ECONOMIC STUDIES

DEPARTMENT OF ECONOMICS
SCHOOL OF BUSINESS, ECONOMICS AND LAW
UNIVERSITY OF GOTHENBURG
257

Wetland Mitigation Banking in the United States

Ville Inkinen

UNIVERSITY OF
GOTHENBURG

ISBN 978-91-88199-73-7 (printed)
ISBN 978-91-88199-74-4 (pdf)
ISSN 1651-4289 (printed)
ISSN 1651-4297 (online)

Printed in Sweden,
Stema 2023

SVANENMÄRKET
Trycksak
3041 0234

Acknowledgements

I would like to express my deepest gratitude for the guidance and mentorship from my supervisors, Jessica Coria, Mikael Lindahl, and Fredrik Carlsson. Your advice has been invaluable in shaping my research direction, and your support and encouragement have helped me immensely over the past five years. Jessica, I deeply value your patience and generosity with time. I am also extremely grateful to Randi Hjalmarsson for her guidance throughout the PhD program and to Andreas Dzemski for his support and advice during the final year.

I want to acknowledge the efforts of my co-authors Yann Clough, Jessica Coria, and João Vaz. Thank you for the productive collaboration that has greatly contributed to the successful completion of this work. I am also indebted to Martin Quaas and Mathias Reynaert for providing thoughtful comments on my thesis chapters in the final seminars. Your suggestions have significantly improved the research presented in this thesis.

It has been a privilege to pursue my PhD education at the Department of Economics in Gothenburg. The research expertise and collegiality in the Department have significantly contributed to my professional growth and development. I want to thank everyone who dedicated their time and effort to discuss research and give me feedback and suggestions in various stages. To Thomas Sterner, thank you for your generous advice and the opportunity to participate in organizing our seminars. I also extend special thanks to everyone in the environmental group for numerous interesting discussions.

To Anders Boman, Håkan Eggert, Alexander Herbertsson, and Elina Lampi, it has been a real pleasure to teach courses together with you. To the administrative staff at the Department, a heartfelt thank you for all the help you have provided.

To my fellow PhD students, I thank you all for your companionship and collegiality. David, Dominik, Magnus, and Vũ, it has been a pleasure to share with you the early years of the journey and beyond. I cherish that time and our friendship.

Words cannot express my gratitude to all my family and friends who have stood by me and supported me throughout my time in Gothenburg.

My parents, Kari and Rauni, and my godmother, Raili, thank you for always having been there for me. Kiitos kaikesta avusta ja tuesta. To you I dedicate this work.

Ville Inkinen
Exeter, August 2023

Contents

List of Figures

Chapter 3

List of Tables

Introduction

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services states in its Global Assessment Report (Brondizio et al., 2019) that 25% of species in assessed animal and plants groups are threatened and that 1 million species already face extinction within decades, a rate which amounts to up to hundreds of times the average extinction rate over the past 10 million years. The uncertainty and risks associated with the scale and speed of biodiversity loss require urgent and effective action from national governments (Dasgupta, 2021).

In December 2022, the parties to the Convention on Biological Diversity adopted the Kunming-Montreal Global Biodiversity Framework, which calls for conserving 30% of the Earth's land area and restoring 30% of previously degraded ecosystems. Reaching these targets is a necessary step to halt the ongoing extinction rates. Still, they incur significant economic costs in terms of forgone opportunities to exploit land area and natural resources. Offsetting mechanisms can alleviate the total costs by allowing environmental impacts in some locations while requiring a commensurate environmental improvement elsewhere. In particular, market-based offsetting policies have a strong potential to reduce the costs of biodiversity conservation (OECD, 2016).

In this thesis, I study market-based biodiversity conservation policies in the context of the wetland compensatory mitigation program under the US Clean Water Act. The program requires developers to compensate for adverse impacts on wetlands by purchasing credits that specialized firms have generated in advance from wetland conservation activities. As a result, the program has created a thriving private conservation industry.

Several aspects of the performance of the mitigation banking program remain ambiguous. First, it is unclear whether the program is achieving its long-standing environmental goal of no net loss of wetland area or functions. Second, within the market-based program, a developer has the option to realize a conservation project instead of resorting to buying credits, and it is ambiguous in which conditions resorting to this option is preferable to using the market mechanism. Third, the geographic extent of the Clean Water Act has been contested in courts and is under significant political debate. The economic costs of the program at various margins are ambiguous, and so are the effects of the market-based compensatory mitigation program in alleviating these costs. In this work, I engage with these questions, with each forming the main focus of a chapter.

In the first chapter, *Evaluation of Wetland Area Gains and Losses under the US Clean Water Act* (with Jessica Coria, João Vaz, and Yann Clough), we evaluate environmental outcomes in the compensatory credit markets. We measure wetland area gains at 400 compensation sites over 1995–2020 using a combination of high-resolution satellite imagery and land cover change data. This study makes two novel contributions: we estimate the causal impact of offsetting projects on environmental outcomes robustly, and we use these estimates to provide a program-wide evaluation of a no-net-loss target. Comparing realized compensation projects to planned but withdrawn projects in a difference-in-differences framework, we find that the dedicated conservation activities are additional to a baseline scenario where the activity did not receive funding from the federal scheme.

There is significant heterogeneity in our estimates across the administrative regions of the regulatory program. Wetland area gains are the highest in regions with an abundance of low-cost agricultural land that can be converted to wetlands. Conversely, in regions with less available agricultural land for conversion, firms choose to earn compensation credits by improving the functions of existing wetlands instead of establishing new wetland areas. In other words, the market mechanism allocates the type and location of conservation activities according to the opportunity cost of land use.

The magnitude of the estimated wetland area gains appears insufficient to compensate for the wetland area losses regulated within the program. This makes it unlikely that the program will achieve its environmental goals in the long term. Notably, our analysis is based on an optimistic scenario where the compensation activities perfectly succeed in recreating natural environments comparable to those that were lost due to economic development. Deviations from this assumption will render the implications of our results even less favorable. In other words, our estimates should be considered a lower bound regarding their implications on the net losses of wetland area and functions. Further research on the program should scrutinize the degree of ecological equivalence between the lost wetland resources and the compensation projects.

In the second chapter, *The Choice of Mechanism for Biodiversity Offsetting* (with João Vaz and Jessica Coria), we develop a theoretical model of a market approach to compensating ecological damages. It is unclear whether market-based instruments, such as banking mechanisms that entail third-party offsets for developers to purchase, hold significant promise for implementing no-net-loss regulation at least cost, relative to the conventional command-and-control approach of developer-led offsets. In this paper, we provide a theoretical examination of the costs and benefits of the two approaches. We find that (1) if offsets by banks are of insufficient quality relative to developer-led offsets, a large enough market could compensate for the lack of equivalency due to cost-savings from market expansion, and (2) if entry costs are positively correlated with restoration quality, the market could hold banks of low quality, which is an outcome that favors the relative performance of developer-led offsets. We illustrate our results empirically in the case of the US wetland mitigation program. We find that differentials on offset quality and the opportunity cost of land have a clear positive association with the choice of offsetting mechanism. We also find that these differentials are positively associated with the number of credit providers and credit surplus in the mitigation market.

In the third and final chapter, *The Effect of Water Resource Protection on Construction Employment in the United States*, I analyze the economic effects of recent changes in the geographical coverage of the Clean Water Act. The requirement to compensate for impacts on water resources constitutes a substantial compliance cost for regulated firms. Most of the compensation is financed through the compensation credit market embedded within the regulatory program. A major limitation of the scheme is that the short-run credit supply is inelastic. In August 2018, a policy reform increased compensation requirements and, consequently, compensation credit demand in 22 states. I examine the effects of this policy reform on construction activity in a difference-in-differences framework.

In the analysis, I find that the overall effect of the Rule on construction employment was negligible. However, a negative effect appears in the four states that had unsuccessfully litigated against the Rule (Michigan, Ohio, Oklahoma, and Tennessee). Furthermore, the decrease in construction activity was most prominent in counties where low-cost compliance options through environmental offset markets were limited. Specifically, in counties where few compensation credits were available for purchase, the negative effect of the Rule is more pronounced. This finding also coincides with the absence of *in-lieu fee programs* that are funded by public or non-profit entities. These conservation programs function in a very similar manner as private mitigation banks, with the important distinction that they are allowed to release compensation credits in advance of undertaking a conservation project. This feature brings considerable flexibility to the short-term supply of compensation credits. Overall, the results in the final chapter of this thesis suggest that facilitating the functioning of the compensation credit market may significantly alleviate the economic costs of regulation.

Chapter 1

Evaluation of Wetland Area Gains and Losses under the US Clean Water Act

Abstract

Mitigating the impacts of economic development on biodiversity is an urgent global priority. Offsetting policies reconcile development and conservation objectives by allowing environmental losses in some locations, given that the losses are compensated with equivalent gains elsewhere. In this paper, we quantify net losses of wetland area under the US Clean Water Act compensatory mitigation program, which is the most extensive and longest-running environmental offsetting program in the world. A unique feature of the program is how most of the compensation is financed through a market mechanism where permittees purchase compensation credits that specialized firms have generated from wetland conservation activities. We measure wetland area gains at 400 compensation sites over 1995–2020 using high-resolution satellite imagery and land cover change data. Comparing realized compensation projects to planned but withdrawn projects in a difference-in-differences framework, we find that the majority of the gains would not have occurred without dedicated conservation activities. We also find that the market mechanism allocates the type and location of conservation activities according to the opportunity cost of land use. Nonetheless, the wetland area gains appear insufficient to compensate for the wetland area losses regulated within the program. This raises doubts about whether the program will achieve its environmental goals in the long term.

This chapter is coauthored with Jessica Coria, and João Vaz, and Yann Clough.

1 Introduction

The current rate of biodiversity loss is unprecedented in history and will have potentially catastrophic consequences. Humanity depends on the diversity of nature to sustain the quality of air, water, and soil; regulate the climate; reduce the impact of natural hazards; and produce food, through pollination and pest control (Brondizio et al., 2019). Impairments to the productivity, stability, and resilience of natural ecosystems in providing these services adversely affect human well-being and economic growth, with a disproportionate impact on individuals whose livelihoods depend directly on ecosystem services (Dasgupta, 2021). Although national governments widely recognize the need for ambitious conservation policies, progress on policy implementation has not been sufficient to reduce the global rate of biodiversity loss. The main political obstacle is the prioritization of—and unwillingness to compromise on—short-term economic development goals (Johnson et al., 2017; OECD, 2017).

Biodiversity offsetting has emerged as a policy tool for reconciling development and conservation objectives. These policies balance biodiversity losses in one location with an equivalent biodiversity gain elsewhere (Bull and Milner-Gulland, 2020). Offsetting policies typically aim to achieve *no net loss* of the impacted environmental asset. However, there is a lack of empirical evidence regarding the environmental performance of these policies, largely owing to the unavailability of transparent monitoring data that would allow large-scale evaluations (zu Ermgassen et al., 2019).

In this paper, we quantify net losses of wetland area under the US Clean Water Act (CWA) compensatory mitigation program, which is the most extensive and longest-running biodiversity offsetting program in the world (OECD, 2016). Since the early 1990s, the CWA Section 404 compensatory mitigation program has had an overarching goal of "no net loss of wetland area and functions" (Corps and EPA, 1990). A unique feature of the program is how offsetting is financed through a market mechanism. Private firms can establish *mitigation banks*—conservation sites that establish wetland areas or improve the functions of existing wetlands—to provide an advance supply of offsets to meet the compensation needs of future projects. In turn, developers can buy these offsets to comply with their obligation to compensate for the impacts they are causing to wetlands.

To assess whether the market-based offsetting scheme under the CWA is achieving its no-net-loss target, we collect novel data to quantify wetland area gains at the mitigation bank sites within the program. We analyze satellite imagery to identify and delineate the extent of wetland area gains at 400 mitigation banks established between 2001 and 2020. We combine our own data collection with land cover change detection data, enabling us to assess the timing and causality of the wetland area increases. In a difference-in-differences framework, we compare outcomes at the mitigation bank sites against 141 planned mitigation banks that had withdrawn their permit applications. To our knowledge, our study is the first of its kind in two respects: we estimate the causal impact of offsetting projects on environmental outcomes, and we use these estimates to provide a program-wide evaluation of a no-net-loss target.

Mitigation banking currently generates compensation for the majority of authorized impacts under Section 404 of the CWA, with compensation credits awarded for over 100,000 acres of wetlands conserved over the course of the program (Corps, 2022). Despite the apparent benefits, several studies have argued that mitigation banking is falling short of achieving the no-net-loss target. Reasons include unsuccessful compensation projects, mismatches between the compensation and impact type, and questionable causal impact of the compensation projects. Likewise, existing empirical evidence falls short of enabling a comprehensive assessment of net losses. The lack of monitoring data is a major impediment, and most existing studies on mitigation banks have limited sample size and temporal scope (Griffin and Dahl, 2016; Levrel et al., 2017; Tillman et al., 2022).

Our analysis improves on previous studies in several respects. First, our novel data collection enables us to estimate wetland area gains within the mitigation banking program across all jurisdictional areas of the program. Second, unlike previous studies that typically compare outcomes against a reference target, we contrast the observed outcomes against a counterfactual to allow for a causal interpretation of the wetland area gains. Third, we track outcomes over a long time period, 1995–2020, to allow for temporal variability of the success of restoration activities.

Our results indicate that the majority of the observed gains in wetland area would not have taken place without dedicated conservation activities. However, contrasting the gain estimates to the wetland losses that the program is designed to compensate, we find that the program likely results in a net loss of wetland area. Importantly, our estimates are a lower bound on the wetland area net losses. A limitation of our method is that we cannot assess the hydrological and ecological quality of the newly created wetland areas, and, in our calculations, we assume they are equivalent in functionality to the lost wetland areas. This assumption is unlikely to hold (Tillman et al., 2022), rendering the implications of our results even less favorable.

There is considerable heterogeneity in our wetland gain estimates across administrative regions. Such variability is associated with the opportunity cost of land use. Wetland area gains are the highest in regions with an abundance of low-cost agricultural land that can be converted to wetlands. In contrast, in regions with less available agricultural land for conversion, firms choose to earn compensation credits by improving the functions of existing wetlands instead of establishing new wetland areas. While this pattern may appear beneficial from the perspective of efficient land use allocation, it may have adverse implications for ecosystem services in regions where wetland area losses are high.

Evaluating whether the US mitigation banking program is achieving its target of no net loss of wetlands provides insights into the effectiveness of offsetting policies more generally. This knowledge is particularly important considering that biodiversity offsets have now been introduced in a number of countries, despite the lack of knowledge about their environmental performance (OECD, 2016). Our study contributes to improving the evidence base around the effects of one of the most widely used policies for addressing the environmental impacts of development projects.

The remainder of the paper is organized as follows: Section 2 discusses the paper in relation to the existing literature, while Section 3 describes mitigation banking under the US Clean Water Act. In Section 4 we describe the data. In Section 5 we describe the empirical strategy in estimating wetland area gains, and in Section 6 we describe the estimation results. Section 7 discusses the implications of our estimates on whether the program is achieving its no-net-loss target. Section 8 concludes.

2 Relation to Literature

Wetlands are among the most biodiverse and economically valuable ecosystems in the world. They provide a range of critical ecosystem services, including carbon storage, water purification, flood control, and habitat to animal and plant populations (Mitsch and Gosselink, 2000; Moreno-Mateos et al., 2012a). Since many of these services are public goods and the private benefits of wetlands to landowners do not reflect the total benefits to society, there are limited financial incentives for conservation (Heimlich, 1998; Turner et al., 2000). The purpose of regulating the use and conservation of wetlands is to correct this market failure. Recent evaluations report large benefits to society from wetland protection. For instance, Taylor and Druckenmiller (2022) estimate that US wetlands provide up to $2.9 trillion in flood mitigation value alone.

Our paper contributes to the literature investigating the outcomes of ecological offsetting. Previous empirical literature falls short of providing a robust evaluation of offsetting policies and, in particular, whether they achieve no-net-loss goals. There is a clear gap between the increasing global implementation of offsets and the evidence for their effectiveness (zu Ermgassen et al., 2019). Existing studies suffer from a series of limitations. For instance, studies conducting detailed ecological on-site assessments have small sample sizes and focus on a specific region, which reduces the generality of the findings due to potential selection biases (Tillman et al., 2022). Moreover, studies that measure outcomes at a single point in time face a potential performance bias, since it takes at least four years for a mitigation bank to reach its ecological potential, and this period can be even longer for some specific ecological functions (Moreno-Mateos et al., 2012a). Finally, these studies are characterized by a lack of comparability between the analyzed outcomes. Some studies focus on diverse definitions of ecological performance, while other studies focus on administrative performance (i.e., compliance rates) (Tillman et al., 2022).

The results in previous studies are mixed (see reviews in Levrel et al. 2017; Tillman et al. 2022) but provide insights on the factors affecting the success of mitigation banking. Some studies point out mismatches between the permitted adverse impact and the compensation, as well as declining compliance with regulatory standards over time, as factors undermining the performance of mitigation banking.

Concerning mismatches between impact type and offset type, offset projects have primarily focused on the rehabilitation, enhancement, or preservation of existing wetland areas, instead of creating new ecosystems by establishing new wetland areas or re-establishing former wetland areas. Qualitative improvements to existing wetlands are a poor match to compensate for losses in wetland area extent. Still, the use of qualitative improvements as the compensation type has increased over time, while the use of establishment and re-establishment has decreased (Theis and Poesch, 2022). We analyze the spatial and temporal variation in different compensation types to assess the equivalence between impact and compensation at a national scale.

Concerning compliance, the mandatory monitoring period for a mitigation bank is typically only five years, which is not long enough to guarantee successful restoration in the long term (Tillman et al., 2022). Previous research has primarily examined outcomes within or at the end of the monitoring period (Levrel et al., 2017). Multiple environmental stressors (i.e., increased input of nutrients and pollutants and pressure from invasive species) may affect the performance of a compensation site beyond this time frame and possibly compromise the success of the project (Van den Bosch and Matthews, 2017). We evaluate outcomes up to 19 years after bank establishment to evaluate the temporal variability of the success of restoration activities.

Methodologically, our study also relates to recent literature using remotely sensed land cover data products to measure changes in environmental outcomes (Sonter et al., 2019; Taylor and Druckenmiller, 2022). Although land cover data enable low-cost environmental monitoring at a considerable spatial and temporal resolution and scale, these data can suffer from high error rates in detecting land cover change in specific applications (Stehman and Wickham, 2020; Stehman et al., 2021; Wickham et al., 2021). Mapping and monitoring wetlands with algorithm-based data products are particularly challenging and an active research area in the remote sensing literature (Mahdavi et al., 2018). To overcome these caveats, we delineate newly created wetland areas through visual interpretation of high-resolution satellite imagery (Griffin and Dahl, 2016). Combining our own data collection with land cover change detection data provides a reliable measure of the *extent* and *timing* of wetland area gains.

3 Policy Background

3.1 No net loss and compensatory mitigation

The conterminous United States has lost over half of its original wetlands, mostly due to drainage for agriculture, forestry, and urban expansion (Dahl, 1990). As the understanding of wetlands and the importance of their ecological functions improved, in the 1970s, regulatory efforts began to reverse the trend of losses. Currently, US wetlands and streams are protected under Section 404 of the 1972 Clean Water Act (CWA), which provides the US Army Corps of Engineers (Corps) and the US Environmental Protection Agency (EPA) with the authority to regulate threats to the "physical, chemical, and biological integrity" of water bodies in the US.[1] Under Section 404, any development activity that causes adverse impacts to wetlands and streams must secure a permit from the Corps.

