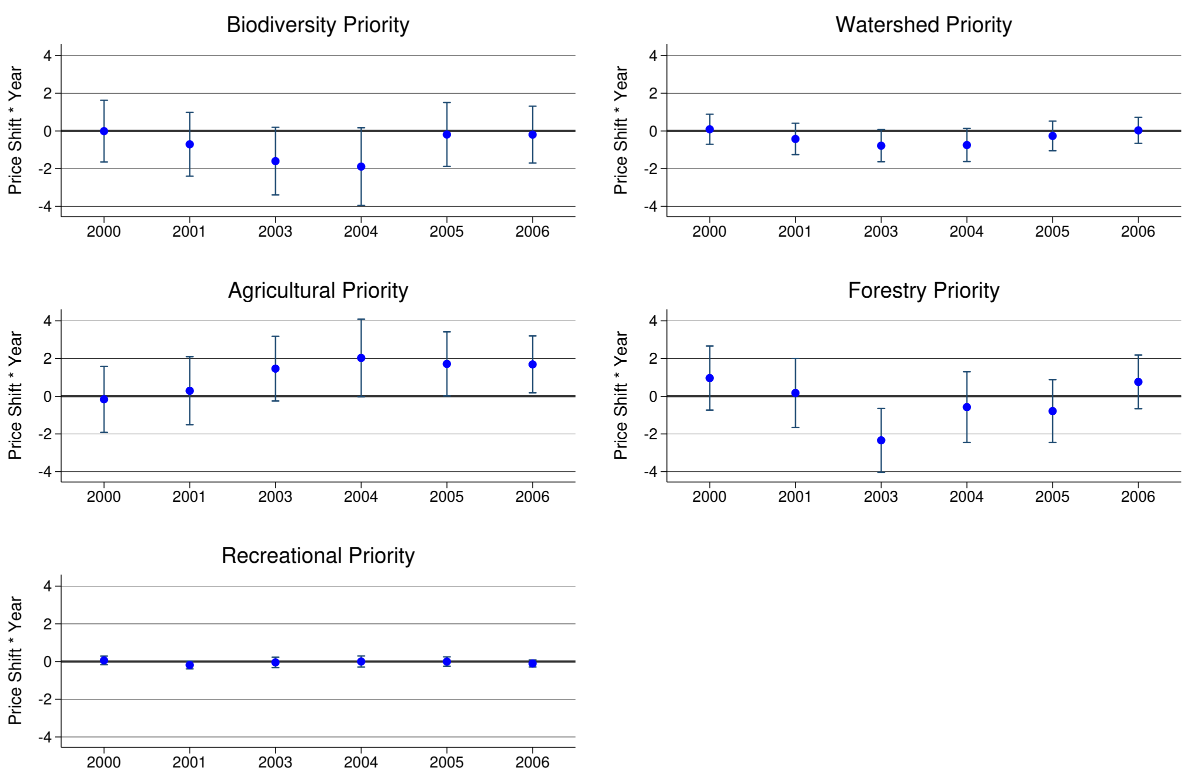
To apply these estimates to the sample of private conservation parcels, I calculate the total use value for parcel i donated in county c at time t as

$$usevalue_{ict} = acres_i * (\sum_{q=1}^{3} agvalue_{ct}^{q} * agshare_i^q + \sum_{q=1}^{3} forestvalue_{ct}^{q} * forestshare_i^q$$

where $acres$ is a parcel's total acreage, $agvalue_{ct}^{q}$ is the agricultural use value of land quality q in county c in time t, and $agshare_i^q$ is the share of parcel i's land that is agricultural and of q quality. $forestvalue_{ct}^{q}$ and $forestshare_i^q$ similarly refer to the use value and land share of forested land. I fill in the missing county's estimates with the mean of the use-value assessments for the surrounding counties. Figure A.2 shows the average per-acre value of these parcels. In line with expectations, calculated use values for land are higher in the more agriculturally productive and more densely populated eastern areas.

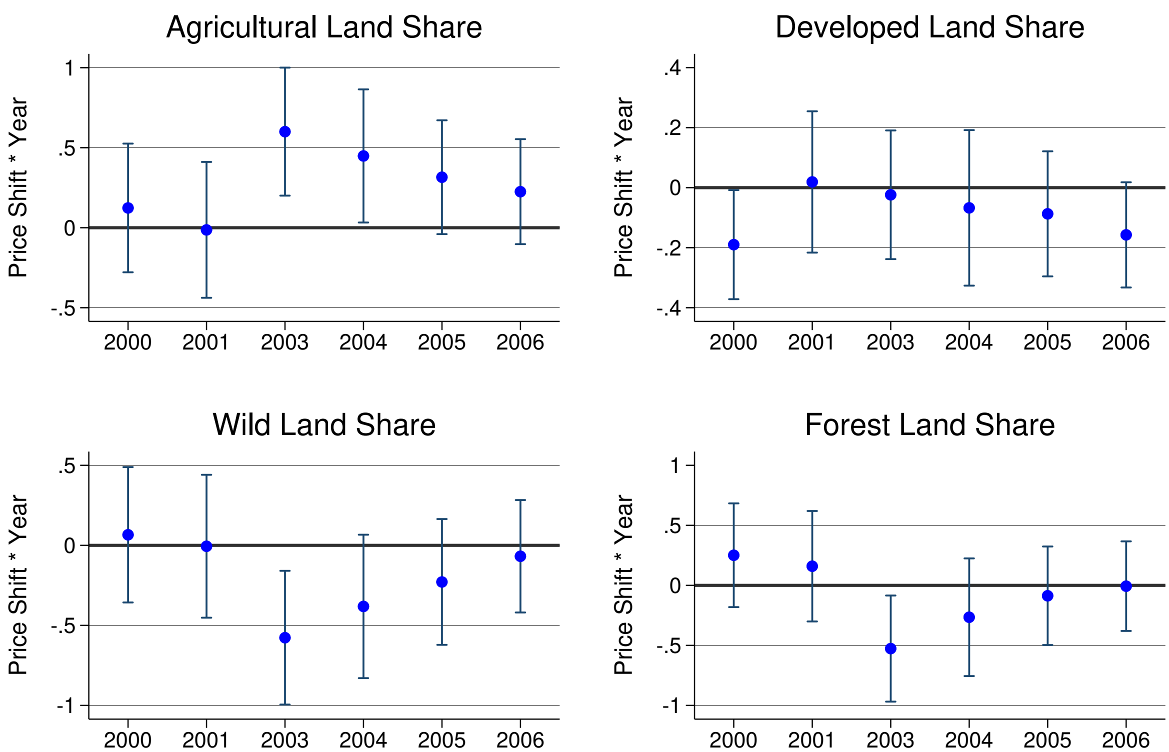
A.4 Event Study Figures

Figure A.3: Event Study Treatment Effect on Conservation Categories



Graph denotes coefficients on year * large, with 2001 omitted as the baseline year of the reform.

Figure A.4: Event Study Treatment Effect on Land Use



Graph denotes coefficients on year * large, with 2001 omitted as the baseline year of the reform.