In 1990, the Corps and the EPA adopted a goal requiring *no net loss* of wetlands as a guiding principle for evaluating Section 404 permit applications (Corps

[1] 33 US Code § 1251 (101)(a).

and EPA, 1990).[2] Since then, under both federal and state regulations, developers are required to follow a sequence of mitigation steps if a project will impact aquatic resources: (1) reconfigure development sites to avoid impacts, (2) minimize unavoidable impacts, and (3) provide compensation for unavoidable impacts in the form of restoration, establishment, enhancement, or preservation of alternate wetlands and streams.[3]

The last step in the mitigation process has led to the development of three compensation methods, in which *offsets* are provided by (1) mitigation banks, or entrepreneurial firms that invest in restoration projects and sell offsets to permittees, (2) government agencies or nonprofits who collect and pool fees for impacts through in-lieu fee (ILF) programs that later fund restoration projects, or (3) the permittees themselves, a process known as permittee-responsible mitigation (PRM).

In this study, we focus on evaluating the performance of mitigation banking, since it is currently the most widely used implementation method.[4] Several reports have determined that onsite PRM has not been effective in terms of ecological outcomes and have highlighted the high rate of non-compliance (Council et al., 2001; GAO, 2005). In contrast, mitigation banking was deemed to improve the efficacy of wetlands offsets, because (1) it reduces the number of stakeholders responsible for the success of compensatory measures, and (2) it increases the probability of success of compensatory mitigation by promoting large-scale ecological restorations and ensuring that ecological gains occur prior to impacts (thus reducing the risk of temporal losses of wetlands).

Changes in regulatory support for third-party mitigation were promulgated in the 2008 Final Compensatory Mitigation Rule (2008 Rule, Corps and EPA 2008). The 2008 Rule explicitly names mitigation banking as the preferred method of compensatory mitigation, while maintaining a proximity criterion whereby mitigation should occur within the boundaries of the watershed of the impacted wetland. Federal guidelines then signaled a shift toward mitigation performed by third parties at off-site locations, where the biophysical characteristics may be more suitable for the protection of high-quality wetlands.

[2]Early policy documents formulated the goal as "no overall net loss of wetlands functions and values". Later, the language evolved to "no net loss of wetland acreage and function". See Corps and EPA (1990, 2008).

[3]33 US Code § 332.2 defines the different compensation types as follows: (i) Restoration: returning natural/historic functions to a former or degraded aquatic resource. (ii) Establishment: developing an aquatic resource that did not previously exist at an upland site. (iii) Enhancement: improving the functions of an existing aquatic resource. (iv) Preservation: the removal of threat to or preventing the decline of an aquatic resource.

[4]Between 2015 and 2020, mitigation banking was mandated for 51%, ILF for 16%, and PRM for the remaining 33% of required mitigation actions (Corps, 2020).

3.2 Mitigation banking

Mitigation banking was first introduced as a method for compensatory mitigation in the 1990s, following the release of the 1995 Banking Guidance (Corps and EPA, 1995). Mitigation banks have four distinct components: (1) a bank site (the property where wetlands are restored, established, enhanced, or preserved), (2) a bank instrument (the formal agreement between the bank owners and regulators, which establishes liability, performance standards, monitoring requirements, and the terms of bank credit approval), (3) an Interagency Review Team (consisting of all state and federal agencies that provide approval and oversight of the bank, with the Corps as lead), and (4) a service area (the geographic area/watershed or political boundaries in which permitted impacts can be compensated for at a given bank).

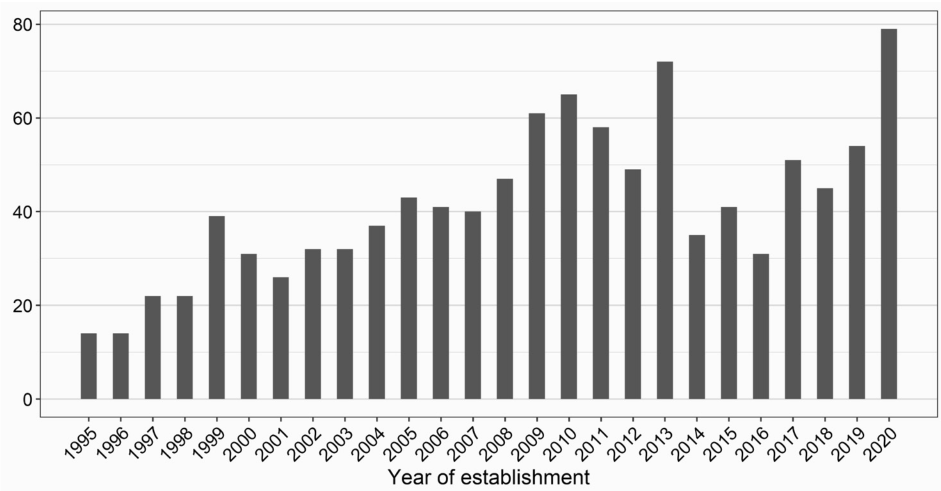
The bank instrument identifies the number of credits available for sale. This determination is based on the ecological value of the compensation project. Once a bank successfully meets its performance standards and monitoring requirements, it receives the specified credits and can sell them to developers. In turn, the developers use the credits to satisfy their compensation obligations.

Since mitigation banking involves the transfer of liability from developer to banker in the face of uncertainty regarding the demand and success of compensatory measures, creating a mitigation bank is a risky venture that requires a high level of specialized expertise. Bankers often need the help of qualified and experienced consultants to navigate the multi-step process. It usually takes from two to five years to secure the initial documentation and approval, and about ten years to complete development (BenDor and Riggsbee, 2011; Levrel et al., 2017). Nevertheless, following a series of institutional responses that supported mitigation banking and reduced some of its economic risks, banking is now the most important mitigation mechanism. Whereas permittee-responsible mitigation represented about 60% of all compensatory measures permitted by the Corps in 2008 (Institute for Water Resources, 2015), it represented only 33% over 2015–2020. By contrast, mitigation banking now represents 51% of all mitigation methods (Corps, 2020).[5]

This increase in banking as a mitigation method has been accompanied by a significant development of the banking industry over the years. Figure 1 plots the counts of established banks over 1995–2020. In 2007, a total of 403 banks with Section 404 wetland credits had been approved, and the rate of approvals aver-

[5] Data on all mitigation methods permitted by the Corps in 2020 were obtained from the Operation and Maintenance Business Information Link, Regulatory Module (ORM), which is the Corps database for tracking Section 404 permitting data. ORM data were obtained through Freedom of Information Act requests.

Figure 1: Annual counts of established wetland mitigation banks



Data source: Corps (2022)

aged about 30 banks per year following the release of the 1995 Banking Guidance (1995–2007). In the 12 years since the 2008 Rule, the rate of approvals has averaged about 53 banks per year (2008–2020), which corresponds to a more than 70% increase relative to the period before. As of December 2020, there were 1091 banks selling wetland credits, and 214 more pending approval.

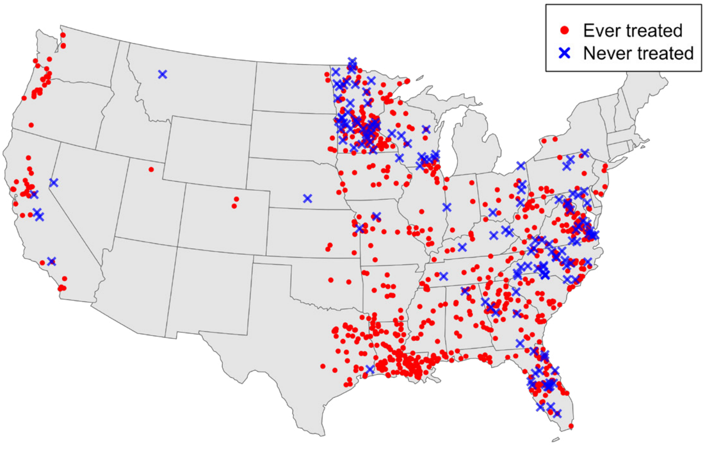
Different states have adopted mitigation banking to varying degrees. For example, in the Northeastern US, state authorities have chosen not to endorse private for-profit mitigation banking and instead rely on non-profit in-lieu fee banking and permittee-responsible mitigation. Conversely, in states such as Louisiana and Minnesota, regulators have endorsed private mitigation banking, with over 94% of impacts permitted over 2012–2020 requiring compensation from mitigation banks (Corps, 2020). In the following section, we describe the distribution of mitigation banks across the US and our estimation sample.

4 Data

4.1 Wetland mitigation banks

We obtain information on mitigation banks from the Regulatory In-lieu fee and Bank Information Tracking System (RIBITS, Corps 2022). The RIBITS database

Figure 2: Estimation sample



Ever treated:	Private wetland mitigation banks established between 2001 and 2020 (N = 952). Outcome data collected for 400 sites.
Never treated:	Candidate mitigation banks that withdrew their application during the permitting process (N = 141)

Data source: Corps (2022)

tracks information on all mitigation banks, including their location, the type and number of credits supplied, and their credit transaction ledger.

We collect data for a sample of 400 banks out of 952 private wetland mitigation banks that were established between 2001 and 2020. Additionally, we collect data for 141 candidate private mitigation bank sites, which constitute a never-treated control group in our analysis. These sites first entered the permitting process to allow operation as a mitigation bank but eventually withdrew from the process before obtaining a permit. Typical reasons for project withdrawals include property ownership disputes and substantial required revisions to the project design due to potential effects on neighboring properties or adjoining water resources.[6] Figure 2 displays the geographic distribution of the wetland mitigation banks included in our estimation sample.

For the purpose of measuring outcomes at the mitigation bank sites, we observe the site polygon for 428 banks. For the remaining 113 banks, we only observe the site centroid coordinates. For these sites, we approximate the site area with a circular buffer around the site centroid such that the buffer area corresponds to the bank area as stated in RIBITS.

[6]Information on reasons for project withdrawals is available in bank documentation in RIBITS.

4.2 Wetland area

We combine two data sources to measure changes in the extent of wetlands at mitigation bank sites. As a first step, we collect novel data by interpreting high-resolution satellite and aerial imagery to delineate newly established wetland areas. As a second step, within the delineated wetland areas, we use land cover change detection data (USGS, 2022) to assess the timing of conversion from cropland to wetland. This process is illustrated in Figure 3 and explained in detail below.

4.2.1 Imagery interpretation

We interpret historical satellite and aerial imagery to delineate the extent of conversion of agricultural land to wetlands within the mitigation bank sites. The analysis relies primarily on observable physical changes that are evident in the imagery and secondarily on the mitigation bank documentation, which provides contextual information such as elevation and soil maps and a description of the permitted conservation project.

Figure 3 provides example imagery of a mitigation bank that was established in 2013. The first image shows how the entire site was in agricultural use in 2009. By 2020, 29% of the site area was converted to wetland, visible in the image as flooding and vegetation change around the flooded areas. To record this change, we draw a polygon around the newly restored wetland plot and calculate its area. We also record the years between which the change is observed. Importantly, for all mitigation bank sites, we confirm that the created or restored wetland area remained intact until the endline year 2020.

Visual interpretation provides an accurate measurement of wetland area at the treated sites at the endline year. In most cases, it is also straightforward to observe that the site was in agricultural use until conversion to a mitigation bank and that the gains do not merely reflect a change in the surrounding landscape, e.g., due to a larger restoration project outside the CWA compensatory mitigation program. However, especially for the sites that were established in the first years in our sample, imagery availability and quality impose constraints on determining the exact timing of the change. First, for over 48% of the examined sites, comparable imagery before and after the treatment year were over five years apart. This causes uncertainty in assessing the exact timing, and thus the additionality, of the changes. Another source of uncertainty is low image quality. Major re-wetting, as visible in Figure 3, is easy to delineate. In contrast, the timing of vegetation change in the absence of visible flooding (30% of examined sites) is difficult to assess without high-resolution imagery. To address these shortcomings, we employ land cover change detection data. These data enable us to measure the timing of change within the recorded wetland polygons at annual intervals.

4.2.2 Land cover change detection

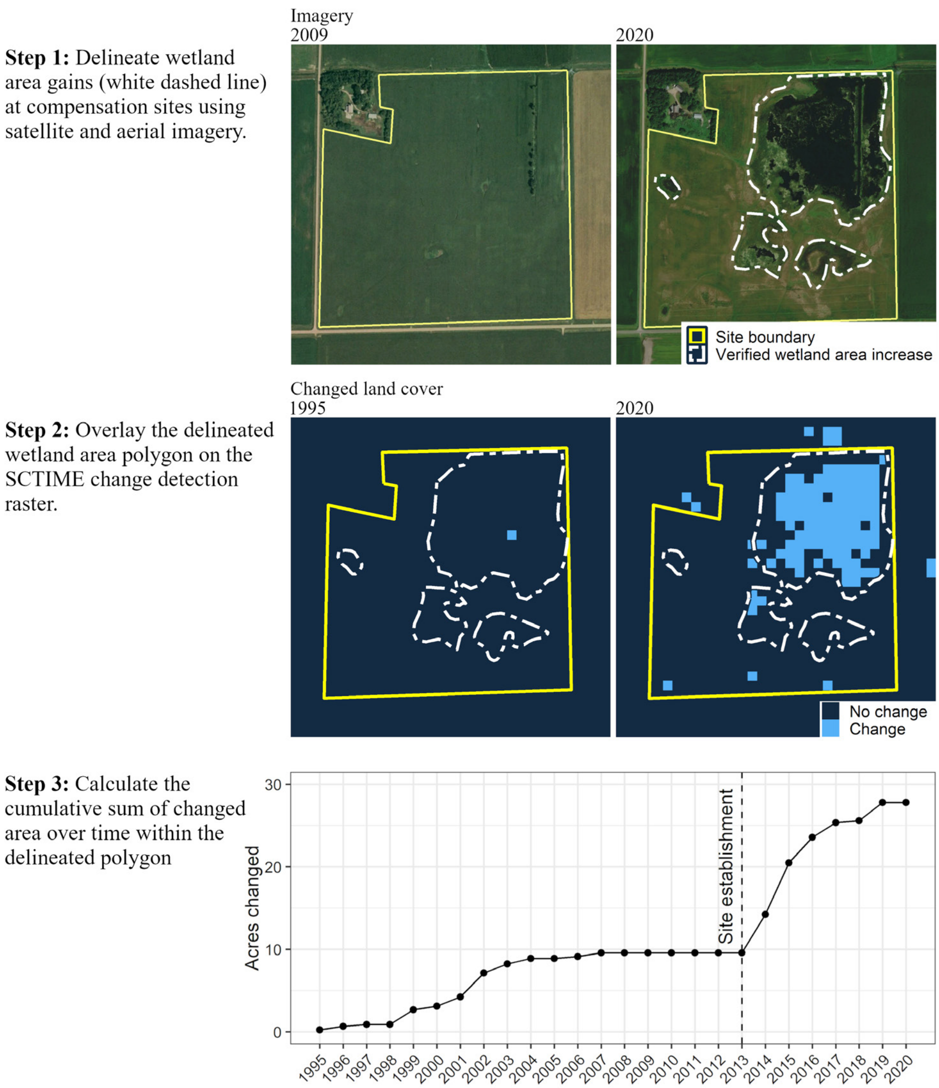
We combine the delineated wetland gain polygons with land cover change detection data to construct an annual time series of wetland gains. To this end, we use the Time of Spectral Change (SCTIME) layer in the LCMAP data suite (USGS, 2022). The SCTIME layer maps land cover change in the continental United States at 30-meter resolution and at annual intervals from 1985 to 2020. Identification of land cover change is based on an algorithm that fits a time series model for the surface reflectance of individual pixels in Landsat satellite imagery. Abrupt and systematic changes in surface reflectance will manifest as a model break, which in turn is identified as land cover change (Dwyer et al., 2018; Xian et al., 2022; Zhu and Woodcock, 2014).

Comparing the imagery and land cover change data in Figure 3 provides an example. The altered surface reflectance of the flooded areas is recorded as land cover change in SCTIME.[7] This example also demonstrates the suitability of SCTIME in our application where we measure wetland area gains on agricultural parcels. Agricultural land, typically monoculture, produces a stable time series of surface reflectance. Conversion to wetland alters the vegetation and may partially inundate the area. The resulting changes in the surface reflectance time series are likely identified as a model break.

Used independently, a major limitation of SCTIME is that it does not identify the type of change or whether the change is permanent. For example, major natural flooding events may be recorded as change although no permanent land cover change took place. Using SCTIME together with the wetland gain polygons resolves this caveat. Within these polygons, we know the land cover type before and after change (cropland and wetland), and SCTIME is a reliable indicator of the timing of the change.

[7] Within the largest delineated polygon, SCTIME records change in approximately 69% of the area over 1995–2020, while approximately 45% of the area changed after site establishment. Vegetation change and flooding within the three smaller polygons remain mostly undetected. This is likely due to the area of changed land cover being too small in comparison to the mapping unit of 30-meter pixels. Alternatively, the changes in the surface reflectance time series may be too small in magnitude in comparison to pre-treatment patterns. The pre-treatment pattern may have been similar, e.g., due to seasonal flooding.

Figure 3: Wetland gain measurement



Data sources: Google Earth (USDA/FPAC/GEO and CNES/Airbus); USGS (2022)

We construct our primary measure of wetland gains as follows:

$$\text{Wetland gain}_{it} = \frac{\text{SCTIME}_{it}}{\sum_t \text{SCTIME}_{it}} \times \text{Verified wetland area}_{i,\ t=2020} \tag{1}$$

where SCTIME_{it} is the recorded cumulative change (acres) within verified wetland polygons at site i in year t. The term $\text{Verified wetland area}_{i,\ t=2020}$ is the area of the polygon that was delineated using imagery. The scaling of SCTIME_{it} corrects for the fact that the extent of the detected change in SCTIME is typically smaller than the extent of the wetland area gains we observe in imagery. This feature is visible in Figure 3. In the SCTIME data from 2020, the area of recorded change (light pixels) is less than the area of delineated wetland gains (white dashed line). In the example, SCTIME succeeds best in detecting a large patch of visible flooding. By contrast, subtle changes in vegetation, as well as changes that are small in comparison to the minimum mapping unit (30-by-30 meter pixel), are less likely to be detected.

To enforce equality between Wetland gain_{it} and the delineated wetland area gain polygon, we divide SCTIME_{it} by the total detected change and multiply by the total verified wetland area. As a result, in the scaled measure it will hold that

$$\text{Wetland gain}_{i,\ t=2020} = \text{Verified wetland area}_{i,\ t=2020}$$

For two sites, no change was recorded in SCTIME over the entire time frame of the analysis, and the denominator in eq. 1 becomes zero. For these sites, we set the gains to zero.

4.2.3 Other data sources

We use county-level data on GDP, population (US Bureau of Economic Analysis) and agricultural land value (US Department of Agriculture) as covariates in regression analysis. We also control for precipitation, available as annual mean in a 1-kilometer resolution grid (Thornton et al., 2020). To describe the landscape characteristics of our estimation sample, we support the analysis with the Primary Land Cover (LCPRI) product in the LCMAP data suite. LCPRI classifies land cover in the continental United States into eight thematic classes. We aggregate these classifications into five categories of interest: wetland, other natural (grass/shrub, tree cover), cropland, developed (artificial land cover such as roads and buildings), and open water. The LCPRI data are available at a 30-meter resolution at annual intervals between 1985 and 2020.

4.3 Data description

Figure 4 depicts the trends of wetland gains over time at the ever-treated and never-treated sites in our estimation sample. The wetland area gains are steady in both groups, although the gains are only about four percent of site area for the never-treated units. Figure 4b shows the average gains in treated units relative to site establishment year, normalized to zero in the year preceding approval. The graph shows a distinct increase in the wetland area starting approximately ten years before bank establishment and continuing as a steady increase throughout the depicted time period.

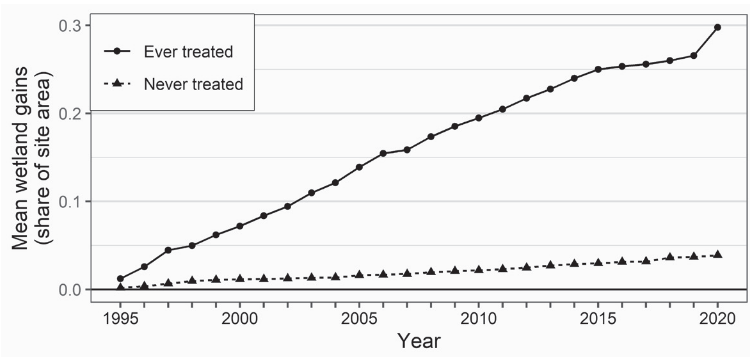
It is important to assess whether the pre-approval changes in the outcome represent an anticipatory treatment effect or whether they would have occurred in the absence of the permit process. As discussed in Section 3.2, the permit process typically takes 2–5 years, and realizing the compensation project can take up to 10 years (BenDor and Riggsbee, 2011; Levrel et al., 2017). Landowners who anticipate a positive administrative decision may start preparatory work prior to approval in order to expedite the completion of the project and the subsequent acquisition of compensation credits. Anticipatory behavior has implications for estimation, as it runs counter to the identifying assumptions in standard difference-in-differences models. We discuss the issue in more detail in Section 5, and we account for possible anticipation in the estimation.

Figure 5 describes the landscape characteristics of our estimation sample using the LCPRI land cover classification map measured in 2000 (USGS, 2022). The full population of wetland mitigation banks and the banks in our estimation sample closely resemble each other in their land cover class distribution. As for the never-treated sites (withdrawn candidate sites), the only distinctive differences are in the shares of wetlands and other natural areas. The never-treated sites had on average 7 percentage points less wetland area than the sampled ever-treated sites, although the combined average share of wetlands and other natural areas is almost exactly equal at 40%.

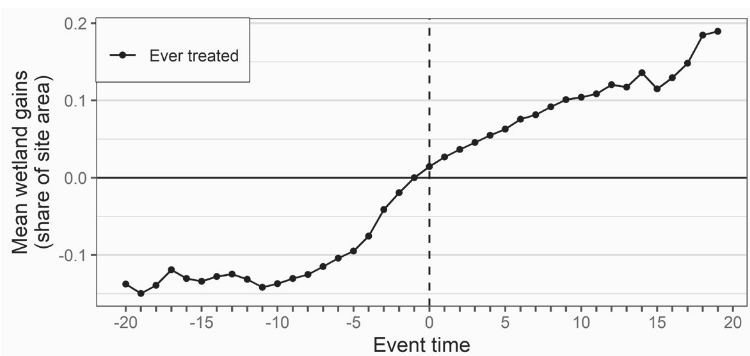
Differences in geographical distribution between the two groups explain the pattern (see Figure 2). For example, none of the never-treated sites are located in Louisiana, which is abundant in wetlands. Another explanation relates to the different types of conservation activities. Most wetland creation and re-establishment take place on agricultural land, whereas wetland enhancement, by definition, takes place on existing wetlands. Since wetland creation and re-establishment are generally more costly than enhancement, these projects are more likely to be withdrawn during the permitting process. Regardless, for the purposes of using the withdrawn candidate sites as a never-treated control group,

Figure 4: Outcome trends

(a) Wetland area gains in estimation sample over 1995–2020.



(b) Wetland area gains at ever-treated sites relative to bank establishment year.



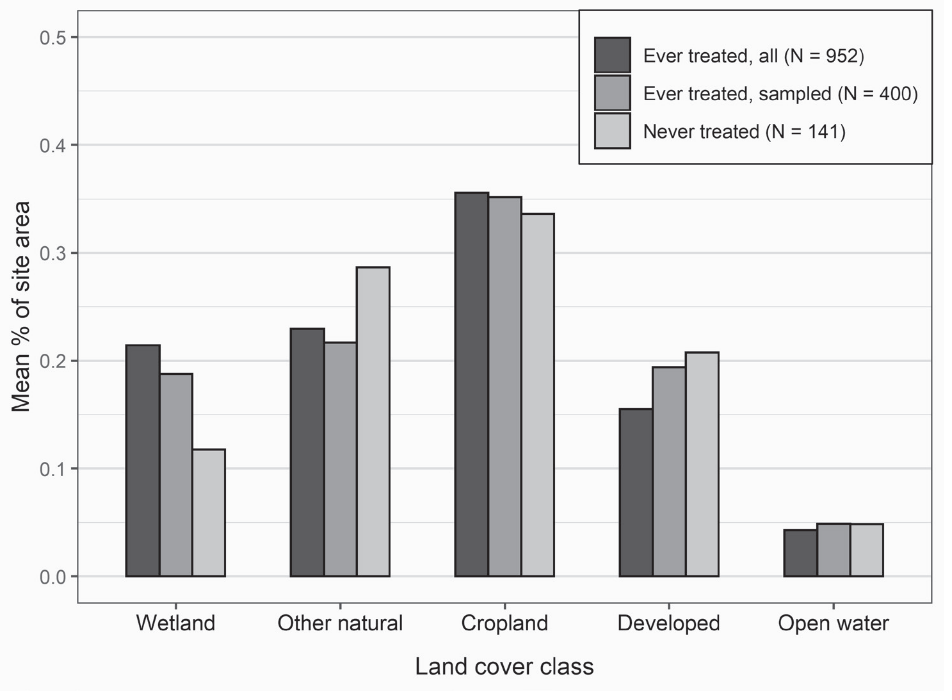
Panel (a): Wetland area gains at ever-treated and never-treated sites in estimation sample over 1995–2020. Wetland gains are defined as in eq. (1), and normalized by mitigation bank site area.

Panel (b): Wetland area gains as in (a) in ever-treated units with time period normalized to zero in the bank establishment year and level normalized to zero in the year preceding approval.

Ever treated: Sampled private commercial wetland mitigation banks established during 2001–2020 (N = 400). Never treated: Candidate mitigation banks that withdrew their application during the permitting process (N = 141).

Data sources: Own data collection, Corps (2022); USGS (2022).

Figure 5: Landscape characteristics of mitigation banks



Land cover class measured in 2000.

Data sources: Corps (2022); USGS (2022).

it is important that they have comparable potential for wetland creation and re-establishment. This potential is best reflected in the baseline cropland area, which is roughly equal across groups.

Table 1 shows descriptive statistics of the covariates used in the analysis. Note that the covariates are measured at an aggregate level: land value, GDP per capita, and population density are recorded at the county level, whereas precipitation is measured within a 1-kilometer grid cell. Overall, the counties where the mitigation bank sites are situated resemble each other across groups.

Table 1: Descriptive statistics

	Ever treated		Never treated
Variable	All	Sampled	
Ag. land value (USD / acre)			
Mean	2138.9	2066.2	2031.9
Std. dev.	1771.3	1326.3	1198.9
GDP per capita (1000 USD)			
Mean	37.8	37.8	34.9
Std. dev.	21.2	19.7	13.7
Population/km^2			
Mean	92.2	95.4	87.0
Std. dev.	154.8	155.9	147.5
Precipitation (mm/year)			
Mean	1035.5	1007.5	957.6
Std. dev.	253.6	247.6	242.6
N	952	400	141

Summary statistics of covariates at mitigation bank sites. Agricultural land value, GDP per capita, and population density measured at the county level. Precipitation measured within a 1-kilometer grid cell. Variables are measured in the year 2000, except land value, which is measured in 1997.

Data sources: US Bureau of Economic Analysis, US Department of Agriculture, and Thornton et al. (2020).

5 Empirical strategy

We model the effect of mitigation bank establishment on wetland area using a long differences model as follows:

$$\Delta y_i = \tau D_i + \Delta X_i \beta + \varepsilon_i \tag{2}$$

where Δy_i is the wetland area gained at site i between 1995 and 2020, measured as a share of the total site area. These area gains correspond to the wetland area delineated using imagery as shown in Figure 3. D_i indicates the treatment group, and X_i is a vector of control variables that include GDP, population, land value, and precipitation.

We also estimate the dynamic path of treatment effects and treatment effects specific to treatment cohorts according to the following models:

$$y_{it} = \sum_{s \in \mathcal{S}} \tau_s D_{its} + X_{it}\beta + \alpha_i + \gamma_t + \varepsilon_{it} \tag{3}$$

$$y_{it} = \sum_{g \in \mathcal{G}} \tau_g D_{itg} + X_{it}\beta + \alpha_i + \gamma_t + \varepsilon_{it} \tag{4}$$

where, y_{it} is the wetland area gained at site i in year t. These annual area gains correspond to the land cover change data as shown in Figure 3 and as defined in equation 1. In equation (3), D_{its} is an indicator equal to one if year t is s years after site i was first treated and, in equation (4), D_{itg} is an indicator equal to one if site i belongs to treatment cohort g (was first treated in year g) and was treated in year t.

Recent literature has shown that OLS is unsuitable for estimating treatment effect parameters in two-way fixed effects models, such as in equation (2), when treatment adoption is staggered and treatment effects are heterogeneous over time or treatment cohorts (Borusyak et al., 2021; Callaway and Sant'Anna, 2021; De Chaisemartin and d'Haultfoeuille, 2020; Gardner, 2022; Goodman-Bacon, 2021; Roth and Sant'Anna, 2021; Sun and Abraham, 2021). As regulatory practices have evolved considerably over the last two decades, this type of heterogeneity is likely in our setting. Accordingly, we report estimates from various alternative estimators suggested in the literature.

The identification of treatment effect parameters in equations (2)–(4) requires treatment assignment to be mean-independent of variables that affect changes in the outcome (*parallel trends*). As a never-treated control group, we use candidate mitigation bank sites that entered the permitting process but eventually withdrew their applications. As discussed in Section 4.3, these candidate sites are similar to our treated sites in terms of their suitability for conversion to wetland and the opportunity costs of different land uses. Their outcome trends arguably reflect the counterfactual outcome trends of the treated sites.

A second identifying assumption requires that there is no treatment effect in pre-treatment periods (*no anticipation*). As depicted in Figure 4b, the wetland area gains at the treated sites are largest during the five years immediately before the administrative bank establishment date. It is important to distinguish whether this pattern should be interpreted as treatment anticipation or endogeneity.

Anticipation is a reasonable interpretation if the economic agents in question have information on future treatment and if there is a benefit to acting before treatment (Malani and Reif, 2015). Both of these characterizations are applicable in the context of our study. Submitting a permit application amounts to information on future treatment, although an approval is not always a certainty. If a bank owner perceives the probability of an approval as sufficiently high, they have an incentive to start the restoration activities at the bank site. This will give the owner earlier opportunities to secure compensation credits and earn revenue from selling them. Therefore, anticipatory behavior is a likely explanation for the observed changes prior to site establishment.

To ensure that treatment anticipation does not confound our estimates, we follow Butts and Gardner (2021); Rambachan and Roth (2019) in their suggestion to adjust the treatment date back in time such that anticipatory effects do not occur before the *adjusted* treatment date. As discussed in Section 3.2, securing a permit for a mitigation bank site typically takes up to five years (BenDor and Riggsbee, 2011; Levrel et al., 2017). Accordingly, we adjust the treatment date back five years and assume that no treatment anticipation existed before that. We also report estimates from alternative adjustments.

6 Results

Long differences. Table 2 presents estimates of τ in eq. (2) across different model specifications. The first model without covariates and state-level fixed effects indicates that, on average, the share of wetland area at a site increased by 0.26 (95% CI: 0.21–0.32), or 26 percentage points, due to mitigation bank establishment. Including covariates does not modify the estimate, but including a state-specific trend decreases the estimate slightly to 0.24 (95% CI: 0.18–0.31). This estimate translates to 19,080 acres (95% CI: 14,310–24,645) of wetland area gains over the estimation sample. The estimated effect is slightly less than the 20,054 acres of wetland area gains that we identified directly in imagery, although the direct measurement is well within the 95% confidence interval of the estimate. We discuss the timing of these gains in the following.

Table 2: The effect of mitigation bank establishment on wetland area. Long differences estimates.

	(1)	(2)	(3)	(4)
Treated	0.26	0.26	0.25	0.24
	(0.03)	(0.03)	(0.03)	(0.03)
Precipitation		0.00		0.00
		(0.00)		(0.00)
Population density		0.00		0.00
		(0.01)		(0.01)
GDP		−0.01		0.02
		(0.03)		(0.03)
Land value		−0.01		0.03
		(0.03)		(0.03)
State FEs			✓	✓
N	541	541	541	541

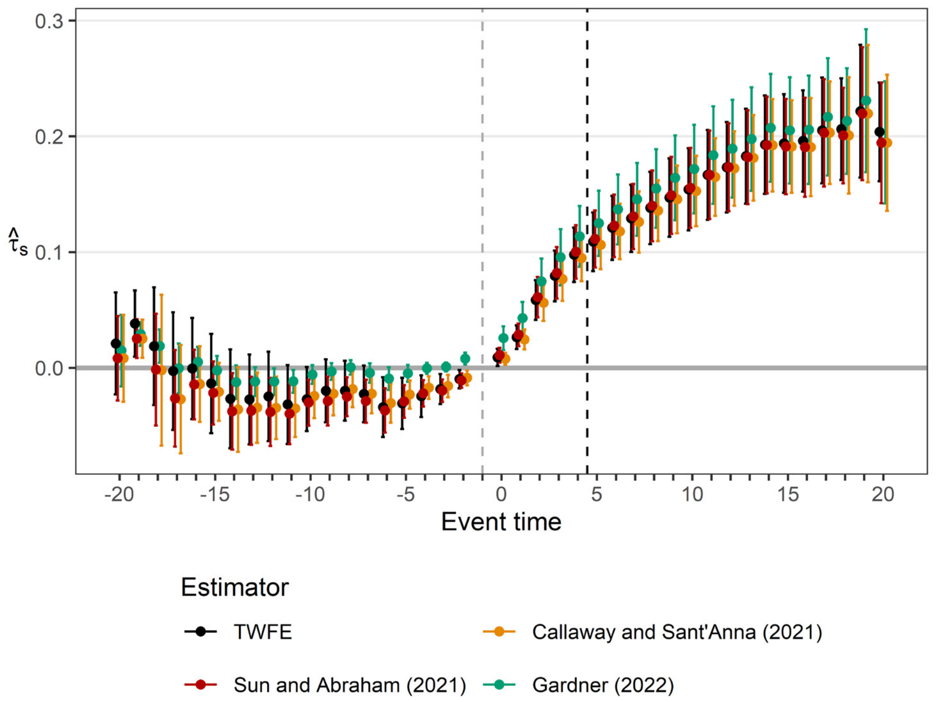
Standard errors in parentheses, clustered by county (N = 272).

Dependent variable:	Wetland area gains at mitigation bank site (share of site area)
Treatment:	Mitigation bank established
Ever treated units:	Private wetland mitigation banks established in 2001–2020 (N = 400)
Never treated units:	Candidate private wetland mitigation bank sites (N = 141)

Dynamic estimates. Figure 6 presents the estimates of τ_s in equation (3), estimated without including covariates. Note that, to prevent treatment anticipation from confounding the estimates, we have adjusted the treatment date back five years; the period enumeration follows the *adjusted* relative years. The adjusted relative period 5 is the date of mitigation bank establishment as stated in the administrative data, and the adjusted relative period 0 is the first treatment year in the estimation.

The dynamic pattern is similar across estimators. The pre-treatment estimates using Gardner (2022) are close to zero and are precisely estimated. In turn, for TWFE, Sun and Abraham (2021) and Callaway and Sant'Anna (2021), the pre-treatment estimates average approximately -0.04 and there is a monotonic in-

Figure 6: The effect of mitigation bank establishment on wetland area. Dynamic estimates.



Point estimates of τ_s in eq. (3) and 95% pointwise confidence intervals. Standard errors clustered by state.

Dependent variable:	Wetland area gains (share of site area)
Treated units:	Private wetland mitigation bank sites (N = 400)
Control units:	Candidate private wetland mitigation bank sites (N = 141)
Dashed line, black:	The relative period 5 in the graph is the bank establishment year as stated in the administrative data. Due to possible treatment anticipation, treatment year is adjusted five years back in the estimation.
Dashed line, grey:	Reference period and the last untreated period in estimation.

crease from relative period -6 to treatment onset. The patterns in post-treatment estimates are also similar across estimators. Estimates for the relative periods 0 to 4 can be interpreted as anticipation effects, reaching approximately 0.1 before treatment onset. A gradual increase in the point estimates continues, with the estimates reaching approximately 0.20 in the last relative periods. Overall, the pattern in the dynamic estimates strongly suggests that the estimated post-treatment gains would not have occurred in the absence of treatment.

The dynamic estimates show that the gains in wetland area have been gradual. Thus, when ascertaining the magnitude of the effect, it is sensible to consider the dynamic estimates only for the last periods. Aggregating the estimates from Sun and Abraham (2021) for the last five relative periods provides a point estimate of 0.20 (95% CI: 0.16–0.24). This corresponds to a 20 percentage point increase in the share of wetland area at a site due to mitigation bank establishment. In total area gains, this estimate translates to 15,900 acres (95% CI: 12,720–19,080) of wetlands at the ever-treated sites summed over the estimation sample.

The estimation results are not sensitive to the choice of estimator. We provide further sensitivity analysis in Section A in the Appendix where we present the dynamic estimates from using different adjustments to the treatment year.

Cohort estimates. Figure 7 presents treatment effect estimates by treatment cohort (τ_g in equation 4). Again, the estimates are very similar across estimators, with the exception of the TWFE estimates for the cohorts 2019–2020. For these cohorts, no increases in wetland area were recorded, although some sites showed signs of conversion activity taking place. This is because any wetland gains are unlikely to be visible in imagery only few years after treatment.

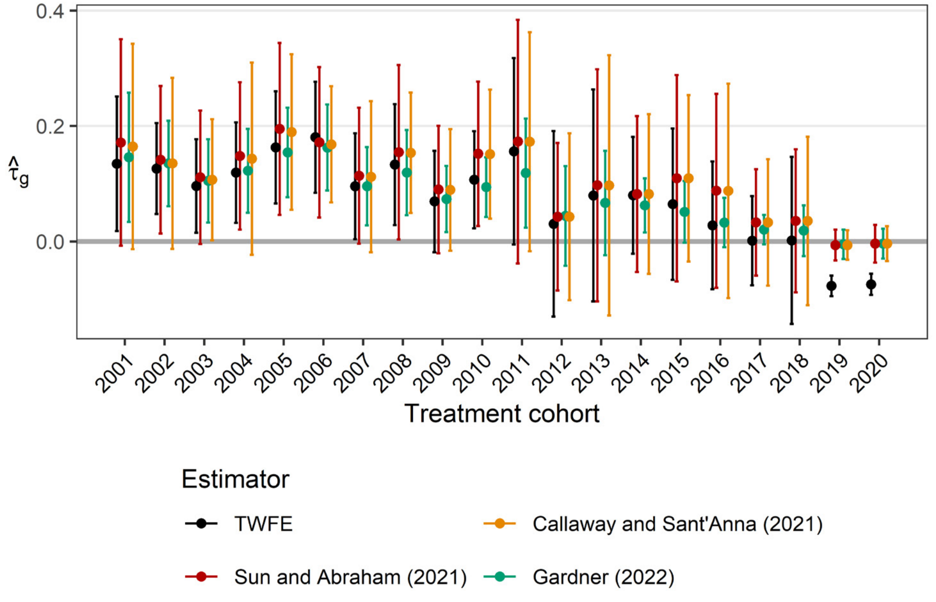
The higher estimates for the earlier cohorts may indicate greater availability of agricultural land with low cost of conversion to wetland. These types of locations will be picked first in the early years of the program. In later years, as locations with low-cost conversion potential become more scarce, mitigation firms increasingly start to generate credits from other types of compensation activities.