References

Albers, Heidi J., Amy W. Ando, and Xiaoxuan Chen (July 1, 2008). “Spatial-econometric analysis of attraction and repulsion of private conservation by public reserves”. In: *Journal of Environmental Economics and Management* 56.1, pp. 33–49. ISSN: 0095-0696. DOI: 10.1016/j.jeem.2008.01.003. URL: https://www.sciencedirect.com/science/article/pii/S0095069608000351 (visited on 11/13/2023).

Alix-Garcia, Jennifer M., Elizabeth N. Shapiro, and Katharine R. E. Sims (2012). “Forest Conservation and Slippage: Evidence from Mexico’s National Payments for Ecosystem Services Program”. In: *Land Economics* 88.4, pp. 613–638. ISSN: 0023-7639. URL: http://www.jstor.org/stable/23272663 (visited on 09/21/2020).

Arbuckle, J. G. and G. Roesch-McNally (Nov. 1, 2015). “Cover crop adoption in Iowa: The role of perceived practice characteristics”. In: *Journal of Soil and Water Conservation* 70.6. Publisher: Soil and Water Conservation Society Section: Research Section, pp. 418–429. ISSN: 0022-4561, 1941-3300. DOI: 10.2489/jswc.70.6.418. URL: https://www.jswconline.org/content/70/6/418 (visited on 09/15/2021).

Associated Press (June 2024). *Heavy Snows and Drought of Deadly 'Dzud' Kill More Than 7 Million Head of Livestock in Mongolia*. en. URL: //www.usnews.com/news/business/articles/2024-06-12/heavy-snows-and-drought-of-deadly-dzud-kill-more-than-7-million-head-of-livestock-in-mongolia (visited on 10/24/2024).

Bandiera, Oriana, Robin Burgess, Narayan Das, Selim Gulesci, Imran Rasul, and Munshi Sulaiman (May 2017). “Labor Markets and Poverty in Village Economies”. en. In: *The Quarterly Journal of Economics* 132.2, pp. 811–870. ISSN: 0033-5533, 1531-4650. DOI: 10.1093/qje/qjx003. URL: https://academic.oup.com/qje/article/132/2/811/3075123 (visited on 08/31/2020).

Barnes, Jessica C., Mary Sketch, Ashley R. Gramza, Michael G. Sorice, Rich Iovanna, and Ashley A. Dayer (2020). “Land use decisions after the Conservation Reserve Program: Re-enrollment, reversion, and persistence in the southern Great Plains”. In: *Conservation Science and Practice* 2.9. _eprint: https://onlinelibrary.wiley.com/doi/pdf/10.1111/csp2.254, e254. ISSN: 2578-4854. DOI: 10.1111/csp2.254. URL: https://onlinelibrary.wiley.com/doi/abs/10.1111/csp2.254 (visited on 03/19/2024).

Barrett, Christopher B. and Winnie K. Luseno (Aug. 2004). “Decomposing producer price risk: a policy analysis tool with an application to northern Kenyan livestock markets”. en. In: *Food Policy*. Less Favoured Areas 29.4, pp. 393–405. ISSN: 0306-9192. DOI:

10.1016/j.foodpol.2004.07.008. URL:
http://www.sciencedirect.com/science/article/pii/S0306919204000478 (visited on 11/09/2020).

Barrett, Christopher B., Paswel Phiri Marenya, John Mcpeak, Bart Minten, Festus Murithi, Willis Oluoch-Kosura, Frank Place, Jean Claude Randrianarisoa, Jhon Rasambainarivo, and Justine Wangila (Feb. 2006). "Welfare dynamics in rural Kenya and Madagascar". In: *The Journal of Development Studies* 42.2. Publisher: Routledge _eprint: https://doi.org/10.1080/00220380500405394, pp. 248–277. ISSN: 0022-0388. DOI: 10.1080/00220380500405394. URL: https://doi.org/10.1080/00220380500405394 (visited on 12/06/2020).

Baumgart-Getz, Adam, Linda Stalker Prokopy, and Kristin Floress (Apr. 15, 2012). "Why farmers adopt best management practice in the United States: A meta-analysis of the adoption literature". In: *Journal of Environmental Management* 96.1, pp. 17–25. ISSN: 0301-4797. DOI: 10.1016/j.jenvman.2011.10.006. URL:
https://www.sciencedirect.com/science/article/pii/S0301479711003598 (visited on 06/28/2024).

Bergtold, Jason S., Steven Ramsey, Lucas Maddy, and Jeffery R. Williams (Feb. 2019). "A review of economic considerations for cover crops as a conservation practice". In: *Renewable Agriculture and Food Systems* 34.1, pp. 62–76. ISSN: 1742-1705, 1742-1713. DOI: 10.1017/S1742170517000278. URL:
https://www.cambridge.org/core/journals/renewable-agriculture-and-food-systems/article/abs/review-of-economic-considerations-for-cover-crops-as-a-conservation-practice/488B324AD119DDC661D5F4271035FC1B (visited on 08/01/2024).

Bertram-Huemmer, Veronika and Kati Kraehnert (Jan. 2018). "Does Index Insurance Help Households Recover from Disaster? Evidence from IBLI Mongolia". en. In: *American Journal of Agricultural Economics* 100.1. Publisher: John Wiley & Sons, Ltd, pp. 145–171. ISSN: 1467-8276. DOI: 10.1093/ajae/aax069. URL:
http://onlinelibrary.wiley.com/doi/10.1093/ajae/aax069 (visited on 05/27/2020).

Blanco-Canqui, Humberto, M. M. Claassen, and D. R. Presley (2012). "Summer Cover Crops Fix Nitrogen, Increase Crop Yield, and Improve Soil–Crop Relationships". In: *Agronomy Journal* 104.1. _eprint: https://onlinelibrary.wiley.com/doi/pdf/10.2134/agronj2011.0240, pp. 137–147. ISSN: 1435-0645. DOI: 10.2134/agronj2011.0240. URL:
https://onlinelibrary.wiley.com/doi/abs/10.2134/agronj2011.0240 (visited on 10/15/2024).

Boyer, Christopher N., Dayton M. Lambert, James A. Larson, and Donald D. Tyler (2018). "Investment Analysis of Cover Crop and No-Tillage Systems on Tennessee Cotton". In: *Agronomy Journal* 110.1. _eprint:
https://onlinelibrary.wiley.com/doi/pdf/10.2134/agronj2017.08.0431, pp. 331–338. ISSN: 1435-0645. DOI: 10.2134/agronj2017.08.0431. URL:
https://onlinelibrary.wiley.com/doi/abs/10.2134/agronj2017.08.0431 (visited on 12/17/2024).

Brandão, Miguel and Annie Levasseur (Jan. 1, 2011). *Assessing Temporary Carbon Storage in Life Cycle Assessment and Carbon Footprinting: Outcomes of an expert workshop*. JRC Scientific and Technical Reports.

Braza, Mark (2017). "Effectiveness of conservation easements in agricultural regions". In: *Conservation Biology* 31.4. _eprint: https://onlinelibrary.wiley.com/doi/pdf/10.1111/cobi.12909, pp. 848–859. ISSN: 1523-1739. DOI: 10.1111/cobi.12909. URL: http://conbio.onlinelibrary.wiley.com/doi/abs/10.1111/cobi.12909 (visited on 09/01/2020).