Heterogeneity by region. Figure 8 presents the point estimates of treatment effects by the administrative districts of the US Army Corps of Engineers. We produce the estimates only for districts with five or more ever-treated sites in the full estimation sample. The estimates display heterogeneity, ranging from close to zero estimated gains in some districts to up to 0.60 in others.[8] Panel (b) in

[8]Negative estimates imply that the delineated wetland area gains were zero or close to zero in the district.

Figure 8 shows how the heterogeneity is associated with the opportunity cost of land use. The estimates are generally higher in regions where low-cost agricultural land was available. Further heterogeneity analysis in Appendix C shows that wetland area gains have occurred mostly in agricultural landscapes.

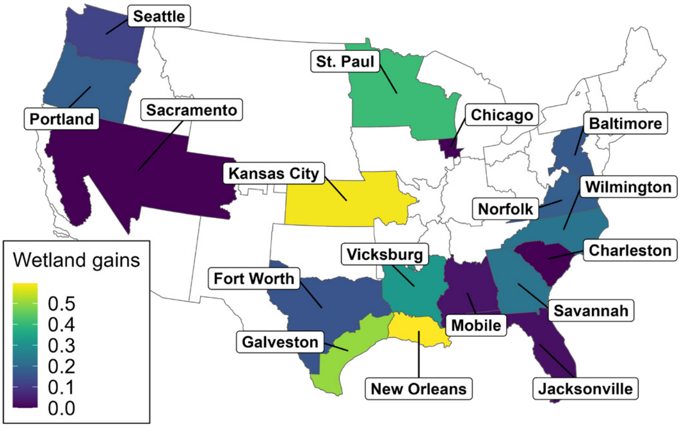
Figure 7: The effect of mitigation bank establishment on wetland area. Cohort estimates.



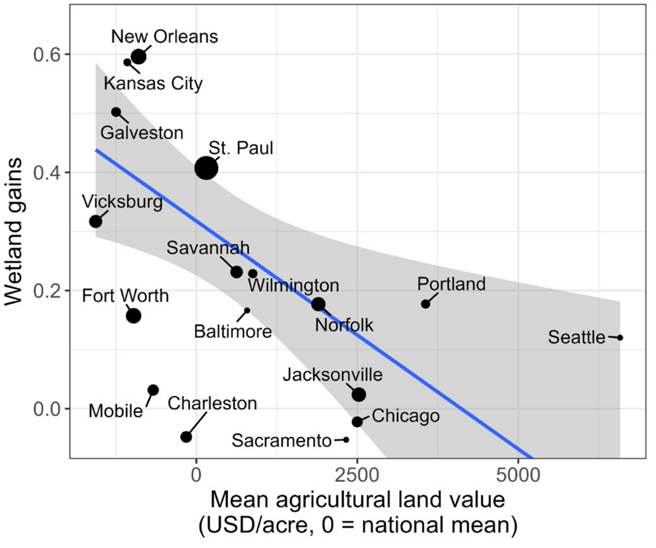
Point estimates and pointwise 95% confidence intervals of τ_g in eq. (4) by treatment cohort. See the notes in Figure 6 for further details on estimation.

Figure 8: Heterogeneity by region

(a) Heterogeneity of wetland area gains by the administrative districts of the US Army Corps of Engineers



(b) Land value and heterogeneity of wetland area gains



Panel (a): Long differences treatment effect estimates by region (estimates of τ in eq. (2)). Only including regions with five or more ever-treated sites.

Panel (b): Treatment effect estimates are as shown in panel (a). Agricultural land value is measured in the county of the mitigation bank site, in the year of bank establishment, and demeaned against the national mean in that year. The fitted line and 95% CI are based on a linear regression with weights (point sizes) corresponding to the number of ever-treated sites.

7 Policy implications

Given our estimates of wetland area gains, we can assess whether the compensatory mitigation program is achieving no net loss of wetland area. Recall that compensation credits are awarded both for wetland area gains and for improving the functions of existing wetland areas. Using the estimates of wetland area gains, we can estimate the average share of wetland area gains out of the total wetland area that was used for compensating losses. In turn, this quantity allows us to assess whether the compensatory program as a whole is succeeding in offsetting wetland area losses with gains. The details of this calculation are given in Appendix B. We display the results for this calculation from using the point estimates from the long differences estimator and by aggregating the last five dynamic estimates reported in Figure 6 using the estimator from Sun and Abraham (2021).

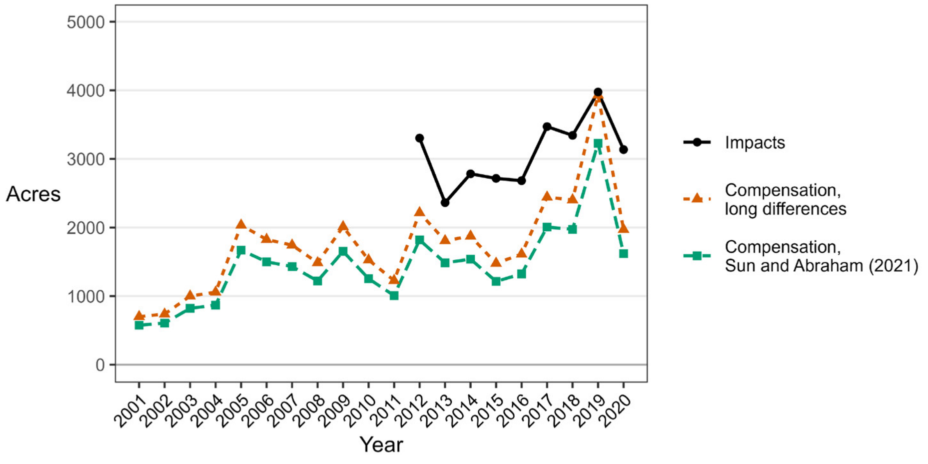
In Figure 9, the resulting estimated wetland area gains embedded in compensation transactions are depicted together with the wetland area losses they were supposed to compensate. Even the upper bound of the gain estimates stays well below the total impacts, indicating net loss of wetland area. Over 2012–2020, using the long differences estimator, the estimated net loss is on average 893 acres per year (confidence bounds according to the 95% CI of $\widehat{\tau}$: 289–1497) Using the estimator from Sun and Abraham (2021), the estimated net loss is on average 1285 acres per year (confidence bounds according to the 95% CI of $\widehat{\tau}$: 924–1645). Heterogeneity in the estimated net losses is depicted in Figure D.1 in the Appendix, showing how the total net losses appear to be driven by only few regions. Moreover, the gains are driven by available agricultural area for mitigation and losses are driven by the amount of wetland areas at baseline.

A net loss of wetland area does not directly imply a net loss of ecological or hydrological functions of wetlands. Instead, the area net loss estimates can be interpreted as a *quality gap*: To achieve no net loss of wetland functions, the area net losses must be offset with qualitative improvements. At the same time, a question is whether the ratio of gains and losses is sufficiently high to maintain wetland functionality. For example, as a guideline, the existing regulations call for a minimum of a one-to-one acreage compensation ratio in the absence of any functional assessment.[9]

To address the gap, there are readily available regulatory solutions both on the demand and the supply side. A demand-side solution is to require more compensatory credits per impacted acre of wetland. Looking at Figure 9, had the

[9] 33 US Code § 332.3 (f). See also Corps and EPA (1990)

Figure 9: Net loss of wetland area in the Clean Water Act compensatory mitigation program



Impacts:	Wetland loss from impacts that required compensation from private mitigation banks. Only including impacts that would be included in administrative no net loss calculations. (Source: Corps 2020)
Compensation:	Estimated acres of total wetland area gains embedded in compensation transactions. (Source: own estimates and RIBITS)

regulator doubled the compensation requirements, the gap could have approximately closed.[10] On the supply side, the regulator may mandate more wetland establishment and re-establishment (area gains) instead of wetland enhancement. This can be achieved by adjusting the ratios according to which different types of compensation activities are awarded compensation credits.

We want to emphasize that our net loss estimates are based on an optimistic scenario. Our calculation assumes functional equivalence between the wetland area that was created as compensation and the wetland area that was lost due to impacts. The degree to which this assumption is fulfilled is pivotal to the success of the program. Yet, this aspect of compensatory mitigation remains a contested topic in ecological literature (Levrel et al., 2017; Tillman et al., 2022).

8 Conclusion

In this paper, we estimate the causal impact of environmental offsetting activities on wetland area extent at mitigation banking sites within the US Clean Water

[10]This is assuming that, on the supply side, the ratio between area gains and improvements on existing wetlands would have stayed the same in the face of increased credit demand.

Act Section 404 compensatory mitigation program. Using these estimates, we evaluate the degree to which wetland mitigation banking is achieving its stated goal of no net loss of wetland area.

Our analysis provides three major findings. First, we measure wetland area gains amounting to over 19,484 acres of established or re-established wetlands at the mitigation bank sites in our sample. Comparing these gains to those at planned but withdrawn project sites, our upper estimate of the causal impact of compensation site designation totals 19,080 acres (95% CI: 14,310–24,645). In other words, the majority of the gains would not have occurred without dedicated compensation activities.

Our second finding is that there is significant heterogeneity in the wetland area gains across administrative districts and that the opportunity cost of land use is likely the strongest driver of these differences. Wetland area increases are the greatest in regions with an abundance of low-cost agricultural land that can be converted to wetlands. In contrast, in regions where less agricultural land is available for conversion, compensation firms choose to earn credits by improving existing wetlands instead of converting agricultural land to wetlands. This suggests that the market-based offsetting scheme is functioning as intended in how it allocates land use, both between mitigation methods and between conservation and other land uses.

Third, we compare the estimated wetland area gains to the area losses for which they are compensating. Our results indicate that the mitigation banking program is resulting in a net loss of wetland area. This is a consequence of how the regulations allow a particular type of mismatch between compensation and impact: it is possible to compensate losses of wetland areas with qualitative improvements and preservation of existing wetland areas. When firms consider the choice of mitigation method, the opportunity cost of converting agricultural land to wetlands is arguably higher than the cost of improving existing wetlands that likely have few other profitable uses. In order to achieve the regulatory mandate of no net loss of wetland area, the regulator could incentivize wetland area increases (relative to qualitative improvements) by awarding more compensatory credits per acre of wetland created. Alternatively, the regulator could mandate permittees to surrender more compensation credits per impacted acre.

The most important limitation of our study is that we do not assess the functional quality of the created wetland areas. Instead, for the purposes of our calculations, we assume qualitative equivalence between a created and a lost acre of wetland. This assumption likely results in overstating the estimated environmental benefits and understating the estimated net losses. Further research should scrutinize the functional quality of both the compensation wetlands and the lost wetland areas.

Overall, our analysis contributes to a better understanding of the functioning of market-based environmental offsetting. While the mitigation banking program under Section 404 of the US Clean Water Act has succeeded in several aspects, our analysis identifies design choices that have resulted in net losses of wetland areas. It is important to consider these dynamics in the design of other existing and future offsetting policies. As in several studies before us, we want to emphasize the importance of transparent and temporally consistent monitoring data in making informed decisions about policy design. Producing such data should be a priority when implementing environmental offsetting schemes.

A Sensitivity

Outcome coding. Figure A.1 presents the regression results with the outcome coded as absolute acres of wetland area instead of normalizing by site area. In the main results, we chose to normalize the outcome because the average size of mitigation banks varies considerably across cohorts and regions. If the outcome is not normalized, the results in our heterogeneity analysis merely reflect the variability in mitigation bank site area. This heterogeneity would not be informative of our ultimate parameters of interest: the share of wetland area gains out of all types of compensation activities.

Nonetheless, when coding the outcome as absolute acres, the dynamic estimates display a pattern that is very similar to the pattern with the normalized outcome. The TWFE estimator is an exception in showing a stronger pre-treatment trend.

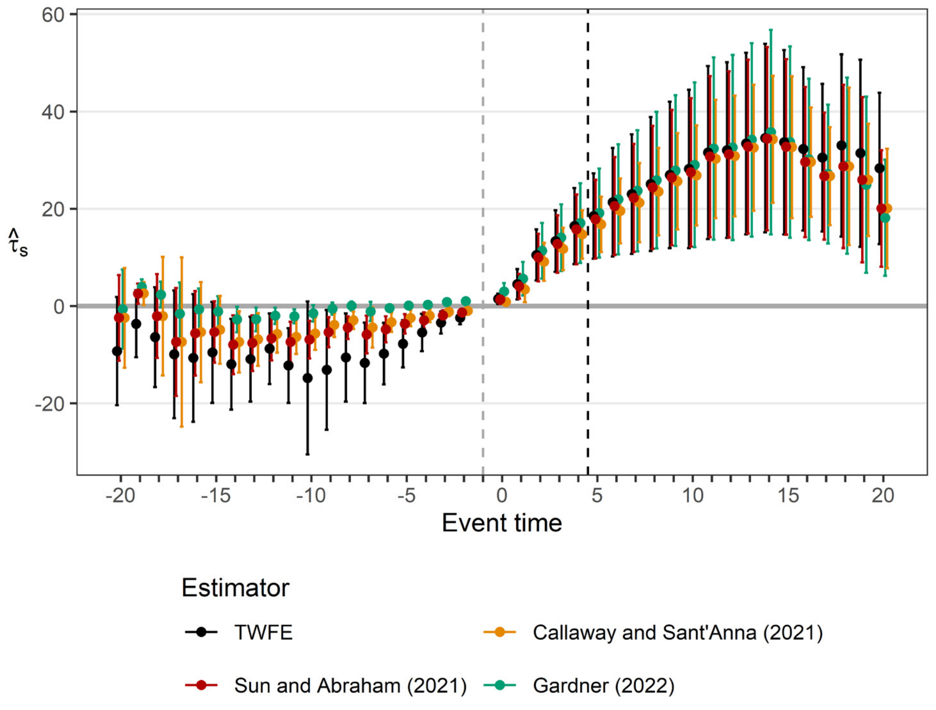
Treatment year and anticipation. The second robustness check concerns the definition or treatment year in estimation. In our main results, we chose to adjust the treatment year in estimation to five years before mitigation bank site establishment. The purpose was to account for possible treatment anticipation that would run counter to the identifying assumptions of our estimators.

Figure A.2 presents the estimation results when defining the treatment year as the site establishment year. The estimates show a pattern similar to the main results, in that the largest gains seem to have occurred during the first five years *before* site establishment. However, the estimates diverge substantially between estimators during the earliest pre-treatment periods.

Figure A.3 presents the results when defining the treatment year as 10 years before site establishment. In these estimates, the relative periods between 0 and 9 can be interpreted as anticipation effects. Here, a more stable and monotonic pre-treatment trend emerges, although the estimates increase the most during periods 5-9. Note that in this regression we lose the ever-treated sites from the five earliest cohorts; this happens because, with the treatment year adjusted 10 years back, they no longer have observed non-treated periods. This reduction in sample size explains the variance of the estimates in the last relative periods.

The estimates for the last five treated periods are higher than our preferred estimates in Figure 6. However, it is questionable whether the slightly increasing trend in relative periods 0–4 should be interpreted as an anticipatory effect. Considering the trend as endogeneity and deducting the projected increase (projection based on the rate of increase during relative periods 0–4) from the post-treatment estimates yields an aggregate estimate of 0.17 (95% CI: 0.12-0.22) in the last five relative periods.

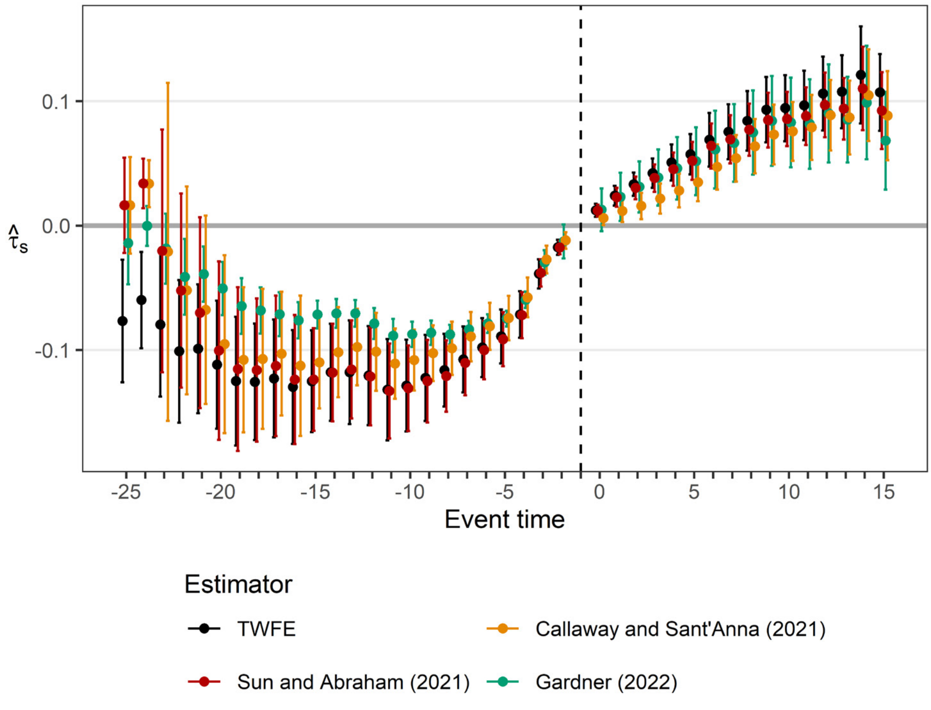
Figure A.1: Dynamic estimates. Outcome coded as acres of wetland area gains.



Point estimates of τ_s in eq. (3) and 95% pointwise confidence intervals. Standard errors clustered by state.

Dependent variable:	Wetland area gains (acres)
Treated units:	Private wetland mitigation bank sites (N = 400)
Control units:	Candidate private wetland mitigation bank sites (N = 141)
Dashed line, black:	The relative period 5 is the year of site establishment.
Dashed line, grey:	Reference period and the last untreated period in estimation.

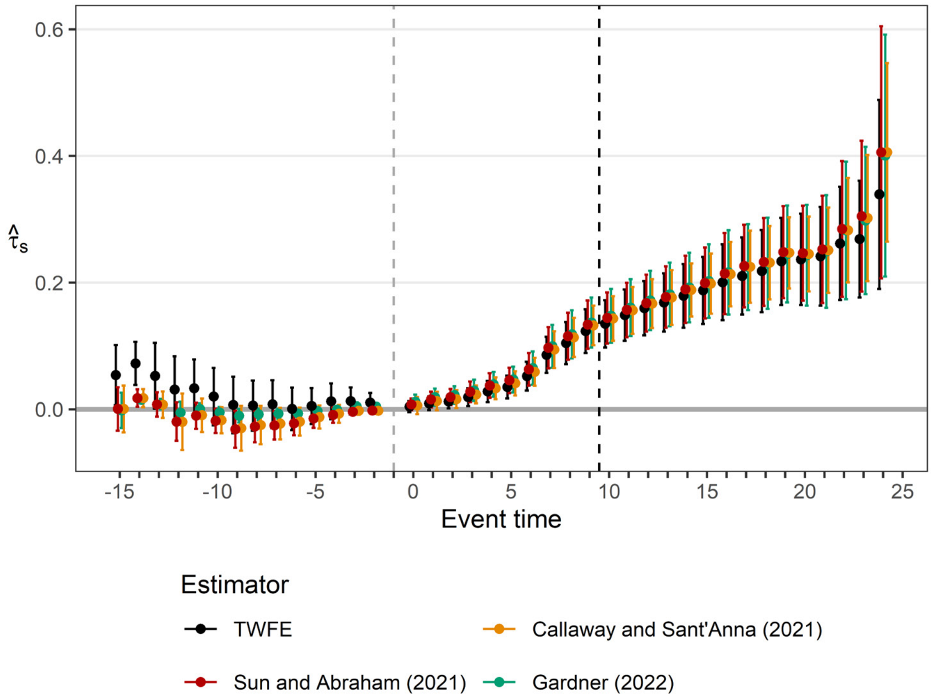
Figure A.2: Dynamic estimates. Treatment year in estimation = site establishment year.



Point estimates of τ_s in eq. (3) and 95% pointwise confidence intervals. Standard errors clustered by state.

Dependent variable:	Wetland area gains (share of site area)
Treated units:	Private wetland mitigation bank sites (N = 400)
Control units:	Candidate private wetland mitigation bank sites (N = 141)
Dashed line, black:	Reference period and the last untreated period in estimation. Relative period 0 is the year of site establishment.

Figure A.3: Dynamic estimates. Treatment year in estimation = site establishment year − 10 years.



Point estimates of τ_s in eq. (3) and 95% pointwise confidence intervals. Standard errors clustered by state.

Dependent variable:	Wetland area gains (share of site area)
Treated units:	Private wetland mitigation bank sites (N = 307)
Control units:	Candidate private wetland mitigation bank sites (N = 141)
Dashed line, black:	The relative period 10 is the year of site establishment.
Dashed line, grey:	Reference period and the last untreated period in estimation.

B Net loss of wetland area

This section describes the procedure to calculate net loss of wetland area, as discussed in Section 7 and presented in Figure 9.

We want to calculate net loss of wetland area within the compensatory mitigation programs as follows:

$$A_t^L - A_t^G \tag{5}$$

where

$$\begin{aligned} A_t^L &= \text{Wetland area loss in year } t \\ A_t^G &= \text{Wetland area gains that were used to compensate for } A_t^L \end{aligned}$$

We observe A_t^L in the ORM data (Corps, 2020). We do not directly observe A_t^G, but we can estimate it based on our empirical results.

B.1 Impacts

We observe wetland losses, A_t^L, in the ORM2 data over 2012–2020. Data for years 2014 and 2015 are incomplete and total impacts for those years are projected. We only include the impacts that fulfill all of the following criteria:

1. The impact results from a discharge of dredged or fill material (regulated under CWA Section 404).

2. The impact results in a permanent loss of the aquatic resource.

3. The impact is compensated exclusively through private mitigation banks. (For impacts that are compensated through a combination of permittee-responsible mitigation, in-lieu fees and/or mitigation banks, we do not observe how many acres of lost wetland were compensated by each compensation method. Excluding these impacts will adjust the impact estimates slightly downward.)

B.2 Compensation

Let

$$A_t^G = \theta \cdot \overline{A}_t \tag{6}$$

where

$$\overline{A}_t = \text{Total compensation area, inclusive of all compensation activities.}$$
$$\theta = \frac{A_t^G}{\overline{A}_t} \; \forall \; t$$

The parameter θ represents the share of area gains out of the total area that was used to compensate for losses. We observe $\overline{A}_t$ in the credit ledger but we do not observe θ.

Above, θ is expressed in terms of acres embedded in credits that were used to compensate for impacts. Assume that we can also express θ in terms of *acres embedded in the credits that have been released to the banks but not yet used for compensation*. Then,

$$\theta = \frac{A_{rel}^G}{\overline{A}_{rel}} \tag{7}$$

where

$$A_{rel}^G = \text{Wetland area gains for which banks received compensation credits}$$
$$\overline{A}_{rel} = \text{Total area for which banks received compensation credits}$$

In other words, $\overline{A}_{rel}$ includes wetland area gains and enhanced wetland area. We observe $\overline{A}_{rel}$ in the credit ledger and we can estimate A_{rel}^G as follows:

$$\widehat{A_{rel}^G} = \widehat{ATT} \cdot \sum_i F_i \tag{8}$$

where $\widehat{ATT}$ is the estimated average percentage point increase in wetland area at a bank site. F_i is the area of the footprint of bank i (the property or conservation easement area).[11]

With an estimate for A_{rel}^G, we also have an estimate for θ and A_t^G:

$$\widehat{\theta} = \frac{\widehat{A_{rel}^G}}{\overline{A}_{rel}} \tag{9}$$

$$\iff \widehat{A_t^G} = \widehat{\theta} \cdot \overline{A}_t \tag{10}$$

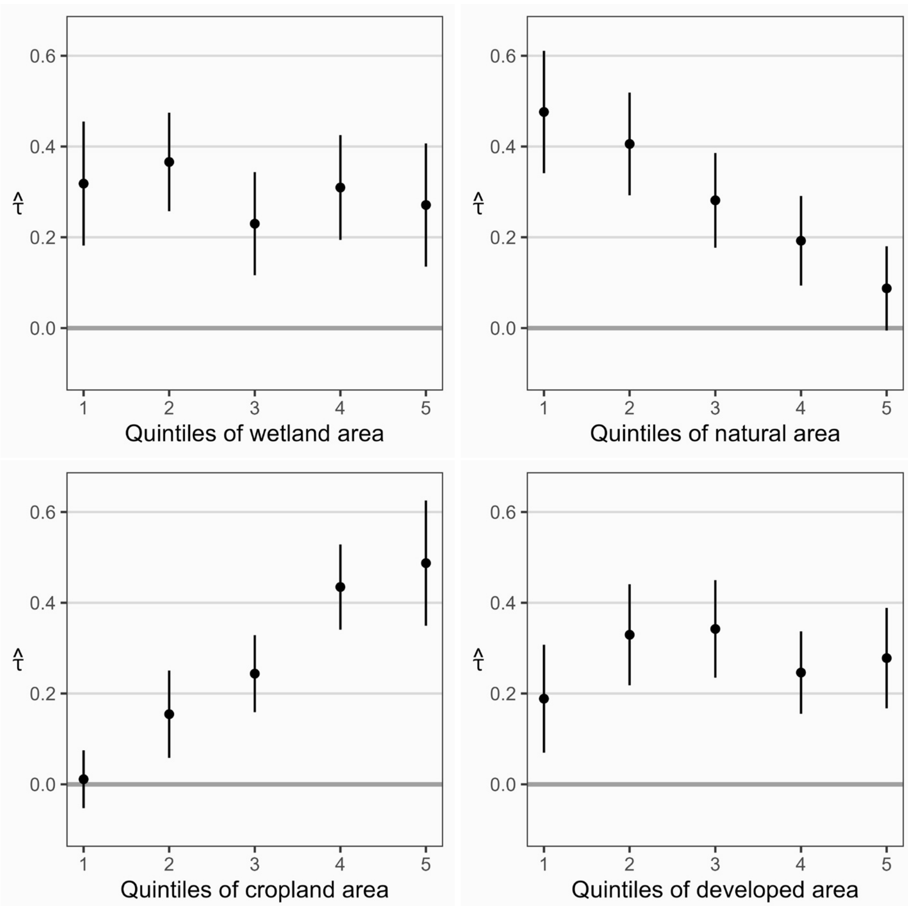
[11]Note that in many cases $F_i \neq \overline{A}_{rel,i}^G$. We only observe the coordinates of F_i and not of $\overline{A}_{rel,i}^G$ and therefore we use F_i in obtaining the estimates.

The wetland area losses, A_t^L, and estimates of the wetland area gains that were used to compensate for those impacts, $\widehat{A_t^G}$, are depicted in Figure 9.

C Heterogeneity

Figure C.1 shows how the estimated wetland gains vary according to the landscape characteristics of the treated units.

Figure C.1: Wetland area gains across landscape characteristics



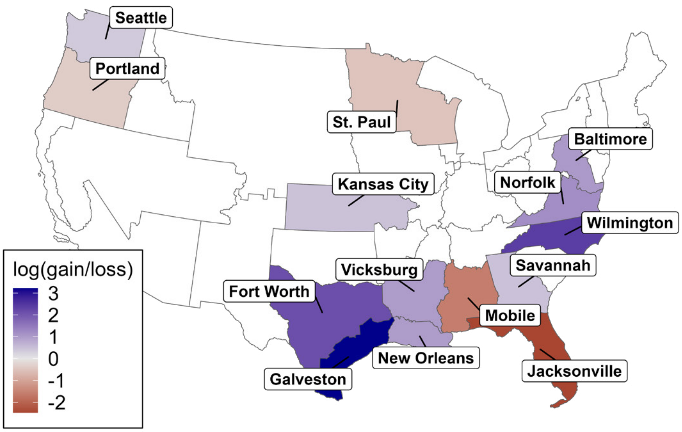
Long differences estimates and pointwise 95% confidence intervals of the effect of site establishment on wetland area by subsamples according to quintiles of landscape characteristic at baseline (τ in eq. (2) interacted by quintile indicators). Landscape characteristics are measured at 1 km buffers surrounding the treated sites. Quintiles of land cover variables are constructed according to the share of the land cover class out of the total site area. See the notes in Table 2 for details on estimation. Land cover data source: USGS (2022).

Two findings emerge from the estimates in Figure C.1. First, the wetland gains have occurred equally between landscapes with differing amounts of pre-existing wetlands. An initial concern was that the wetland gains have occurred in landscapes with relatively large existing wetland areas. In such landscapes, the marginal functional value of additional wetland area is smaller than in landscapes with few existing wetlands. The results in Figure C.1 indicate that this is not the case. Second, most increases in wetland extent have taken place in an agricultural landscape. The availability of suitable land area for conversion is the single most important determinant for the type and success of compensation projects.

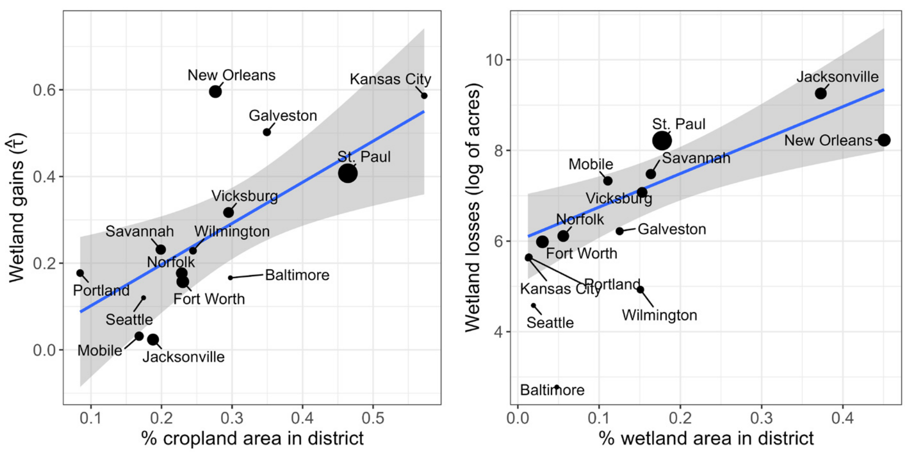
D Supplementary figures

Figure D.1: Heterogeneity of gains and losses by region

(a) Heterogeneity of net losses



(b) Wetland area gains and losses and landscape characteristics



Panel (a): Wetland gain/loss ratios by region. Gains: Acres of wetland area gains at mitigation bank sites. Obtained from long differences estimates by the administrative districts of the US Army Corps of Engineers. Losses: Acres of wetland area losses by district from ORM data (Corps, 2020). Only including regions with more than five treated sites and a positive point estimate.

Panel (b): Wetland gains and losses as in (a). Cropland and wetland shares within district from USGS (2022) measured in 2000. Fitted line and 95% CI from a weighted linear regression where the weights (point sizes) correspond to the number of ever-treated sites in the full estimation sample.

Chapter 2

The Choice of Mechanism for Biodiversity Offsetting

Abstract

Policy makers worldwide are giving increasing attention to the use of markets for offsetting biodiversity losses from development projects. It is unclear whether market-based instruments, such as banking mechanisms that entail third-party offsets for developers to purchase, hold significant promise for implementing no-net-loss regulation at least cost, relative to the conventional command-and-control approach of developer-led offsets. In this paper, we provide a theoretical examination of the costs and benefits of the two approaches. We find that (1) if offsets by banks are of insufficient quality relative to developer-led offsets, a large enough market could compensate for the lack of equivalency due to cost-savings from market expansion, and (2) if entry costs are positively correlated with restoration quality, the market could hold banks of low quality, which is an outcome that favors the relative performance of developer-led offsets. We illustrate our results in the case of the US wetland mitigation program. We find evidence that differentials on offset quality and the opportunity cost of land are significant drivers of the choice of offsetting mechanism, though cost savings have a much larger impact on the choice than environmental benefits.

This chapter is coauthored with João Vaz and Jessica Coria.

1 Introduction

Development activities have altered about 30% of the land surface of the planet, causing considerable impacts to biodiversity (Winkler et al., 2021). Biodiversity offsets are a potentially powerful tool for relieving the tension between development and conservation objectives. Offsets aim to compensate biodiversity loss by enhancing, creating, or protecting environmental values on separate land. At present, more than 100 countries have biodiversity compensation policies in place or enabled (GIBOP, 2019). Offset projects have been estimated to occur across upward of 150,000 km^2 (Bull and Strange, 2018) and the annual value of offset transactions has grown to $US 6.3-9.2 billion (Deutz et al., 2020).

Offsetting policies differ both within and across countries, depending on the environmental values that different programs seek to protect.[12] There are, nevertheless, commonalities across regulatory frameworks. First, offsetting is usually considered to be the final step in a mitigation hierarchy to compensate residual damages after appropriate steps have been taken to avoid and minimize impacts on project sites. Second, offsets serve to meet the policy goal of "no net loss" (NNL) of biodiversity alongside development. And finally, offset policies generally fall into two broad categories: "command systems," in which land developers are responsible for designing and constructing offset projects, and "market-like" approaches, such as biodiversity, or mitigation banking, whereby a third party undertakes restoration work and sells damage-equivalent restoration credits to developers.

The oldest NNL regulatory framework is the compensatory mitigation policy of wetlands in the US, which laid the foundation for offsetting programs observed elsewhere (OECD, 2016). In the early period of the program, regulators specified a clear preference for on-site, and in-kind, replacement of wetlands performed by developers (Corps and EPA, 1990). This preference was based on the premise that full and equivalent replacement of losses is best achieved by compensating with the same type of ecosystem functions that were lost in the course of development activities. Recent regulations, however, have expressed an explicit preference for compensation off-site by a third-party (Corps and EPA, 2008), which led to the development of regional mitigation banking markets, whereby developers can purchase offsets from private firms that conserve land within the same watershed.[13]

[12] A wide variety of schemes exist, including "Conservation Species Banking" and "Wetland and Stream Mitigation" in the US, "Habitat Compensation" in Canada, "Green Offsets" and "BioBanking" in Australia, the "Habitat and Birds Directive" and "Natura 2000" in the EU, and "Biodiversity Offsets" in South Africa.

[13] In 2008, developer-led offsets represented 59% of the compensatory measures in the US (Madsen, 2011), while between 2015-2020, mitigation banking represented 51% (Corps, 2020).

The main criticisms of developer-led offsets are that developers often fail to implement the required mitigation and that, when they do, the restoration projects are insufficient to meet conservation goals, due to the patchiness of small-scale restoration projects, which fail to account for how biodiversity responds to ecological processes at the landscape level (Council et al., 2001; GAO, 2005; Marsh et al., 1997). The advantages touted for mitigation banking, on the other hand, include the ability to carry out large-scale restoration projects that have a higher chance of success than work done in a piecemeal, project-by-project fashion, and that restoration can be performed in more favorable locations for the creation of environmental values. Although other regulatory agencies have echoed the advantages of banking (e.g., in the UK, Australia, and Sweden) and several countries are contemplating the adoption of similar strategies to improve biodiversity protection, whether a market approach is more effective at promoting NNL, as opposed to the developer-led mitigation option, remains an open question that we address in this study.

Despite the emphasis on market-led mechanisms to mainstream conservation efforts, there are insufficient economic analyses of biodiversity offsetting schemes. Most studies focus on offset production decisions by banks (Coggan et al., 2013; Drechsler and Hartig, 2011; Fernandez and Karp, 1998), the optimal release of offset credits (BenDor et al., 2014; Drechsler, 2022), spatial considerations in the determination of ecological units of exchange (Bonds and Pompe, 2003; Drechsler and Wätzold, 2009), or the determination of the value of ecosystem services (Boyd and Banzhaf, 2007; Holmes et al., 2004; Yang et al., 2008), though not on the dynamics of offset markets themselves. Notable exceptions include Kangas and Ollikainen (2019) and Simpson et al. (2021b), who analyze how changes in the ecological trading currency affect the performance of offsets markets, and Doyle and Yates (2010) who examine how market entry and the size of restoration projects affect the implementation of NNL. Previous studies nevertheless assume that offset markets operate under regular competitive conditions. However, unlike standard tradable permit markets, regulators not only dictate property rights and market rules but also grant legitimacy to restoration sites, match impact to restoration locations, determine the supply and demand of biodiversity units, and employ an active oversight of market activities (Koh et al., 2019; Needham et al., 2019a). It is, thus, important to understand how the unique features of biodiversity markets inform the characteristics of participating banks and how the outcomes of mitigation banking compare to those from developer-led mitigation.

Herein, we contrast offset markets against developer-led offsets in the context of NNL regulation. We model the decision by an authority that wants to achieve NNL at the least possible cost using one of two policies: permittee-responsible mitigation (PRM), which corresponds to the "command" approach of developer-led offsets, and mitigation banking (MB), which corresponds to the "market" approach of third-party offsets. Our model is based on three observations about the features of existing markets. First, MB programs are usually constrained to large geographic or ecological boundaries, such as watersheds, with vast availability of private land that could be used for restoration purposes. [14] We therefore model market entry using a free-entry equilibrium, in which landowners entering the market as banks must pay a fixed entry cost, which corresponds to the upfront legal and technical costs related to setting up a mitigation bank plus the location-specific opportunity cost of land (which is never recovered, since banks are required to put their land into a conservation easement that permanently limits land-use for protection of environmental values).

Second, we allow heterogeneous quality of offsets by banks, since these are performed at different locations with different baseline quality conditions. The distribution of restoration effort across banks is decided by the regulator in a cost-minimizing way. Unlike in regular competitive settings, in which firms may influence market outcomes by manipulating their production decisions, the amount of restoration produced by banks is established at the moment a bank is created, i.e., bank owners and the regulator sign a bank-enabling instrument, which is a formal agreement establishing liability, performance standards, monitoring requirements, and the terms of offset credit approval, which includes the restoration actions required.

Third, the regulator establishes the currency, or the trading ratio, which reflects the amount of restoration units (credits) that must be provided to compensate one unit of loss at a project site (debits). These ratios are customized based on several factors, including the type of compensation (restoration, preservation, creation), equivalence of the offset (in-kind or out-of-kind), and location (on-site or off-site). Once the trading ratio is established and the developer is required to purchase credits from banks, the developer meets with individual banks to agree on the terms of the transaction. We therefore assume that the price of credits originates from a bargaining agreement between the developer and each individual bank.