Brophy, Pauric and Michael Cuddy (Jan. 2007). "The Market for Financial Services in Rural Mongolia. Rural Finance and Sustainable Livelihoods". In.

Burke, Marshall, Lauren Falcao Bergquist, and Edward Miguel (2018). "Sell Low and Buy High: Arbitrage and Local Price Effects in Kenyan Markets". en. In: p. 166.

Carter, Michael, Alain de Janvry, Elisabeth Sadoulet, and Alexandros Sarris (Oct. 2017). "Index Insurance for Developing Country Agriculture: A Reassessment". In: *Annual Review of Resource Economics* 9.1. Publisher: Annual Reviews, pp. 421–438. ISSN: 1941-1340. DOI: 10.1146/annurev-resource-100516-053352. URL: https://www.annualreviews.org/doi/10.1146/annurev-resource-100516-053352 (visited on 11/05/2020).

Carter, Michael R. and Travis J. Lybbert (Nov. 2012). "Consumption versus asset smoothing: testing the implications of poverty trap theory in Burkina Faso". en. In: *Journal of Development Economics* 99.2, pp. 255–264. ISSN: 0304-3878. DOI: 10.1016/j.jdeveco.2012.02.003. URL: http://www.sciencedirect.com/science/article/pii/S0304387812000077 (visited on 11/24/2020).

Chantarat, Sommarat, Andrew G Mude, and Christopher B Barrett (2009). "Willingness to Pay for Index Based Livestock Insurance: Results from a Field Experiment in Northern Kenya". en. In.

Chantarat, Sommarat, Andrew G. Mude, Christopher B. Barrett, and Calum G. Turvey (June 2017). "Welfare Impacts of Index Insurance in the Presence of a Poverty Trap". en. In: *World Development* 94, pp. 119–138. ISSN: 0305-750X. DOI: 10.1016/j.worlddev.2016.12.044. URL: http://www.sciencedirect.com/science/article/pii/S0305750X16305988 (visited on 09/24/2020).

Claassen, Roger, Eric N. Duquette, and David J. Smith (Feb. 1, 2018). "Additionality in U.S. Agricultural Conservation Programs". In: *Land Economics* 94.1. Publisher: University of Wisconsin Press, pp. 19–35. ISSN: 0023-7639, 1543-8325. DOI: 10.3368/le.94.1.19. URL: http://le.uwpress.org/content/94/1/19 (visited on 06/21/2021).

Clarke, Daniel J. (Feb. 2016). "A Theory of Rational Demand for Index Insurance". en. In: *American Economic Journal: Microeconomics* 8.1, pp. 283–306. ISSN: 1945-7669. DOI: 10.1257/mic.20140103. URL: https://www.aeaweb.org/articles?id=10.1257/mic.20140103 (visited on 11/05/2020).

Cole, S., X. Giné, J. Tobacman, R. Townsend, Petia Topalova, and J. Vickery (2013). "Barriers to Household Risk Management: Evidence from India." In: *American economic journal. Applied economics*. DOI: 10.1257/APP.5.1.104.

Cole, Shawn, Daniel Stein, and Jeremy Tobacman (May 2014). "Dynamics of Demand for Index Insurance: Evidence from a Long-Run Field Experiment". en. In: *American*

Economic Review 104.5, pp. 284–290. ISSN: 0002-8282. DOI: 10.1257/aer.104.5.284. URL: https://www.aeaweb.org/articles?id=10.1257/aer.104.5.284 (visited on 11/02/2020).
Colinvaux, Roger (2012). "The Conservation Easement Tax Expenditure: In Search of Conservation Value". In: *Columbia Journal of Environmental Law* 37.1, pp. 1–62. URL: https://heinonline.org/HOL/P?h=hein.journals/cjel37%5C&i=3 (visited on 03/11/2022).
Conservation Technology Information Center (Aug. 2023). *National Cover Crop Survey Report 2022-2023*.
Convention on Biological Diversity (Dec. 18, 2022). *Kunming-Montreal Global Biodiversity Framework*. Version Number: 1. UN Convention on Biological Diversity. DOI: 10.5281/ZENODO.3831673. URL: https://zenodo.org/record/3831673 (visited on 03/14/2024).
Cunha, Jesse M, Giacomo De Giorgi, and Seema Jayachandran (Jan. 2019). "The Price Effects of Cash Versus In-Kind Transfers". en. In: *The Review of Economic Studies* 86.1, pp. 240–281. ISSN: 0034-6527, 1467-937X. DOI: 10.1093/restud/rdy018. URL: https://academic.oup.com/restud/article/86/1/240/4962153 (visited on 08/11/2020).
Curran, William S. (2016). "Persistence of herbicides in soil". In: *Crops & Soils* 49.5. _eprint: https://onlinelibrary.wiley.com/doi/pdf/10.2134/cs2016-49-0504, pp. 16–21. ISSN: 2325-3606. DOI: 10.2134/cs2016-49-0504. URL: https://onlinelibrary.wiley.com/doi/abs/10.2134/cs2016-49-0504 (visited on 10/15/2024).
Deines, Jillian M., Sherrie Wang, and David B. Lobell (Dec. 2019). "Satellites reveal a small positive yield effect from conservation tillage across the US Corn Belt". In: *Environmental Research Letters* 14.12. Publisher: IOP Publishing, p. 124038. ISSN: 1748-9326. DOI: 10.1088/1748-9326/ab503b. URL: https://dx.doi.org/10.1088/1748-9326/ab503b (visited on 01/17/2024).
Drechsler, Martin, Karin Johst, and Frank Wätzold (Mar. 2017). "The cost-effective length of contracts for payments to compensate land owners for biodiversity conservation measures". In: *Biological Conservation* 207, pp. 72–79. ISSN: 00063207. DOI: 10.1016/j.biocon.2017.01.014. URL: https://linkinghub.elsevier.com/retrieve/pii/S0006320717301210 (visited on 06/28/2024).
Duquette, Nicolas (Mar. 1, 2016). *Do Tax Incentives Affect Charitable Contributions? Evidence from Public Charities' Reported Revenues*. 3163840. Rochester, NY. DOI: 10.2139/ssrn.3163840. URL: https://papers.ssrn.com/abstract=3163840 (visited on 11/13/2023).
Dynarski, Katherine A., Deborah A. Bossio, and Kate M. Scow (2020). "Dynamic Stability of Soil Carbon: Reassessing the "Permanence" of Soil Carbon Sequestration". In: *Frontiers in Environmental Science* 8. Publisher: Frontiers. ISSN: 2296-665X. DOI: 10.3389/fenvs.2020.514701. URL: https://www.frontiersin.org/articles/10.3389/fenvs.2020.514701/full#B185 (visited on 11/24/2020).
Einav, Liran, Amy Finkelstein, and Mark R. Cullen (Aug. 1, 2010). "Estimating Welfare in Insurance Markets Using Variation in Prices". In: *The Quarterly Journal of Economics* 125.3, pp. 877–921. ISSN: 0033-5533. DOI: 10.1162/qjec.2010.125.3.877. URL: https://doi.org/10.1162/qjec.2010.125.3.877 (visited on 04/28/2022).

Elkind, Peter (Dec. 20, 2017). *The Billion-Dollar Loophole*. ProPublica. ProPublica. URL: https://www.propublica.org/article/conservation-easements-the-billion-dollar-loophole (visited on 03/19/2021).

Environmental Working Group (2023). *EWG Conservation Database*. URL: https://conservation.ewg.org/index.php.

Fafchamps, Marcel, Christopher Udry, and Katherine Czukas (Apr. 1998). "Drought and saving in West Africa: are livestock a buffer stock?" In: *Journal of Development Economics* 55.2, pp. 273–305. ISSN: 0304-3878. DOI: 10.1016/S0304-3878(98)00037-6. URL: https://www.sciencedirect.com/science/article/pii/S0304387898000376 (visited on 10/26/2024).

Fearnside, Philip M., Daniel A. Lashof, and Pedro Moura-Costa (Sept. 1, 2000). "Accounting for time in Mitigating Global Warming through land-use change and forestry". In: *Mitigation and Adaptation Strategies for Global Change* 5.3, pp. 239–270. ISSN: 1573-1596. DOI: 10.1023/A:1009625122628. URL: https://doi.org/10.1023/A:1009625122628 (visited on 07/01/2021).

Feld, Alan, Theodore Sims, and Jacob Nielson (Oct. 2022). "Green, or Greed? A Fresh Perspective on the Valuation of Conservation Easements". In: *Tax Law Review* 76.1, p. 1. URL: https://scholarship.law.bu.edu/faculty_scholarship/1513.

Fishburn, Isla S., Peter Kareiva, Kevin J. Gaston, and Paul R. Armsworth (Mar. 26, 2009). "The Growth of Easements as a Conservation Tool". In: *PLOS ONE* 4.3. Publisher: Public Library of Science, e4996. ISSN: 1932-6203. DOI: 10.1371/journal.pone.0004996. URL: https://journals.plos.org/plosone/article?id=10.1371/journal.pone.0004996 (visited on 09/01/2020).

Fleming, Patrick, Erik Lichtenberg, and David A. Newburn (Nov. 1, 2018). "Evaluating impacts of agricultural cost sharing on water quality: Additionality, crowding In, and slippage". In: *Journal of Environmental Economics and Management* 92, pp. 1–19. ISSN: 0095-0696. DOI: 10.1016/j.jeem.2018.08.007. URL: https://www.sciencedirect.com/science/article/pii/S0095069617303467 (visited on 06/28/2024).

Galle, Brian D. (Nov. 29, 2015). *How Do Nonprofit Firms Respond to Tax Policy?* SSRN Research Papers. Rochester, NY. DOI: 10.2139/ssrn.2443621. URL: https://papers.ssrn.com/abstract=2443621 (visited on 11/13/2023).

Gibbens, Sarah (Jan. 27, 2021). *Biden commits to ambitious 30x30 conservation target*. Environment. National Geographic. National Geographic. URL: https://www.nationalgeographic.com/environment/article/biden-commits-to-30-by-2030-conservation-executive-orders (visited on 03/10/2022).

Gonzalez-Ramirez, Jimena, Catherine Kling, and J. Gordon Arbuckle (2015). *Cost-share Effectiveness in the Adoption of Cover Crops in Iowa*. 2015 AAEA & WAEA Joint Annual Meeting, July 26-28, San Francisco, California 205876. Agricultural and Applied Economics Association. URL: https://econpapers.repec.org/paper/agsaaea15/205876.htm (visited on 08/06/2024).

Grant, Laura and Christian Langpap (Apr. 5, 2022). *The Efficiency of Tax Deductions for Charitable Donations Reevaluated: Evidence from Environmental Groups*. SSRN Research Papers. Rochester, NY. DOI: 10.2139/ssrn.4075638. URL: https://papers.ssrn.com/abstract=4075638 (visited on 11/13/2023).

Haddad, Nick M. et al. (Mar. 20, 2015). “Habitat fragmentation and its lasting impact on Earth’s ecosystems”. In: *Science Advances* 1.2. Publisher: American Association for the Advancement of Science, e1500052. DOI: 10.1126/sciadv.1500052. URL: https://www.science.org/doi/10.1126/sciadv.1500052 (visited on 07/21/2022).

Happ, Michael (Sept. 9, 2021). *Closed out: How U.S. farmers are denied access to conservation programs*. URL: https://www.iatp.org/documents/closed-out-how-us-farmers-are-denied-access-conservation-programs (visited on 09/27/2024).

Hellerstein, Dan and Cindy Nickerson (May 2002). *Farmland Protection Programs: What Does the Public Want?* USDA Research Service. USDA Economic Research Service. URL: https://ssrn.com/abstract=2737128.

Honey-Rosés, Jordi, Kathy Baylis, and M. Isabel Ramírez (Oct. 2011). “A Spatially Explicit Estimate of Avoided Forest Loss: Spatial Estimate of Avoided Forest Loss”. In: *Conservation Biology* 25.5, pp. 1032–1043. ISSN: 08888892. DOI: 10.1111/j.1523-1739.2011.01729.x. URL: http://doi.wiley.com/10.1111/j.1523-1739.2011.01729.x (visited on 09/21/2020).

Ikegami, Madoka (May 2016). *Mongolia’s dzud disaster*. en. URL: https://newint.org/features/web-exclusive/2016/05/10/mongolias-dzud-disaster (visited on 08/11/2020).

Ingraham, Molly W. and Shonda Gilliland Foster (Nov. 1, 2008). “The value of ecosystem services provided by the U.S. National Wildlife Refuge System in the contiguous U.S.” In: *Ecological Economics* 67.4, pp. 608–618. ISSN: 0921-8009. DOI: 10.1016/j.ecolecon.2008.01.012. URL: https://www.sciencedirect.com/science/article/pii/S0921800908000396 (visited on 11/07/2023).

IPCC (Mar. 2023). *Climate Change 2023: Synthesis Report.* Assessment Report 6. Intergovernmental Panel on Climate Change. URL: https://www.ipcc.ch/assessment-report/ar6/ (visited on 11/01/2024).

Jack, B. Kelsey and Seema Jayachandran (Mar. 19, 2019). “Self-selection into payments for ecosystem services programs”. In: *Proceedings of the National Academy of Sciences* 116.12, pp. 5326–5333. ISSN: 0027-8424, 1091-6490. DOI: 10.1073/pnas.1802868115. URL: http://www.pnas.org/lookup/doi/10.1073/pnas.1802868115 (visited on 09/21/2020).

Janzen, Sarah A. and Michael R. Carter (2019). “After the Drought: The Impact of Microinsurance on Consumption Smoothing and Asset Protection”. en. In: *American Journal of Agricultural Economics* 101.3. _eprint: https://onlinelibrary.wiley.com/doi/pdf/10.1093/ajae/aay061, pp. 651–671. ISSN: 1467-8276. DOI: https://doi.org/10.1093/ajae/aay061. URL: http://onlinelibrary.wiley.com/doi/abs/10.1093/ajae/aay061 (visited on 11/25/2020).