[14] As a general principle, in the US, wetland mitigation occurs at the watershed level.

Our theoretical model allows us to identify the mechanisms determining whether the market approach performs better or worse than developer-led offsets. First, even if offsets by banks are of insufficient quality relative to developer-led offsets, a large enough market could compensate for the lack of equivalency due to the cost-reduction effect of having several banks contributing to compensating a fixed amount of damages. Second, if entry costs are positively correlated with restoration quality, the market could hold banks of low quality, which favors the relative performance of developer-led offsets. Our empirical analysis of the choice of offsetting mechanism under the US wetland mitigation program illustrates and supports our analytical findings since—despite differentials in the quality of restoration—the benefit/cost ratio of conservation on-site and off-site is similar on average. Furthermore, our empirical analysis suggests that the choice of MB over PRM is dictated by cost savings rather than environmental benefits.

The paper is organized as follows. Section 2 describes the main features of the model. Section 3 compares the performance of PRM and MB given mitigation banks already in place. Section 4 examines entry decisions. In Section 5 we present our empirical analysis. Section 6 summarizes our findings and concludes the paper.

2 Model

There are three types of economic agents in the model: developers, banks, and a regulator. Developers degrade land in the course of their economic activities and banks convert land into restored ecosystems. Developers can also restore land, although, while banks perform land restoration off-site, developers restore land on-site. The regulator implements NNL by allowing one of two policies: PRM, whereby the developer is responsible for restoring land, or MB, whereby banks restore land that is converted into credits to be purchased by the developers to meet their mitigation requirements.

We consider a representative developer that degrades land of a fixed size $Q = 1$. Land conversion generates an economic benefit of $A \times Q = A$. Should the developer be required to restore land on-site, it faces a set-up cost of $F_0 > 0$, which can be interpreted as the cost of land adjacent to the development site where restoration will take place. The quantity of land restored, or abatement, by the developer is denoted by a_0.

There are N landowners, or potential banks, that may retire land for restoration purposes. The set-up, or entry, cost into the MB program is $F_i > 0$ for $i = 1, \ldots, N$, where F_i can be interpreted as the value of alternative uses of land.

The actual number of banks is $1 \leq n \leq N$ and is determined within the model. The abatement of each of the n banks in the market is denoted by a_i.

For simplicity, we let the variable cost of restoring land be identical for both developers and banks, as land restoration often entails the subcontracting of firms specialized in restoring ecosystems. We assume the standard strictly-convex cost relationship $c(a)$ satisfying $c(0) = 0$, $c' > 0$, and $c'' > 0$. Heterogenous effects will nevertheless emerge due to baseline land-quality conditions, which will reflect the "real cost of restoration at different mitigation sites.

The social cost of $Q = 1$ units of developed land is constant and equal to $D \times Q = D$, and the benefit of restoration a_k at site $k = 0$ by the developer or at site $k \in \{1 \ldots, n\}$ by each individual bank in the MB program is given by $B_k(a_k)$. Both D and B_k are measurements of the social value of ecosystem functions lost and restored. The value of land restored depends on the type of restoration, baseline conditions, and location of the land restored, which are specific to each site. For simplicity, we assume that the welfare contribution of restoration at each site is additively separable so that the total benefit from mitigation is $\sum_k B_k(a_k)$. Moreover, although we assume that the marginal value of restoration is strictly increasing in abatement, $B_k' > 0$, we refrain from making assumptions related to the convexity properties of the benefit function, as these have implications regarding returns to scale from restoration, which we address later on. Therefore, if $B_k'' > 0$, restoration benefits exhibit increasing returns to scale so that a larger restored ecosystem gives more function or value than several smaller restoration sites, and if $B'' \leq 0$, benefits exhibit nonincreasing returns, so that smaller restoration projects produce proportionally more benefits than those accrued from larger projects.

The regulator implements NNL of value by ensuring that the constraint $D \leq \sum_k B_k(a_k)$ is met through one of two policies: PRM or MB. We assume that it is technologically feasible that restoration at each site is capable of compensating damages D.[15] The goal of our evaluation is to understand the circumstances under which one policy fares better than the other. That is, whether NNL can be achieved more efficiently through on-site mitigation by the developer itself (PRM), or by having one or several banks performing off-site mitigation (MB). The objective of the regulator is to maximize economic surplus subject to the NNL constraint. Since the size of converted land, and therefore ecological damages, are fixed, maximizing economic surplus is equivalent to minimizing the cost of meeting the NNL constraint.

[15] If otherwise, the model would capture the fact that mitigation comes at a prohibitively high cost.

We proceed to explain the components and structure of each policy in the next two subsections.

2.1 Permittee responsible mitigation

Suppose that the regulator implements the PRM policy. The regulator sets a level of abatement a_0' to be employed by the developer by solving:

$$a_0' = \underset{a_0}{\text{argmin}} \left\{ c(a_0) \text{ subject to } B_0(a_0) \geq D \text{ and } a_0 \geq 0 \right\}$$

where $a_0' > 0$ follows directly from the constraint, which is met at equality, since costs are strictly increasing in abatement, so that $B_0(a_0') = D$. The total surplus under the PRM policy is $W' = A - c(a_0') - F_0$.

2.2 Mitigation banking

Suppose that the regulator implements the MB policy and that n landowners decided to participate in the program. Each landowner incurred in an entry cost F_i for $i \in \{1, \ldots, n\}$ and the distribution of abatement is decided by the regulator in a cost-minimizing way:

$$C(D) = \min_{a_1,\ldots,a_n} \sum_{i=1}^{n} c(a_i) \text{ subject to } \sum_{i=1}^{n} B_i(a_i) \geq D \text{ and } a_i \geq 0 \tag{1}$$

where $C(D)$ is the minimum aggregate (variable) cost of cleaning up damages D.

The Lagrangian associated with the problem above is

$$\mathcal{L} = -\sum_{i=1}^{n} c(a_i) + \lambda \left[\sum_{i=1}^{n} B_i(a_i) - D \right] + \sum_{i=1}^{n} \mu_i a_i$$

and the distribution of abatement is found from the first-order conditions:

$$\begin{aligned}
\frac{\partial \mathcal{L}}{\partial a_i} &= -c'(a_i) + \lambda B_i'(a_i) + \mu_i = 0 \quad i = 1, \ldots, n \\
\frac{\partial \mathcal{L}}{\partial \lambda} &= \sum_{i=1}^{n} B_i(a_i) - D \geq 0 \quad \wedge \quad \lambda \geq 0 \quad \wedge \quad \frac{\partial \mathcal{L}}{\partial \lambda} \lambda = 0 \\
\frac{\partial \mathcal{L}}{\partial \mu_i} &= a_i \geq 0 \quad \wedge \quad \mu_i \geq 0 \quad \wedge \quad \frac{\partial \mathcal{L}}{\partial \mu_i} \mu_i = 0 \quad i = 1, \ldots, n
\end{aligned}$$

Let $\{a_i^*\}_{i=1}^n$ denote the solution to the problem above. Assume that $B_i'(0) > c'(0)$ for all $i \in \{1, \dots, n\}$ so that every bank in the program makes a positive contribution to meeting the NNL constraint: $a_i^* > 0$.[16]

Note that for any $i \neq j$, we have:

$$\lambda^* = \frac{c'(a_i^*)}{B_i'(a_i^*)} = \frac{c'(a_j^*)}{B_j'(a_j^*)}. \tag{2}$$

That is, for an efficient allocation of abatement across banks, the equimarginal principle must hold. This principle establishes that the marginal cost of abatement, divided by the marginal benefit of abatement at every site i, must be equal across all producers of restoration.

At the minimum, $C'(D) = \lambda^* > 0$ by the envelope theorem. It is the case that the aggregate cost of restoration satisfies $C(0) = c(0)$, $C'(0) = c'(0)$, and, since $c''(a_i) > 0$, for all i, it follows that $C''(D) > 0$.

We are now in a position to establish the first general result of our analysis regarding how the cost of meeting the NNL constraint changes with size of MB program. Our evaluation contrasts the efficiency of having the developer performing mitigation on site versus the possibility of having more than one bank compensate damages off site. It is therefore important that we understand how the aggregate clean-up cost varies with the number of banks in the MB program.

There are two key effects at play as we increase the size of the market. The first is the cost-reduction effect of having a larger pool of banks contributing to meeting the same target. This effect drives down the total cost of the MB program since costs are strictly convex and the individual contribution of any one firm decreases as more banks enter the program. The second effect is the scale effect. The fact that benefits from restoration may exhibit increasing returns to scale, means that larger restoration projects lead to higher marginal benefits. Few large banks could be needed to meet the NNL constraint, but this could increase the total cost of cleaning up damages.

The key condition dictating the magnitude of the cost-reduction effect against that of the scale effect is:

$$\frac{c''}{c'} \geq \frac{B''}{B'} \tag{3}$$

[16]If $n = 1$, it must be that $a_1^* > 0$, irrespective of $B_i'(0) > c'(0)$, since the NNL constraint would not be met otherwise. For $n > 1$, suppose that there exists k such that $a_k^* = 0$ and $B_k'(0) \leq c'(0)$. Again, it must be that for some $j \neq k$, we have $a_j^* > 0$ so that the NNL constraint is met. Thus, $\lambda = c'(a_j)/B_j'(a_j) > 0$. Because (1) for $a_k^* = 0$, we must have $c'(0) \geq \lambda B_k'(0)$ and (2) $\lambda > 0$, it follows that $c'(0) > B_k'(0)$, which is a contradiction. Thus, $B_i'(0) > c'(0)$ guarantees that $a_i^* > 0$ for all $i \in \{1, \dots, n\}$.

which relates the log-derivative of marginal costs against the log derivative of marginal benefits. That is, the rate at which marginal costs/benefits change relative to their current size. If the above relationship holds, then the magnitude of the cost-reduction effect is larger than that of the scale effect, and the total cost of cleaning damages decreases with the number of banks. If otherwise, the magnitude of the scale effect is larger, and it is possible that the total cost of the MB program increases as more banks participate in the MB program.

Proposition 1: *If $c''/c' \geq B_i''/B_i'$ for all $i \in \{1, \ldots, N\}$, then $C(D)$ is non-increasing with n.*

Proof: Denote by $C^*(D)$ the cost with n firms and by $C^o(D)$ the cost with $n+1$ firms.

Note that

$$\sum_{i=1}^{n} B_i(a_i^*) = D \quad \text{and} \quad \sum_{i=1}^{n} B_i(a_i^o) + B_{n+1}(a_{n+1}^o) = D.$$

Since $B_{n+1}(a_{n+1}^o) > 0$, it must be that

$$\sum_{i=1}^{n} B_i(a_i^*) > \sum_{i=1}^{n} B_i(a_i^o).$$

Thus, there exists a $k \in \{1, \ldots, n\}$ such that $a_k^* > a_k^o$.

Denote by $\Phi_k(a_k) = c'(a_k)/B_k'(a_k)$, where

$$\text{sign}\left[\Phi_k'\right] = \text{sign}\left[c''B' - c'B''\right].$$

Observe that

$$\frac{\partial C^*}{\partial D} = \lambda^* = \Phi_k(a_k^*) \quad \text{and} \quad \frac{\partial C^o}{\partial D} = \lambda^o = \Phi_k(a_k^o).$$

Since $C'(0) = c'(0)$ for both C^* and C^o, and $a_k^* > a_k^o$, then if $\Phi_k' \geq 0$, we have $\lambda^* \geq \lambda^o$ $\square$

Note that if benefits exhibit decreasing returns to scale, $B'' < 0$, so that smaller restoration projects lead to higher marginal benefits than larger projects, then both the cost-reduction and scale effects work in the same direction: that of a decrease in the overall burden of meeting NNL. If benefits exhibit increasing returns to scale, $B'' > 0$, then the cost and scale effect work in opposite directions. However, increasing returns to scale alone is not sufficient for total costs to be decreasing with market size. If the cost-reduction effect is more pronounced than the scale effect, then aggregate costs would still decrease as the size of the program grows large.

It should be noted that (3) provides us with a sufficient second-order condition for the stationary allocation $\{a_i^*\}_{i=1}^n$ to be the minimizer to the problem. If this condition does not hold, we cannot establish that the allocation found from the equimarginal principal is optimal. Finding the equilibrium distribution of abatement would instead require direct methods. Unless specifically stated otherwise, we assume throughout that (3) holds.

We now turn to the interaction between banks and the developer in the MB program. The regulator allows the developer to buy restoration from banks to meet its mitigation requirements. This involves the setting of a "trading ratio" that adjusts for functional differences between ecosystems lost and those restored at the site of every individual bank. Let σ_i denote the rate that converts units of impacted land into units of restored land at site i. If the developer damages $q > 0$ units of land, then meeting the NNL from source i requires that $B_i(a_i) = Dq$, which implicitly defines $a_i(q)$. It follows that $\sigma_i(a_i) = da_i/dq$:

$$\sigma_i(a_i) = \frac{D}{B_i'(a_i)} \tag{4}$$

so that $a_i = \sigma_i(a_i) \times q$. Thus, if the developer damages q units of land, it can meet its mitigation requirements by purchasing $a_i/\sigma_i(a_i)$ units of damage-equivalent abatement, or credits, from source i.

If n banks participate in the MB program, the regulator stipulates that they perform abatement $\{a_i^*\}_{i=1}^n$ which, on aggregate, corresponds to the total abatement necessary to clean up damages D. Because damages are fixed, the developer must buy all the credits that have been generated by each of the n banks to meet its mitigation requirements. Given $\sigma_i(a_i^*)$, the developer buys a_i^*/σ_i^* credits from every individual bank i.

There will be a price p_i per credit bought from each of the banks. The price is negotiated between the developer and each bank on a case-by-case basis. Unlike standard pricing models, in the MB program, banks cannot influence the price by manipulating the supply of credits. Rather, the supply of credits is set by both the regulator and the landowner when a bank is created. Given the level of damages generated by development activities, the regulator requires that the developer buy a specified number of credits from banks in the program. When a bank is matched to a developer, it is often the case that a broker helps in the decision of a price for the credits. We therefore assume that prices originate from a cooperative, or Nash, bargaining agreement on how to share the surplus that both parties have generated: the economic benefit of land degradation by the developer net of the cost of land restoration by banks. We provide a detailed account of how prices are generated in Section 4, where we discuss the dynamics of market entry.

For now, it is sufficient to note that, given a set of agreed-upon prices $\{p_i\}_{i=1}^n$, the profits of the developer and of each of the banks in the program are given by:

$$\pi_0 = A - \sum_{i=1}^{n} p_i a_i^* / \sigma_i^* \quad \wedge \quad \pi_i = p_i a_i^* / \sigma_i^* - c(a_i^*) - F_i \quad \text{for } i = 1, \ldots, n \tag{5}$$

and the total surplus under the MB policy is:

$$W^* = A - \sum_{i=1}^{n} \left[c(a_i^*) + F_i \right]$$

where $\sum_i c(a_i^*)$ corresponds to the minimum variable cost $C(D)$ introduced earlier. Note that since there is no change in the efficient level of total abatement produced, there is no deadweight loss generated by "market power." Rather, the total surplus is kept constant irrespective of the prices that originate in the model. The distribution of surplus across banks and the developer, however, will be different, as it will depend on each bank's contribution to the total restoration effort, as well as on the bargaining power between the developer and each bank.

3 Comparing PRM vs MB

We are now in position to compare the performance of PRM and MB. We start by ignoring entry costs and focus on an ex-post evaluation of the two policies, assuming that n banks have decided to participate in the MB program. Let us assume that the benefit function of each firm i is given by:

$$B_i(a_i) = \delta_i f(a_i)$$

where δ_i reflects site-specific restoration quality and $f(a)$ is a strictly increasing ($f' > 0$) though potentially nonlinear function that formalizes returns to scale. To help us gather intuition, we consider first the case of linear benefit function (i.e., constant returns to scale) and discuss afterward how the results change in the case of nonlinear benefits (see Appendix A).

Assume that $f'' = 0$, such that $B_i(a_i) = \delta_i a_i$, with $\delta_i > c'(0)$ for $i = 1, \ldots, n$, so that every bank generates a positive abatement $a_i^* > 0$. Further, suppose that each bank has a distinct restoration value δ_i. The allocation of abatement for $n > 1$ banks participating in the MB program follows from the equimarginal principle in (2), such that

$$\frac{c'(a_i^*)}{\delta_i} = \frac{c'(a_j^*)}{\delta_j} \quad \text{for every } i \neq j.$$

Because $c'(a) > 0$, it must be that for any $\delta_i \geq \delta_j$, we have $a_i^* \geq a_j^*$. That is, banks with higher restoration values should produce a larger level of abatement. It is nevertheless the case that, were only one bank to participate in the MB program (i.e., $n = 1$) its abatement would follow from the constraint, $a_i^* = D/\delta_i^*$, so that the larger the marginal restoration value, the lower the level of abatement employed.

Result 1: *Suppose that $f'' = 0$ and each bank $i = 1, \ldots, n$ has a distinct restoration value δ_i.*

(a) If $\max\{\delta_i\} \geq \delta_0$ then MB $\succsim$ PRM for all $n \geq 1$.

(b) If $\max\{\delta_i\} < \delta_0$, then, for every $n \geq 1$, there exists $\tilde{\delta}(n) \leq \delta_0$ such that:

1. *for $\max\{\delta_i\} \geq \tilde{\delta}(n)$, we have MB $\succsim$ PRM.*
2. *for $\max\{\delta_i\} < \tilde{\delta}(n)$, we have MB $\prec$ PRM.*

Moreover, $\tilde{\delta}(n)$ is decreasing with n.

Proof: Let $\max\{\delta_i\} = \delta_1$ so that restoration values are sorted in descending order: $\delta_1 \geq \cdots \geq \delta_n$.

Let $a_i^*|_n$ define the equilibrium abatement of bank $i = 1, \ldots, n$ given an MB program of size n.

- If $n = 1$ and $\delta_1 \geq \delta_0$, then $a_1^*|_1 \leq a_0'$, so that $c(a_0') \geq c(a_1^*|_1)$. Since $f'' = 0$, it follows from Proposition 1 that $c(a_1^*|_1) \geq \sum_i c(a_i^*|_n)$, with strict inequality for $n > 1$. Thus $c(a_0') \geq \sum_i c(a_i^*|_n)$ or MB $\succsim$ PRM whenever $\delta_1 \geq \delta_0$.

- From $C(D)$ in (1) and using the envelope theorem, we have $\partial C/\partial \delta_i = -\lambda^* a_i^* < 0$. That is, the total cost of MB program is decreasing with restoration value δ_i. Therefore, cost with heterogeneous $\delta_1 \geq \cdots \geq \delta_n$ is larger than cost with homogeneous values equal to δ_1. Thus, $\sum_i c(a_i^*|_n) \geq nc(D/n\delta_1)$.

Result 1(a) establishes that if there is at least one bank with a marginal restoration benefit that is larger than that of the developer, then the MB program will necessarily fare better than PRM, irrespective of the size of the program. Result 1(b) establishes that if all banks in the program exhibit restoration values that are smaller than that of the developer, then, for MB to fare better than PRM, the restoration value of the more productive bank needs to lie above a threshold that depends on the size of the market. Because costs with the MB program decrease as the program grows large, this minimum restoration value decreases as the size of the market increases. We illustrate the importance of the restoration value of the more productive bank with the following example.