Jayachandran, Seema (2006). “Selling Labor Low: Wage Responses to Productivity Shocks in Developing Countries”. In: *Journal of Political Economy* 114.3. Publisher: The University of Chicago Press, pp. 538–575. ISSN: 0022-3808. DOI: 10.1086/503579. URL: https://www.jstor.org/stable/10.1086/503579 (visited on 11/02/2020).

Jensen, Nathaniel D., Christopher B. Barrett, and Andrew G. Mude (Nov. 2017). “Cash transfers and index insurance: A comparative impact analysis from northern Kenya”. en. In: *Journal of Development Economics* 129, pp. 14–28. ISSN: 0304-3878. DOI: 10.1016/j.jdeveco.2017.08.002. URL:

http://www.sciencedirect.com/science/article/pii/S0304387817300627 (visited on 05/26/2020).
Johnson, Leigh, Brenda Wandera, Nathan Jensen, and Rupsha Banerjee (June 2019). "Competing Expectations in an Index-Based Livestock Insurance Project". In: *The Journal of Development Studies* 55.6. Publisher: Routledge _eprint: https://doi.org/10.1080/00220388.2018.1453603, pp. 1221–1239. ISSN: 0022-0388. DOI: 10.1080/00220388.2018.1453603. URL: https://doi.org/10.1080/00220388.2018.1453603 (visited on 05/26/2020).
Kazianga, Harounan and Christopher Udry (Apr. 2006). "Consumption smoothing? Livestock, insurance and drought in rural Burkina Faso". en. In: *Journal of Development Economics* 79.2, pp. 413–446. ISSN: 03043878. DOI: 10.1016/j.jdeveco.2006.01.011. URL: https://linkinghub.elsevier.com/retrieve/pii/S030438780600006X (visited on 02/12/2024).
Kladivko, E. J., T. C. Kaspar, D. B. Jaynes, R. W. Malone, J. Singer, X. K. Morin, and T. Searchinger (July 1, 2014). "Cover crops in the upper midwestern United States: Potential adoption and reduction of nitrate leaching in the Mississippi River Basin". In: *Journal of Soil and Water Conservation* 69.4. Publisher: Soil and Water Conservation Society Section: Research Section, pp. 279–291. ISSN: 0022-4561, 1941-3300. DOI: 10.2489/jswc.69.4.279. URL: https://www.jswconline.org/content/69/4/279 (visited on 08/01/2024).
Kulp, Stephen (2019). "Virginia Local Tax Rates, 2018". In: *Weldon Cooper Center for Public Service* 37th Annual Edition.
Kusano, Eiichi and Izuru Saizen (2013). "Spatial Market Integration of Livestock Products and Road Conditions in Mongolia". en. In: *Japan Agricultural Research Quarterly: JARQ* 47.4, pp. 423–433. ISSN: 0021-3551, 2185-8896. DOI: 10.6090/jarq.47.423. URL: https://www.jstage.jst.go.jp/article/jarq/47/4/47_423/_article (visited on 08/31/2023).
Laloy, Eric and C. L. Bielders (2010). "Effect of Intercropping Period Management on Runoff and Erosion in a Maize Cropping System". In: *Journal of Environmental Quality* 39.3. _eprint: https://onlinelibrary.wiley.com/doi/pdf/10.2134/jeq2009.0239, pp. 1001–1008. ISSN: 1537-2537. DOI: 10.2134/jeq2009.0239. URL: https://onlinelibrary.wiley.com/doi/abs/10.2134/jeq2009.0239 (visited on 08/01/2024).
Land Trust Alliance (2019). *Income Tax Incentives for Land Conservation*. Land Trust Alliance. Land Trust Alliance. URL: https://www.landtrustalliance.org/topics/taxes/income-tax-incentives-land-conservation (visited on 03/10/2022).
— (Aug. 18, 2021). *2020 Annual Report*. Text. Land Trust Alliance. Land Trust Alliance. URL: https://www.landtrustalliance.org/2020-annual-report (visited on 03/10/2022).
Lichtenberg, Erik, Haoluan Wang, and David Newburn (June 20, 2018). *Uptake and Additionality in a Green Payment Program: A Panel Data Study of the Maryland Cover Crop Program*. 274455. Publication Title: 2018 Annual Meeting, August 5-7, Washington, D.C. Agricultural and Applied Economics Association. URL: https://ideas.repec.org/p/ags/aaea18/274455.html (visited on 08/05/2021).
Looney, Adam (May 2017a). "Charitable Contributions of Conservation Easements". In: *Economic Studies at Brookings*, p. 39. URL: https://www.brookings.edu/wp-content/uploads/2017/05/looney%5C_conservationeasements.pdf.

Looney, Adam (Dec. 20, 2017b). *Estimating the rising cost of a surprising tax shelter: the syndicated conservation easement*. Brookings. Brookings. URL: https://www.brookings.edu/blog/up-front/2017/12/20/estimating-the-rising-cost-of-a-surprising-tax-shelter-the-syndicated-conservation-easement/ (visited on 08/31/2020).

Lötjönen, Sanna, Kati Kulovesi, Kristiina Lång, and Markku Ollikainen (Mar. 12, 2024). "Offset ratios and temporary contract designs for climate integrity in carbon farming". In: *Carbon Management* 15.1. Publisher: Taylor & Francis _eprint: https://doi.org/10.1080/17583004.2024.2329593, p. 2329593. ISSN: 1758-3004. DOI: 10.1080/17583004.2024.2329593. URL: https://doi.org/10.1080/17583004.2024.2329593 (visited on 06/28/2024).

Lybbert, Travis J., Christopher B. Barrett, Solomon Desta, and D. Layne Coppock (2004). "Stochastic wealth dynamics and risk management among a poor population*". en. In: *The Economic Journal* 114.498. _eprint: https://onlinelibrary.wiley.com/doi/pdf/10.1111/j.1468-0297.2004.00242.x, pp. 750–777. ISSN: 1468-0297. DOI: https://doi.org/10.1111/j.1468-0297.2004.00242.x. URL: http://onlinelibrary.wiley.com/doi/abs/10.1111/j.1468-0297.2004.00242.x (visited on 11/24/2020).

Lybbert, Travis J. and John McPeak (Mar. 2012). "Risk and intertemporal substitution: Livestock portfolios and off-take among Kenyan pastoralists". en. In: *Journal of Development Economics* 97.2, pp. 415–426. ISSN: 03043878. DOI: 10.1016/j.jdeveco.2011.06.004. URL: https://linkinghub.elsevier.com/retrieve/pii/S0304387811000617 (visited on 01/05/2021).

Mahul, Olivier and Jerry Skees (Nov. 2007). *Managing Agricultural Risk At The Country Level : The Case Of Index-Based Livestock Insurance In Mongolia*. en. Tech. rep. 4325. The World Bank. DOI: 10.1596/1813-9450-4325. URL: https://hdl.handle.net/10986/7273 (visited on 10/24/2024).

McCann, Laura and Roger Claassen (May 1, 2016). "Farmer Transaction Costs of Participating in Federal Conservation Programs: Magnitudes and Determinants". In: *Land Economics* 92.2. Publisher: University of Wisconsin Press Section: Articles, pp. 256–272. ISSN: 0023-7639, 1543-8325. DOI: 10.3368/le.92.2.256. URL: https://le.uwpress.org/content/92/2/256 (visited on 01/17/2024).

Merenlender, A. M., L. Huntsinger, G. Guthey, and S. K. Fairfax (2004). "Land Trusts and Conservation Easements: Who Is Conserving What for Whom?" In: *Conservation Biology* 18.1. _eprint: https://onlinelibrary.wiley.com/doi/pdf/10.1111/j.1523-1739.2004.00401.x, pp. 65–76. ISSN: 1523-1739. DOI: 10.1111/j.1523-1739.2004.00401.x. URL: http://conbio.onlinelibrary.wiley.com/doi/abs/10.1111/j.1523-1739.2004.00401.x (visited on 09/01/2020).

Meurs, Mieke, Amarjargal Amartuvshin, and Otgontugs Banzragch (Jan. 2017). "Livestock Income of Mongolian Herders: A Path to Rural Prosperity?" In: *Nomadic Peoples* 21, pp. 87–110. DOI: 10.3197/np.2017.210105.

Mezzatesta, Mariano, David Newburn, and Richard Woodward (Nov. 1, 2013). "Additionality and the Adoption of Farm Conservation Practices". In: *Land Economics* 89. DOI: 10.3368/le.89.4.722.

Miles, Lera, Raquel Agra, Sandeep Sengupta, Adriana Vidal, and Barney Dickson (Nov. 4, 2021). *Nature-based solutions for climate change mitigation*. UN Environmental Programme. URL: https://portals.iucn.org/library/node/49781 (visited on 11/02/2024).

Mobarak, Ahmed Mushfiq and Mark Rosenzweig (Jan. 2014). *Risk, Insurance and Wages in General Equilibrium*. en. Tech. rep. w19811. Cambridge, MA: National Bureau of Economic Research. DOI: 10.3386/w19811. URL: http://www.nber.org/papers/w19811.pdf (visited on 08/11/2020).

Morton, Trent, Jason Bergtold, and Andrew Price (Jan. 1, 2006). "The Economics of Cover Crop Biomass for Corn and Cotton". In.

Murray, Brian C., Brent Sohngen, and Martin T. Ross (Dec. 21, 2006). "Economic consequences of consideration of permanence, leakage and additionality for soil carbon sequestration projects". In: *Climatitic Change* 2007.80. Last Modified: 2018-06-08T10:25-04:00 Publisher: Kluwer Academic Publishers, pp. 127–143. URL: https://nicholasinstitute.duke.edu/environment/publications/naturalresources/economic-consequences-of-consideration-of-permanence-leakage-and-additionality-for-soil-carbon-sequestration-projects-1 (visited on 01/22/2024).

Myers, Rob, Alan Weber, and Tellatin (June 2019). *Cover Crop Economics*. SARE. URL: https://www.sare.org/resources/cover-crop-economics/ (visited on 01/17/2024).

Nandintsetseg, Banzragch, Masato Shinoda, Chunling Du, and Erdenebadrakh Munkhjargal (Oct. 2018). "Cold-season disasters on the Eurasian steppes: Climate-driven or man-made". In: *Scientific Reports* 8. ISSN: 2045-2322. DOI: 10.1038/s41598-018-33046-1. URL: https://www.ncbi.nlm.nih.gov/pmc/articles/PMC6170381/ (visited on 10/06/2020).

National Statistics Office of Mongolia (2024). *1212.mn Mongolia National Statistics Database*. URL: https://1212.mn/en/statistic/.

Niker, Lauren (Mar. 10, 2022). *The Environment Act 2021: Conservation Covenants*. Field Seymour Parkes. Field Seymour Parkes. URL: https://www.fsp-law.com/the-environment-act-2021-conservation-covenants/ (visited on 03/21/2023).

NRCS (2024). *Financial Assistance Program Dashboard*. Farmers.gov. URL: https://www.farmers.gov/data/financial-assistance-overview (visited on 12/18/2024).

O'Bannon, John M. (Sept. 28, 2012). *Dedicated Revenue Sources for Land Conservation in Virginia*. URL: http://jlarc.virginia.gov/pdfs/reports/Rpt429.pdf.

O'Connor, Claire (2013). "Soil Matters: How the Federal Crop Insurance Program should be reformed to encourage low-risk farming methods with high-reward environmental outcomes". In: *2013 AAEA: Crop Insurance and the Farm Bill Symposium*. Number: 156789 Publisher: Agricultural and Applied Economics Association. URL: https://ideas.repec.org//p/ags/aaeaci/156789.html (visited on 10/15/2024).

Oldfield, Emily E., Alison J. Eagle, Rachel L. Rubin, Joseph Rudek, Jonathan Sanderman, and Doria R. Gordon (Mar. 18, 2022). "Crediting agricultural soil carbon sequestration". In: *Science* 375.6586. Publisher: American Association for the Advancement of Science, pp. 1222–1225. DOI: 10.1126/science.abl7991. URL: https://www.science.org/doi/10.1126/science.abl7991 (visited on 10/17/2023).

Oliva, Malena and Eduardo García Frapolli (Mar. 2024). "Conservation backfire: Local effects of international protected area policy". In: *Environmental Science & Policy* 153, p. 103676. ISSN: 1462-9011. DOI: 10.1016/j.envsci.2024.103676. URL:

https://www.sciencedirect.com/science/article/pii/S1462901124000108 (visited on 03/14/2024).

Ooluun, B (Jan. 2024). *Meat Export Increases Amid Harsh Winter*. en. URL: https://montsame.mn/en/read/335547 (visited on 09/30/2024).

Pannell, David J. and Roger Claassen (2020). "The Roles of Adoption and Behavior Change in Agricultural Policy". In: *Applied Economic Perspectives and Policy* 42.1. _eprint: https://onlinelibrary.wiley.com/doi/pdf/10.1002/aepp.13009, pp. 31–41. ISSN: 2040-5804. DOI: 10.1002/aepp.13009. URL: https://onlinelibrary.wiley.com/doi/abs/10.1002/aepp.13009 (visited on 09/14/2021).

Park, Byungyul, Roderick M. Rejesus, Serkan Aglasan, Yuyuan Che, Stephen C. Hagen, and William Salas (2023). "Payments from agricultural conservation programs and cover crop adoption". In: *Applied Economic Perspectives and Policy* 45.2. _eprint: https://onlinelibrary.wiley.com/doi/pdf/10.1002/aepp.13248, pp. 984–1007. ISSN: 2040-5804. DOI: 10.1002/aepp.13248. URL: https://onlinelibrary.wiley.com/doi/abs/10.1002/aepp.13248 (visited on 06/28/2024).

Parker, Dominic P. and Walter N. Thurman (Apr. 2018). "Tax Incentives and the Price of Conservation". In: *Journal of the Association of Environmental and Resource Economists* 5.2, pp. 331–369. ISSN: 2333-5955, 2333-5963. DOI: 10.1086/695615. URL: https://www.journals.uchicago.edu/doi/10.1086/695615 (visited on 11/02/2020).