Example: Suppose that $c(a) = a^2$ and $\delta_{i+1} = \varepsilon\delta_i$ for $i = 1, \ldots, n-1$ and $0 < \varepsilon \leq 1$ with $\delta_1 > 0$. The PRM policy entails $a'_0 = D/\delta_0$, whereas the MB policy with n banks entails:

$$a_i^* = D \times \frac{\delta_i}{\sum_{i=1}^{n} \delta_i^2} \text{ for } i = 1, \ldots, n. \tag{6}$$

Observe that

$$\sum_{j=1}^{n} \delta_j^2 = \delta_1^2 \sum_{k=0}^{n-1} \varepsilon^{2k} = \begin{cases} \delta_1^2 \times \dfrac{1-\varepsilon^{2n}}{1-\varepsilon^2} & \text{if } \varepsilon < 1 \\ \delta_1^2 \times n & \text{if } \varepsilon = 1. \end{cases}$$

If we define

$$\phi(\varepsilon, n) = \begin{cases} \dfrac{1-\varepsilon^{2n}}{1-\varepsilon^2} & \text{if } \varepsilon < 1 \\ n & \text{if } \varepsilon = 1 \end{cases}$$

we may rewrite a_i^* as

$$a_i^* = \frac{D}{\delta_1} \times \frac{\varepsilon^{i-1}}{\phi(\varepsilon, n)}$$

such that

$$\sum_{i=1}^{n} c(a_i^*) = \left(\frac{D}{\delta_1}\right)^2 \times \frac{1}{\phi(\varepsilon, n)}.$$

Now, if we compare PRM to MB:

$$c(a_0) - \sum_{i=1}^{n} c(a_i^*) = D^2 \times \left[\frac{1}{\delta_0^2} - \frac{1}{\delta_1^2 \phi(\varepsilon, n)}\right].$$

Thus, MB $\succsim$ PRM as long as:

$$\delta_1 \geq \frac{\delta_0}{\sqrt{\phi(\varepsilon, n)}}.$$

By defining the threshold value $\overline{\delta}_1(n,\varepsilon)$ from meeting the above at equality, we find:

$$[A] \quad \frac{\partial \overline{\delta}_1}{\partial n} = \delta_0 \times (-1/2) \times \phi(\varepsilon, n)^{-3/2} \times \frac{\partial \phi}{\partial n} < 0$$
$$[B] \quad \frac{\partial \overline{\delta}_1}{\partial \varepsilon} = \delta_0 \times (-1/2) \times \phi(\varepsilon, n)^{-3/2} \times \frac{\partial \phi}{\partial \varepsilon} < 0.$$

$[A]$ is a reiteration of Result 1(b): for MB $\succsim$ PRM, the minimum restoration value of the more productive bank decreases as the size of the MB market increases. $[B]$ tells us that the minimum restoration value decreases with increases in the ε parameter, which formalizes the degree of heterogeneity among banks. An increase in ε may be interpreted in two ways. As ε approaches one, (1) the restoration value of the more productive banks becomes relatively homogeneous, or (2) the decay rate of restoration quality is small at the extensive margin. Both interpretations chime with the observed MB policy where banks must be within the same watershed of impact locations. First, marginal benefits of land restored within the same watershed would be more homogenous, which might increase the likelihood that off-site mitigation is an adequate replacement to damages at impact locations, and therefore decrease the minimum quality required for the MB program to fare better than PRM. Or second, if banks that are located at farther locations are of lower ecological equivalency, the smaller the decay rate, the lower the minimum quality of the more productive bank (or that which is closest) to compensate for the lower productivity of more remote banks.

Concerning the non-linearity of the benefits function, it is unclear what type of returns to expect since the extent to which the scale of on- and off-site restoration projects is important for compensating damages depends on the functions and values lost due to development activities. For instance, whereas a large-scale restoration project may be appropriate when degradation is extreme, (e.g. when hydrologic processes have been modified through the alteration of flow regimes of waterbodies), small-scale projects may provide more benefits if the type of restoration needed entails plant diversity, the removal of invasive weeds through hand-pulling methods, or the control of nutrients at isolated locations to improve water quality (Woodward and Wui, 2001). In Appendix A, we show that Result 1 also holds if the benefit function exhibits decreasing/increasing return to scale, and that minimum level of individual quality of banks required for MB to fare better than PRM decreases (increases) with decreasing (increasing) returns to scale.

4 Market entry

In the previous sections, we performed an ex-post welfare analysis of the two policies. Here, we examine the entry decision by individual banks. To do so, we reintroduce costs of entry. Landowners will enter the market as restoration banks if there are positive profits to be made in the MB program. The profit of each individual bank is that given by (5). Thus, banks will enter the program if

$$\frac{p_i}{\sigma_i^*} \geq \frac{F_i + c(a_i^*)}{a_i^*} \tag{7}$$

That is, if the price per unit of abatement is greater than average total costs (ATC).

The price p_i is determined on a case-by-case basis through a bilateral agreement between the developer and any one bank. In equilibrium, the developer is required to buy all the credits have been generated by each of the n banks that decided to participate in the program. Each bank i produces a_i^*/σ_i credits. Denote the total amount of credits by Ω, where $\Omega = \sum_i a_i^*/\sigma_i$. The benefit that the developer derives from a bilateral agreement with bank i is $A \times (a_i^*/\sigma_i)/\Omega$, where A is the total benefit from imposing damages D and $(a_i^*/\sigma_i)/\Omega$ is the share of credits bought from source i, or the fraction of benefit A that emerges from the transaction with bank i. If a bilateral agreement is reached, the payoff from a successful negotiation to the developer is:

$$\tilde{\pi}_0 = \left[A/\Omega - p_i\right] \times (a_i^*/\sigma_i^*)$$

and that of the bank is:

$$\tilde{\pi}_i = \left[p_i - c(a_i^*)/(a_i^*/\sigma_i^*)\right] \times (a_i^*/\sigma_i^*).$$

As mentioned in Section 2.2, the environment determining the price emerging from the interaction between the developer and banks is that of Nash bargaining (Nash, 1953). The bargaining problem consists of a set of feasible agreements and a pair of disagreement payoffs which specify what each side would obtain in the absence of an agreement. For simplicity, we set disagreement payoffs to zero. The Nash-bargaining solution arises from the product of the differences between the payoff each party assigns to an agreement. Formally, the Nash-bargaining solution is the value of p_i which solves:[17]

$$\max_{p_i} \left[A/\Omega - p_i\right]^{1-\eta} \left[p_i - c(a_i^*)/(a_i^*/\sigma_i^*)\right]^{\eta}$$

where $0 \leq \eta \leq 1$ is interpreted as a measure of bargaining power of the bank, where $\eta = 1$ indicates that the bank has maximum bargaining power (Moene et al., 1992). If an agreement is reached:

$$p_i^* = \eta \frac{A}{\Omega} + (1-\eta) \frac{c(a_i^*)}{a_i^*/\sigma_i^*}$$

where the price has the reasonable form of a weighted average between the maximum amount that the developer would be willing to pay and the minimum amount that the bank would be willing to accept as the outcome of the negotiation. That is, a weighted average between the benefit generated by the production of one credit, A/Ω, and the variable cost per credit, $c(a_i^*)/(a_i^*/\sigma_i^*)$, or the cost of abatement divided by the suppliable credits that the abatement produced.

Observe that, if a firm decides to enter the MB program by paying the fixed cost of entry, the negotiated price must generate enough revenues to cover variable costs, since otherwise the bank would "shutdown." The entry condition in (7) can thus be recast in terms of whether the surplus above variable costs per credit that a bank can capture through bargaining can compensate fixed costs per credit:

$$\eta \times \left[\frac{A}{\Omega} - \frac{c(a_i^*)}{a_i^*/\sigma_i^*}\right] \geq \frac{F_i}{a_i^*/\sigma_i^*}. \tag{8}$$

The objective now is to understand how the different quality of banks and the potentially heterogeneous fixed costs of entry inform the characteristics of the banks that participate in the MB program.

[17] Nash (1953) sets forth six readily acceptable assumptions regarding the outcome of a cooperative bargaining solution. The solution is (1) contained within a set of feasible solutions, (2) rational in outcome, in the sense that each player receives a payoff no less than its threat point payoff, (3) independent of irrelevant alternatives, (4) independent of linear transformations of the set of payoffs, (5) unaffected by the numbering of players, and (6) pareto optimal. He then proves that if these assumptions are satisfied, a unique solution can be obtained by maximizing (14). For more details, see Binmore et al. (1986).

4.1 Entry decisions

Suppose that each landowner $i = 1, \ldots, N$ exhibits a heterogeneous entry cost F_i. The entry condition in (8) is replaced by:

$$\eta A \geq \Psi(a_i^*, F_i) = \begin{cases} F_i + \eta c(a_i^*) & \text{for } n = 1 \\ \lambda^* \Omega D \times \Gamma(a_i^*, F_i) & \text{for } n > 1 \end{cases} \tag{9}$$

where

$$\Gamma(a_i^*, F_i) = \frac{F_i + \eta c(a_i^*)}{a_i^*} \Big/ c'(a_i^*).$$

Define w_i as the abatement level associated with the minimum ATC of bank i, such that $w_i = \underset{a}{\operatorname{argmin}}\{\text{ATC}(a, F_i)\}$, where $c'(w_i) = (F_i + w_i)/w_i$ and $dw_i/dF_i > 0$ — that is, the larger the F_i, the larger the minimizer w_i. Denote by $\underline{w} = \max\{w_i\}$. Note that, if $a_i^* > \underline{w}$, we have:

$$\frac{\partial \Gamma}{\partial a_i^*} < 0 \quad \text{and} \quad \frac{\partial \Gamma}{\partial F_i} > 0.$$

Proposition 2: *Suppose that each landowner $k \in \{1, \ldots, N\}$ exhibits an entry cost $F_k > 0$.*

(a) For $n = 1$, if $corr(a_k^*, F_k) > 0$, then $\Psi(a_i^*, F_i) > \Psi(a_j^*, F_j)$ for $a_i^* > a_j^*$.

(b) For $n > 1$, if $corr(a_k^*, F_k) < 0$ and $a_i^* > \underline{w}$ for $i \in \{1, \ldots, n\}$, then $\Psi(a_i^*, F_i) < \Psi(a_j^*, F_j)$ for $a_i^* > a_j^*$.

Note that the smaller the $\Psi(a_i, F_i)$, the more likely that entry condition (9) is satisfied, or the greater the surplus of bank i upon entry. For $n = 1$, if equilibrium abatement is positively related with entry costs, $\Psi(a_i, F_i)$ is monotonically increasing with abatement. Thus, the surplus is highest among banks that face the lowest entry costs and that are required to perform a lower level of abatement by the regulator. On the other hand, for $n > 1$, if equilibrium abatement is negatively related with entry costs, $\Psi(a_i, F_i)$ is monotonically decreasing with abatement. Thus, the surplus is highest among banks that face the lowest entry costs and are required to perform a higher level of abatement.

From previous sections, we know how equilibrium abatement is related to the quality parameter δ_i. For $n = 1$, the higher the δ_i, the lower the abatement level required to clean up damages. For $n > 1$, the higher the δ_i, the higher the abatement level allocated to bank i. This observation combined with the provisions in Proposition 2 reinforces the same idea: For any market size $n \geq 1$, when firms with the highest quality also face the smallest entry costs, they stand to gain the most from entering the MB program.

An implication from Proposition 2 is that if $n \geq 1$ banks enter the MB program, and $\text{corr}(\delta_i, F_i) < 0$, the MB program hosts the banks with the highest quality. This is to say, if the market is populated by at least one bank, and if the correlation between quality and entry costs is negative, so that banks with land of the highest restoration quality face the lowest costs of entry, we can be sure that the banks that have entered are those of the highest quality. If the regulator knows that off-site restoration is of sufficient quality to match that of on-site restoration, then the MB policy will always fare better than PRM. If otherwise, then a large enough market will ensure that MB fares better than PRM due to the cost-reduction effect of a larger market at the extensive margin. We summarize this point below.

Result 2: *Suppose that* $corr(\delta_i, F_i) < 0$.

(a) If $\max\{\delta_i\} \geq \delta_0$, *then MB* $\succsim$ *PRM for any* $n \geq 1$.

(b) If $\max\{\delta_i\} < \delta_0$, *there exists a* $\overline{n} > 1$ *such that, if* $n \geq \overline{n}$, *then MB* $\succsim$ *PRM.*

However, if quality and entry costs are positively correlated, then it is ambiguous which types of banks have entered. If on the one hand banks with the highest quality face the lowest post-entry costs (and thus the largest post-entry surplus), those same banks also face higher entry costs, which might deter them from entering. Overall, because the relationship between equilibrium abatement and surplus is no longer monotonic, we could have a situation where either only high- or only low-quality banks enter the market. If low-quality banks perform restoration of insufficient quality to match that performed by the developer, PRM will fare better than MB. We illustrate this point with an example.

Example: Suppose that $c(a) = a^2$ and $B_i(a_i) = \delta_i a_i$. Suppose that $N = 2$, and denote the two landowners by A and B, where $\delta_A > \delta_0 > \delta_B$.

As before, let a_0' denote the equilibrium abatement of the developer, but, for ease of exposition, let $a_i^*|_n$ denote the equilibrium abatement of bank $i \in \{A, B\}$ given that n banks have entered the program. In this example, n is either 1 or 2.

If $n = 1$, then $a_0' = D/\delta_0$ and $a_i^*|_1 = D/\delta_i$ for $i \in \{A, B\}$. The entry condition for bank i is $\eta[A - c(a_i^*|_1)] \geq F$, where $c(a_i^*|_1) = (D/\delta_i)^2$.

Let $S_i|_1$ denote the ex-post surplus for $n = 1$ with bank i:

$$S_i|_1 = [A - c(a_i^*|_1)] = \left[A - \left(\frac{D}{\delta_i}\right)^2\right]$$

Note that, since $\delta_A > \delta_B$, we have $S_A|_1 > S_B|_1$.

If $n = 2$, one may show that:

$$a_A^*|_2 = \delta_A \times D/\theta \quad \wedge \quad a_B^*|_2 = \delta_B \times D/\theta \quad \wedge \quad \lambda^* = 2 \times D/\theta$$

where $\theta = \delta_A^2 + \delta_B^2$.

From $\sigma_i^* = D/\delta_i f'(a_i^*)$, one may show that $\Omega = 1$.

The entry condition in (9) can thus be simplified to $\eta[A - \sum_{i=1}^{2} c(a_i^*|_2)] \times \kappa_i \geq F$, where

$$\sum_{i=1}^{2} c(a_i^*|_2) = D^2/\theta \quad \text{and} \quad \kappa_i = \delta_i^2/\theta$$

where $0 \leq \kappa_i \leq 1$ and $\kappa_A + \kappa_B = 1$.

Let $S|_2$ denote the total ex-post surplus from $n = 2$ with banks A and B and $S_i|_2$ the ex-post surplus gathered by bank i:

$$S|_2 = \left[A - \sum_{i=1}^{2} c(a_i^*|_2)\right] \quad \text{and} \quad S_i|_2 = S|_2 \times \kappa_i$$

where the individual surplus of bank i is the share κ_i of the total surplus $S|_2$. Note that $S|_2 > S_i|_1$. That is, the ex-post surplus with $n = 2$ is greater than that with $n = 1$. One may show that:

$$S_A|_1 > S_A|_2 > S_B|_2 > S_B|_1 \tag{10}$$

That is, the ex-post surplus of the high-quality bank decreases if it shares the market with the low-quality bank, $S_A|_1 > S_A|_2$, whereas the ex-post surplus of the low-quality bank increases if it shares the market with the high-quality bank, $S_B|_2 > S_B|_1$.

Recall that the entry condition for bank i given that n banks participate in the market is $\eta S_i|_n \geq F_i$. Also note that MB $\succsim$ PRM whenever bank A enters (either by itself or with bank B). If only bank B enters, then PRM is better than MB.

Assume that quality is negatively correlated with entry costs: $\delta_A > \delta_B$ and $F_A < F_B$, and that at least one bank has entered the market. It is enough to show that there cannot be an equilibrium where only bank B enters by itself. If $\eta S_B|_1 \geq F_B$ (so that bank B enters alone), because (1) $\eta S_A|_2 > \eta S_B|_2 > \eta S_B|_1$ by (10) and (2) $F_B > F_A$, then it must be that if the entry condition is satisfied for bank B to enter by itself, then it must necessarily satisfy the entry condition where both banks would wish to enter together. Therefore, the market will always include high-quality banks. Because $\delta_A > \delta_0$, it follows that MB $\succsim$ PRM.

Now, suppose that quality is positively correlated with entry costs: $\delta_A > \delta_B$ and $F_A > F_B$. Again, assume that at least one bank enters the market. Here, we cannot guarantee that there will not be an equilibrium where bank B enters by itself. In particular, if

$$\eta S_A|_1 > F_A > \eta S_A|_2 > \eta S_A|_2 > \eta S_B|_1 > F_B$$

then both banks would wish to enter by themselves, $\eta S_i|_1 > F_i$, bank B would strictly prefer to share the market with the other bank, $\eta S_B|_2 > F_B$, but bank A would never enter in tandem with bank B, $F_A > \eta S_A|_2$. Therefore, we are left with two possible equilibria: either bank enters by itself. It is nevertheless more likely that the equilibrium with the low-quality bank is the one observed, since there is a "credible threat" that it enters the market when the high-quality bank enters as well.

5 Empirical analysis

The analysis above has shown that despite of quality differentials between the restoration values of on-site and off-site mitigation, MB might fare better than PRM if there are many banks in place. This might hold even if some of the banks are of low quality. In what follows, we investigate whether such results hold in the case of the US wetland mitigation program. We examine the actual differentials in quality of restoration between impacted sites and banks, as well as the correlation between land value and the quality of restoration. Furthermore, we examine if—in line with our analytical results—differential in offset-quality and opportunity costs of land have affected the actual choice of mitigation mechanism and the market entry of mitigation banks.

5.1 Offsetting under the US Wetland Mitigation Program

US wetlands and streams are protected under Section 404 of the 1972 Clean Water Act (CWA) which provides the US Army Corps of Engineers (Corps) and the US Environmental Protection Agency (EPA) the authority to regulate impacts on aquatic resources. In a Memorandum of Agreement adopted in 1990, the Corps and the EPA established no net loss as a guideline for reviewing permits for impacts to wetlands and streams (Corps and EPA, 1990). Since then, compensatory mitigation has been required to compensate for unavoidable impacts to aquatic resources. The two primary methods of compensatory mitigation are permittee-responsible mitigation (PRM), where the permittees themselves undertake the compensation project; and mitigation banking, where entrepreneurial firms invest in compensation projects and sell offsets to permittees.[18]

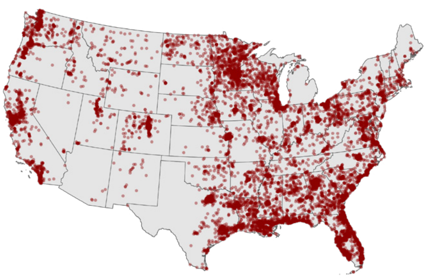
In the 1990 Memorandum of Agreement, the agencies expressed a preference for on-site and in-kind mitigation, i.e. permittee-responsible mitigation. Over the course of the following two decades, this preference gradually changed to favor mitigation banking. In a 1995 guidance document (Corps and EPA, 1995), the agencies identified two distinct benefits of mitigation banking. First, compliance monitoring is more effective when the regulator only needs to visit one mitigation bank site as opposed to many PRM sites. Second, mitigation banks would make compensatory mitigation practicable in cases where PRM was not enforced at that time (Council et al., 2001). At the same time, several reports indicated that on-site PRM was performing less than adequate in terms of ecological outcomes and that non-compliance rates were high (Council et al., 2001; GAO, 2005). Explicit regulatory preference for mitigation banking was incorporated in the 2008 Final Compensatory Mitigation Rule (2008 Rule, Corps and EPA 2008). After the adoption of the 2008 Rule, mitigation banking became increasingly used as the preferred compensation method. While PRM represented about 60% of all compensatory measures permitted by the Corps in 2008 (Institute for Water Resources, 2015), it represented only 33% over 2015–2020. By contrast, mitigation banking represented 51% of all mitigation methods in 2020 (Corps, 2020).