— (Oct. 4, 2019). "Private Land Conservation and Public Policy: Land Trusts, Land Owners, and Conservation Easements". In: *Annual Review of Resource Economics* 11.1. Publisher: Annual Reviews, pp. 337–354. ISSN: 1941-1340. DOI: 10.1146/annurev-resource-100518-094121. URL: http://www.annualreviews.org/doi/10.1146/annurev-resource-100518-094121 (visited on 12/11/2020).

Peloza, John and Piers Steel (2005). "The Price Elasticities of Charitable Contributions: A Meta-Analysis". In: *Journal of Public Policy & Marketing* 24.2. Publisher: American Marketing Association, pp. 260–272. ISSN: 0743-9156. URL: https://www.jstor.org/stable/30000664 (visited on 11/13/2023).

Phillips, Spencer and Beth McGee (May 3, 2016). "Ecosystem Service Benefits of a Cleaner Chesapeake Bay". In: *Coastal Management* 44.3, pp. 241–258. ISSN: 0892-0753, 1521-0421. DOI: 10.1080/08920753.2016.1160205. URL: http://www.tandfonline.com/doi/full/10.1080/08920753.2016.1160205 (visited on 11/07/2023).

Poeplau, Christopher and Axel Don (Feb. 1, 2015). "Carbon sequestration in agricultural soils via cultivation of cover crops – A meta-analysis". In: *Agriculture, Ecosystems & Environment* 200, pp. 33–41. ISSN: 0167-8809. DOI: 10.1016/j.agee.2014.10.024. URL: https://www.sciencedirect.com/science/article/pii/S0167880914004873 (visited on 08/01/2024).

Pratt, Bryan, Laura Paul, Maria Bowman, Paul Ferraro, Kent Messer, and Jing Gao (Nov. 3, 2023). "Longitudinal Measurement of Cover Crop Use and Persistence". In: Annual Meetings of the Southern Economics Association. New Orleans, NY.

Racinska, Inga and Siim Vahtrus (2018). *The Use of Conservation Easements in the European Union*. European Private Land Conservation Network. European Private Land Conservation Network.

Rasmussen, Deborah and Shombodon Dorlig (June 2011). *Mongolia - Improving feed and fodder supply for dzud management*. en. Text/HTML. World Bank. URL: https://documents.worldbank.org/en/publication/documents-reports/documentdetail/158631468274239063/Mongolia-Improving-feed-and-fodder-supply-for-dzud-management (visited on 10/03/2024).

Reimer, Adam P., Denise Klotthor Weinkauf, and Linda Stalker Prokopy (Jan. 1, 2012). "The influence of perceptions of practice characteristics: An examination of agricultural best management practice adoption in two Indiana watersheds". In: *Journal of Rural Studies* 28.1, pp. 118–128. ISSN: 0743-0167. DOI: 10.1016/j.jrurstud.2011.09.005. URL: https://www.sciencedirect.com/science/article/pii/S0743016711000878 (visited on 01/17/2024).

Robinson, Clay and David Nielsen (2015). "The water conundrum of planting cover crops in the Great Plains: When is an inch not an inch?" In: *Crops & Soils* 48.1. _eprint: https://onlinelibrary.wiley.com/doi/pdf/10.2134/cs2015-48-1-7, pp. 24–31. ISSN: 2325-3606. DOI: 10.2134/cs2015-48-1-7. URL: https://onlinelibrary.wiley.com/doi/abs/10.2134/cs2015-48-1-7 (visited on 01/17/2024).

Roesch-McNally, Gabrielle E., Andrea D. Basche, J. G. Arbuckle, John C. Tyndall, Fernando E. Miguez, Troy Bowman, and Rebecca Clay (Aug. 2018). "The trouble with cover crops: Farmers' experiences with overcoming barriers to adoption". In: *Renewable Agriculture and Food Systems* 33.4. Publisher: Cambridge University Press, pp. 322–333. ISSN: 1742-1705, 1742-1713. DOI: 10.1017/S1742170517000096. URL: https://www.cambridge.org/core/journals/renewable-agriculture-and-food-systems/article/trouble-with-cover-crops-farmers-experiences-with-overcoming-barriers-to-adoption/732DAC57E92E1C9EFC5A451F7EAF454A (visited on 08/19/2021).

Rosenberg, Andrew B., Bryan Pratt, and David Arnold (2022). "Land Use Impacts of the Conservation Reserve Program: An Analysis of Rejected CRP Offers". In: 2022 ASSA Annual Meeting. DOI: 10.22004/ag.econ.316533.

Rosenberg, Andrew B., Bryan Pratt, and Daniel Szmurlo (Aug. 11, 2024). "Impacts of an increase in federal assistance for cover cropping: Evidence from the Environmental Quality Incentives Program". In: *American Journal of Agricultural Economics* n/a (n/a). _eprint: https://onlinelibrary.wiley.com/doi/pdf/10.1111/ajae.12502. ISSN: 1467-8276. DOI: 10.1111/ajae.12502. URL: https://onlinelibrary.wiley.com/doi/abs/10.1111/ajae.12502 (visited on 12/10/2024).

Saez, Emmanuel (2004). "The optimal treatment of tax expenditures". In: *Journal of Public Economics* 88, pp. 2657–2684.

Sainju, Upendra M. and Bharat P. Singh (2001). "Tillage, Cover Crop, and Kill-Planting Date Effects on Corn Yield and Soil Nitrogen". In: *Agronomy Journal* 93.4. _eprint: https://onlinelibrary.wiley.com/doi/pdf/10.2134/agronj2001.934878x, pp. 878–886. ISSN: 1435-0645. DOI: 10.2134/agronj2001.934878x. URL: https://onlinelibrary.wiley.com/doi/abs/10.2134/agronj2001.934878x (visited on 10/15/2024).

Santos, Paulo and Christopher B. Barrett (Nov. 2011). "Persistent poverty and informal credit". en. In: *Journal of Development Economics* 96.2, pp. 337–347. ISSN: 0304-3878. DOI: 10.1016/j.jdeveco.2010.08.017. URL:

http://www.sciencedirect.com/science/article/pii/S030438781000101X (visited on 12/06/2020).

Sawadgo, Wendiam and Alejandro Plastina (Dec. 2021). “Do cost-share programs increase cover crop use? Empirical evidence from Iowa”. In: *Renewable Agriculture and Food Systems* 36.6, pp. 527–535. ISSN: 1742-1705, 1742-1713. DOI: 10.1017/S1742170521000132. URL: https://www.cambridge.org/core/journals/renewable-agriculture-and-food-systems/article/do-costshare-programs-increase-cover-crop-use-empirical-evidence-from-iowa/83B2FCF05B192C5BFFB374B115E9A55A (visited on 08/01/2024).

Schipanski, Meagan E. et al. (Mar. 1, 2014). “A framework for evaluating ecosystem services provided by cover crops in agroecosystems”. In: *Agricultural Systems* 125, pp. 12–22. ISSN: 0308-521X. DOI: 10.1016/j.agsy.2013.11.004. URL: https://www.sciencedirect.com/science/article/pii/S0308521X13001492 (visited on 10/15/2024).

Shagdar, Enkhbayar (2002). *The Mongolian livestock sector: Vital for the economy and people, but vulnerable to natural phenomena.* Tech. rep. 47.

Snapp, S. S., S. M. Swinton, R. Labarta, and D. Mutch (Feb. 2005). “Evaluating Cover Crops for Benefits, Costs and Performance within Cropping System Niches”. In: *Agronomy Journal* 97.1. Num Pages: 11 Place: Madison, United States Publisher: American Society of Agronomy Section: REVIEW AND INTERPRETATION, pp. 322–332. ISSN: 00021962. URL: https://www.proquest.com/docview/194507436/citation/A73C0A2EC94F4CBAPQ/1 (visited on 10/15/2024).

Soppelsa, Maria Edisa (2017). “Do State Tax Breaks for Land Conservation Work?” In: Semantic Scholar, p. 45. URL: https://api.semanticscholar.org/CorpusID:197643942.

Stubbs, Megan (May 9, 2011). *Environmental Quality Incentives Program (EQIP): Status and Issues*. R40197. Congressional Research Service.

Sundberg, Jeffrey (Mar. 1, 2013). “Using conservation easements to protect open space: public policy, tax effects, and challenges”. In: *Journal of Property Tax Assessment & Administration* 10.1, pp. 5–20. ISSN: 1357-1419. URL: https://researchexchange.iaao.org/jptaa/vol10/iss1/1.

Sundberg, Jeffrey O. (2011). *State Income Tax Credits for Conservation Easements: Do Additional Credits Create Additional Value?* Lincoln Institute of Tax Policy Working Paper. URL: https://www.lincolninst.edu/sites/default/files/1961%5C_1282%5C_Sundberg%5C_FinalWP11JS1.pdf.

Suter, Jordan, Dissanayake Sahan, and Lewis Lynne (2014). “Public Incentives for Conservation on Private Land”. AgEcon Search. In: *Agricultural & Applied Economics Association’s 2014 AAEA Annual Meeting*. Number: 329-2016-12786. DOI: 10.22004/ag.econ.170706. URL: https://ageconsearch.umn.edu/record/170706 (visited on 04/20/2021).

Tax Policy Center (2022). *How large are individual income tax incentives for charitable giving?* Tax Policy Center’s Briefing Book. Tax Policy Center. URL: https://www.taxpolicycenter.org/briefing-book/how-large-are-individual-income-tax-incentives-charitable-giving (visited on 11/27/2023).

Thornton, Philip K., R. L. Kruska, N. Henninger, Patricia M. Kristjanson, Robin S. Reid, F. Atieno, A. N. Odero, and T. Ndegwa (2002). *Mapping poverty and livestock in the*

developing world. en. Accepted: 2010-03-25T05:51:50Z. ILRI. ISBN: 978-92-9146-109-7. URL: https://cgspace.cgiar.org/handle/10568/915 (visited on 10/06/2020).
Turner, Benjamin, Melissa Wuellner, Timothy Nichols, and Roger Gates (Oct. 1, 2014). "Dueling Land Ethics: Uncovering Agricultural Stakeholder Mental Models to Better Understand Recent Land Use Conversion". In: *Journal of Agricultural and Environmental Ethics* 27.5, pp. 831–856. URL: https://openprairie.sdstate.edu/nrm_pubs/88.
USDA NRCS (Apr. 2018). *Title 440-Conservation Programs Manual, Part 512 - Conservation Program contracting*. 440-512-M. USDA NRCS.
Vercammen, James (Jan. 2019). "A Welfare Analysis of Conservation Easement Tax Credits". In: *Journal of the Association of Environmental and Resource Economists* 6.1, pp. 43–71. ISSN: 2333-5955, 2333-5963. DOI: 10.1086/700721. URL: https://www.journals.uchicago.edu/doi/10.1086/700721 (visited on 12/14/2020).
Villamagna, A., L. Scott, and J. Gillespie (Sept. 2015). "Collateral benefits from public and private conservation lands: a comparison of ecosystem service capacities". In: *Environmental Conservation* 42.3. Publisher: Cambridge University Press, pp. 204–215. ISSN: 0376-8929, 1469-4387. DOI: 10.1017/S0376892914000393. URL: http://www.cambridge.org/core/journals/environmental-conservation/article/collateral-benefits-from-public-and-private-conservation-lands-a-comparison-of-ecosystem-service-capacities/E0E2EF80D150B720F57A7060DBFDAB47 (visited on 09/01/2020).
Vogelsong, Sarah (Aug. 24, 2021). *Va.'s reliance on tax credits for land conservation is 'certainly not equitable,' review finds*. Virginia Mercury. Virginia Mercury. URL: https://www.virginiamercury.com/2021/08/24/northam-admin-criticizes-state-land-conservation-policy-as-inequitable/ (visited on 08/31/2021).
Wallander, Steven, Maria Bowman, Peter Beeson, and Roger Claassen (Dec. 1, 2017). *Farmers and Habits: The Challenge of Identifying the Sources of Persistence in Tillage Decisions*. 2018 Allied Social Sciences Association (ASSA) Annual Meeting, January 5-7, 2018, Philadelphia, Pennsylvania 266307. Agricultural and Applied Economics Association. URL: https://econpapers.repec.org/paper/agsassa18/266307.htm (visited on 01/22/2024).
Wallander, Steven, Roger Claasen, Alexandra Hill, and Jacob Fooks (Mar. 2019). *Working Lands Conservation Contract Modifications: Patterns in Dropped Practices*. Economic Research Report 262. USDA, p. 54.
Wang, Tong, Zheng Xu, Deepthi Kolady, Jessica D. Ulrich-Schad, and David Clay (2021). "Cover Crops Usage in South Dakota: Farmer Perceived Profitability and Future Adoption Decisions". In: *Journal of Agricultural and Resource Economics* 46.2, pp. 287–307. URL: https://jareonline.org/articles/cover-crops-usage-in-south-dakota-farmer-perceived-profitability-and-future-adoption-decisions/ (visited on 01/09/2024).
Wilson, Janette (2005). *Individual Noncash Contributions, 2005*. IRS Report. IRS, p. 22. URL: https://www.irs.gov/pub/irs-soi/05incontribul.pdf.
Wittwer, Raphaël A., Brigitte Dorn, Werner Jossi, and Marcel G. A. van der Heijden (Feb. 3, 2017). "Cover crops support ecological intensification of arable cropping systems". In: *Scientific Reports* 7.1. Publisher: Nature Publishing Group, p. 41911. ISSN: 2045-2322. DOI: 10.1038/srep41911. URL: https://www.nature.com/articles/srep41911 (visited on 08/01/2024).

Wyles, Kayleigh J., Mathew P. White, Caroline Hattam, Sabine Pahl, Haney King, and Melanie Austen (Feb. 1, 2019). "Are Some Natural Environments More Psychologically Beneficial Than Others? The Importance of Type and Quality on Connectedness to Nature and Psychological Restoration". In: *Environment and Behavior* 51.2. Publisher: SAGE Publications Inc, pp. 111–143. ISSN: 0013-9165. DOI: 10.1177/0013916517738312. URL: https://doi.org/10.1177/0013916517738312 (visited on 02/01/2022).