Figure 1 shows the distribution of impacts and mitigation banks in the contiguous United States. The maps indicate persisting differences between the preferences of local jurisdictions. For example, in the Northeastern US, state authorities have favored PRM and ILF over mitigation banking. Likewise, Wisconsin has significantly fewer mitigation banks than Minnesota, despite a high density of impacts in both states. Figure 1c displays the evolution of the offset market over 1990–2020. Offset supply increased drastically following the 2008 Rule and the establishment of new mitigation banks. Offset purchases have also steadily increased, reflecting the change in regulatory preference and the increased opportunity to resort to mitigation banking instead of PRM.

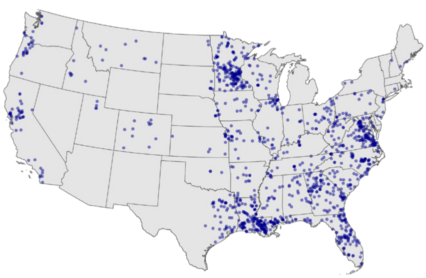
[18] More recently, in-lieu fee (ILF) programs have also emerged as a mitigation method. ILF programs function in a similar manner to mitigation banking with two important distinctions. First, only government agencies or nonprofits may establish these programs. Second, while a mitigation bank must complete the conservation project before selling any credits, ILF programs may first sell compensation credits and then use the funds for a conservation project at a later point in time. In this analysis, we only consider the comparison between PRM and mitigation banking.

Figure 1: Impact and mitigation bank locations

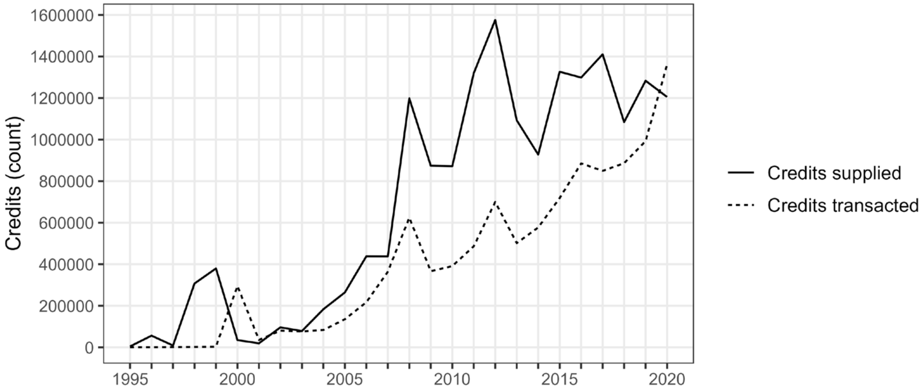
(a) Impacts to wetlands



(b) Mitigation banks



(c) Credit supply and transactions in the US mitigation banking program



Credits supplied: Credits released to mitigation banks upon compensation project completion.
Credits transacted: Credits purchased by Clean Water Act Section 404 permit holders.

Data sources: Corps (2020, 2022). Recorded impacts over 2012–2020. Michigan and New Jersey are not included in the data as they administer the CWA Section 404 program independently from the federal authorities.

5.2 Data sources

Information on impacts was obtained through Freedom of Information Act requests from the Operations and Maintenance Business Information Link Regulatory Module (ORM) administrated by the US Army Corps of Engineers (Corps, 2020). The data are observed over 2012–2020. The ORM database includes detailed characteristics of each impact, including impact coordinates, size of the impact, and type of affected water resource. Importantly, for each impact, we observe whether compensatory mitigation was required and whether PRM or mitigation banking was the chosen mitigation method. One permit entry typically has several recorded impacts. We aggregate the data to the permit level, resulting in 21,061 unique observed development projects where compensatory mitigation was required for impacts to wetlands.

We obtained a complete record of wetland mitigation banks and credit transactions from the Regulatory In-lieu fee and Bank Information Tracking System (RIBITS) (Corps, 2022). In addition to the characteristics and coordinates of individual banks, the database includes the service areas of the banks and a complete credit transaction ledger. These data enable us to identify the banks that were available to supply compensation credits for a given impact and the total quantity of credits available for purchase.

To measure the costs and benefits of wetland conservation at the impact and mitigation bank sites, we use two spatially explicit data sources. For the conservation costs, we retrieve land value estimates from Nolte (2020). These estimates comprise predicted sale prices for all properties in the contiguous US in 2010. These data provide us a proxy for the fixed set-up cost of conservation actions on land, or, the opportunity cost of conservation. For the benefits of wetland conservation, we use wetland conservation values estimated by Taylor and Druckenmiller (2022). The data are provided as the payback period of wetland conservation. The payback period is the year in which, in a given location, the expected value of the benefits from wetland conservation exceeds the opportunity cost of conservation i.e. the value of land as measured by Nolte (2020).[19] The underlying benefit estimates are based on avoided flood damages which in turn are estimated from flood insurance claim data. That is to say, the benefit measure excludes the conservation value of wetlands along other dimensions, such as water purification and quality control, or recreation values. Still, the measure provides us with a lower bound on the value of preserving wetlands.

[19] From the payback period and land value data we calculate the implied present value of conservation measured in USD per hectare per year.

In addition, to proxy for variable costs of conservation, we use the Potential Wetland Areas (PWA) data layer from the US Environmental Protection Agency (US Environmental Protection Agency, 2022). The PWA data delineate, at a 30-meter resolution, areas that naturally accumulate water due to topography and have historically had poorly or very poorly draining soils. The data are provided as an index which we aggregate into a binary variable that indicates if any wetland potential is evident.

5.3 Data description

We begin by illustrating in Figure 2 how impacts and mitigation banks are distributed and how the distribution relates to land values and conservation benefits. Figure 2a shows the distribution of impacts and mitigation banks against land value within the state of Georgia. Impacts are concentrated in urban and suburban areas with high economic activity and also high land values. Mitigation banks are located more evenly across the landscape and in less expensive areas. This observation highlights one dimension of potential cost savings from relocating conservation efforts from the impact locations to mitigation banks.

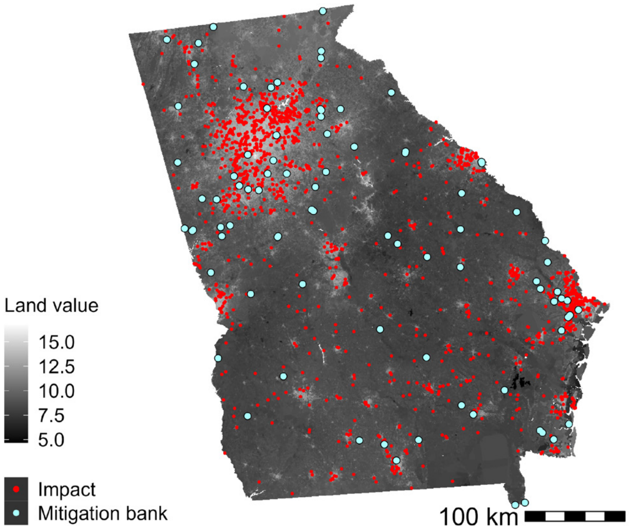
Figure 2b illustrates the relationship between land value, conservation benefit, and payback period across the contiguous US. Here, the statistics are calculated against regional means and thus represent within-region correlations.[20] First, the correlation between land value and conservation benefit is positive and monotonous across the land value distribution. Recall that the benefit measure is based on avoided flood damages which in turn are estimated from flood insurance claims. These damages increase with economic activity and population density and they are thus also highly correlated with land value. This observed positive correlation relates to the implications of Proposition 2 in our model. Namely, in a setting where the correlation between mitigation quality and entry cost is positive, it is ambiguous whether mitigation banks of high or low quality have entered the market.

Second, assessing the relative magnitudes of conservation benefit and the opportunity cost of conservation, the left panel in Figure 2b depicts how the conservation payback period evolves with land value. The relationship is nonlinear, and the graph suggests that the most favorable comparison between benefits and costs occurs between the 40th and 80th percentiles of land value.

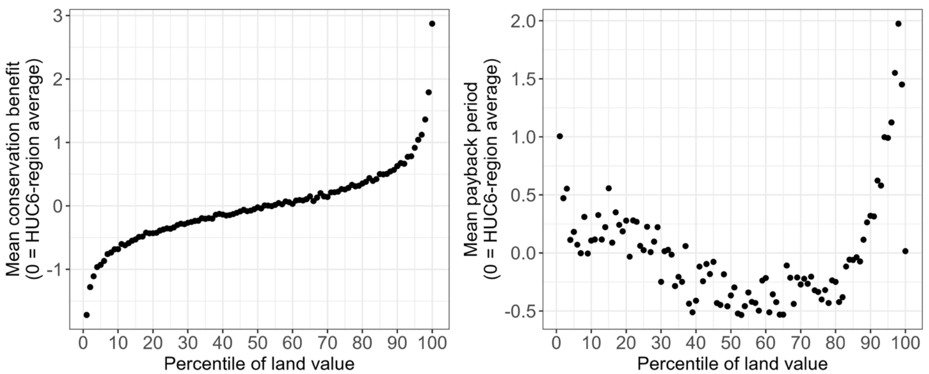
[20]Throughout the analysis, we use the 6-digit level of Hydrological Unit Code (subwatershed) as the region unit. This level best approximates the size of one service area of a mitigation bank, i.e. the geographic extent of one compensatory credit market. In practise, the service areas overlap considerably between banks. For this reason, we choose to approximate the markets with the HUC 6-digit regions.

Figure 2: Land value and conservation benefits

(a) Distribution of impacts and mitigation banks in Georgia



(b) Land value and wetland conservation benefit in the US



Data sources: Corps (2020, 2022); Nolte (2020); Taylor and Druckenmiller (2022). In panel (b), the unit of analysis is a 12-digit Hydrological Unit Code region. The statistics are calculated by averaging the data within 12-digit HUC regions and then demeaning against 6-digit HUC region means. Land value measured in log USD per hectare. Conservation benefit is measured in log USD per hectare per year. Payback period is measured in years.

Table 1 provides further descriptive statistics to compare the impact sites and mitigation bank sites in our sample. The differentials in conservation benefit and land value follow directly from the observations made in Figure 2. The average conservation benefit is substantially higher at the impact locations in comparison to the mitigation bank sites. Likewise, the average land value is higher at the impact sites than at the mitigation bank sites. However, the land value differential is not enough to offset the payback period calculation in favor of mitigation banking. The difference in the average payback period is over three years in favor of permittee-responsible mitigation.

Mitigation bank sites do, however, have a higher share of area with wetland potential. This indicates that mitigation banks likely have an advantage over PRM in terms of the variable costs of wetland restoration. It is equally important to observe the difference between the average sizes of impacts and mitigation banks. While the conservation benefit and payback period measures would favor permittee-responsible mitigation on a per-acre basis, the average cost of restoration is likely substantially lower at mitigation banks due to the influence of fixed costs and scale effects. Overall, while the payback period measure would favor PRM by a small margin, the cost-reduction potential of mitigation banking is strong and it is reasonable to assume that this feature has dictated the evolution of the compensatory mitigation program.

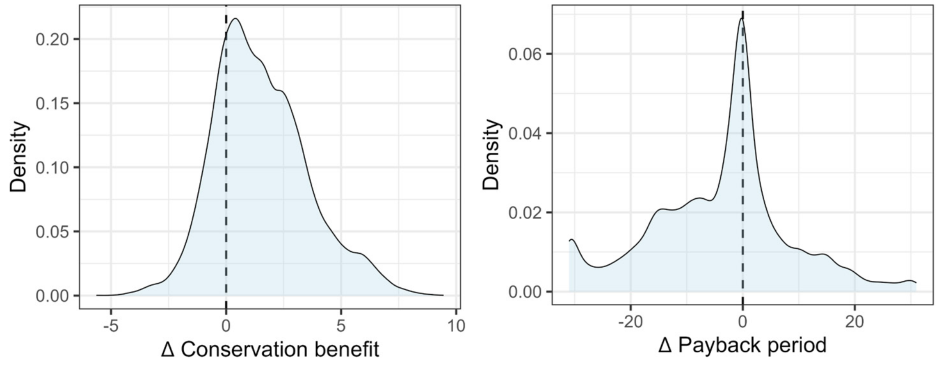
Table 1 only presents aggregate statistics, and it is important to emphasize that the choice of mitigation method depends on the specific comparison between the possibility of PRM at the impact site and the banks that service that particular impact. Figure 3 exemplifies this comparison and particularly the heterogeneity of the comparison in a given market setting. Here, for a specific impact, the features of PRM are evaluated in comparison to the attributes of all available mitigation banks that were available for compensation. These banks are those whose service areas overlap with the impact site and have credits available for acquisition. The left panel shows that the conservation benefit is strictly larger at the impact sites than at the average available mitigation bank in 76% of the observed projects. However, in terms of payback period, permittee-responsible mitigation would (strictly) dominate the average mitigation bank in 59% of the observations. Compared to the best available bank (lowest payback period), PRM would strictly dominate in only 22% of the observations. These figures demonstrate that the particular features of a given market greatly influence the comparison between PRM and mitigation banking.

Table 1: Descriptive statistics

Variable	Impact sites	Mitigation banks	Difference
Conservation benefit (US $1,000/ha/year)			
Mean	89.38	13.83	75.55***
Std. dev.	279.60	58.66	
Land value (US $1,000/ha)			
Mean	132.74	25.53	107.21***
Std. dev.	297.11	66.08	
Payback period (years)			
Mean	7.23	10.45	−3.22***
Std. dev.	9.86	12.78	
Wetland potential (% of site area)			
Mean	0.44	0.62	−0.18***
Std. dev.	0.39	0.34	
Area (acres)			
Mean	2.48	489.13	−486.64***
Std. dev.	25.61	1,362.94	
Mitigation method			
PRM	30.5 %		
MB	69.5 %		
N	21,061	1,251	

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. Welch two-sample t-test.
Sources: Corps (2020, 2022); Nolte (2020); Taylor and Druckenmiller (2022); US Environmental Protection Agency (2022)

Figure 3: Differences between characteristics of impact sites and mitigation bank sites



Δ indicates the difference between the value of the variable at the impact site and the average value among available mitigation banks. Available mitigation banks comprise banks whose service areas overlap with the impact site and have credits available for acquisition. These mitigation banks could potentially have sold credits to the permittee to compensate for the impact.

Sample: Impacts to wetlands over 2012–2020 where compensatory mitigation was required and at least one mitigation bank was available (N = 18,669).

Data sources: Corps (2020, 2022); Nolte (2020); Taylor and Druckenmiller (2022).

5.4 Choice of offsetting mechanism in practice

In this section, we turn to analyze the extent to which the factors included in the theoretical model (related to the benefits of conservation, fixed set-up costs, variable costs of conservation, and market size) predict the choice of mitigation method for individual impacts.

5.4.1 Model variables and choice of mitigation method

To analyze the relationship between the model variables and the observed choice of mitigation method, we regress the choice of mitigation method against the characteristics of PRM and mitigation banking in a given permit decision setting. In particular, we estimate a logit model as follows:

$$\log\left(\frac{\mathbb{P}_i}{1-\mathbb{P}_i}\right) = \beta_1 \Delta p_i + \beta_2 \Delta c_i + \alpha_r + \gamma_t + \varepsilon_i \tag{11}$$

where $\mathbb{P}_i$ is the conditional probability of choosing permittee-responsible mitigation to mitigate for impact i. The model variables Δp_i and Δc_i compare the characteristics of on-site mitigation to the characteristics of the most advantageous bank in the dimension of the respective variable. Δp_i is the difference between the payback period at the impact site i and the lowest payback period among available mitigation banks, and Δc_i is the difference between the wetland potential at the impact site and the highest wetland potential among available mitigation banks. Region fixed effects α_r and year fixed effects γ_t control for the cross-regional level differences and for the common evolution of regulator preferences. We use Hydrological Unit Code 6-digit (HUC6) regions as the geographic unit of the fixed effects. These HUC6 regions approximate the size of a mitigation bank service area i.e. the region where a mitigation bank can sell credits.

In a variation of the empirical model, we decompose the payback period variable to respective land value and conservation benefit variables as follows:

$$\log\left(\frac{\mathbb{P}_i}{1-\mathbb{P}_i}\right) = \beta_1 \Delta F_i + \beta_2 \Delta \delta_i + \beta_3 \Delta c_i + \alpha_r + \gamma_t + \varepsilon_i \tag{12}$$

where ΔF_i is the difference between land value at the impact site and the lowest land value among available mitigation banks and $\Delta \delta_i$ is the difference between conservation benefit at the impact site and the highest benefit among available mitigation banks.

The estimation results are presented in Table 2. Columns (1) and (2) present estimation results without region and year fixed effects. Noting the influence of regulator preferences and the large cross-regional variation in the outcome variable, these estimates should be interpreted as describing associations between regions. All estimated coefficients have the expected signs. Starting from model (1), the marginal effect of the difference in payback period is moderate in magnitude. Between the first and last deciles of the estimation sample in terms of payback period, the difference in the predicted probability of choosing PRM is 15.1% (95% CI: 11.2–18.3%). Wetland potential is also an important determinant, with the respective difference in predicted probability at 8.1% (95% CI: 6.9–9.3%) between the first and last deciles of the sample.

In model (2), the average marginal effects would imply a large influence of both land value and conservation benefit when keeping the other variable constant. However, one should note the positive correlation between land value and conservation benefit in the estimation sample. Between the first and last deciles of the sample in terms of land value difference, the difference in the predicted probability of choosing PRM is 26.8% (95% CI: 22.7–30.3%). The same statistic with respect to conservation benefit is 10.2% (95% CI: 7.8–12.8%). Here, one may infer that in the choice of mitigation method, the difference in the opportunity cost of land use plays a considerably more important role than the difference in conservation benefit.

Columns (3) and (4) present estimation results with region and year fixed effects included, accounting for the level and the common trend of regulator preferences.[21] The fixed effects also largely capture the variation in market size. Here, the estimates are substantially attenuated toward zero. Also, the predicted differences in choice probabilities between the first and deciles of the variables are attenuated. Still, the differences between the influence of land value and conservation benefit persist. According to the estimates of model (4), between the first and last deciles of the sample in terms of land value, the difference in the predicted probability of choosing PRM is 5.4% (95% CI: 6.3–4.2%) while the respective difference regarding conservation benefit is only 0.7% (95% CI: 0.01–1.5%)

Table 2: Choice of mitigation method

Dependent variable: $\mathbb{I}$(PRM chosen)

	(1)	(2)	(3)	(4)
Δ Payback period	-0.00277^{**}		-0.00129^{*}	
	(0.000936)		(0.000575)	
Δ Land value		-0.0780^{***}		-0.0311^{***}
		(0.00999)		(0.00554)
Δ Conservation benefit		0.0369^{***}		0.0153^{***}
		(0.00613)		(0.00382)
Δ Wetland potential	0.0658^{*}	0.0251	0.0332^{*}	0.0226
	(0.0328)	(0.0296)	(0.0145)	(0.0143)
Region FE	No	No	Yes	Yes
Year FE	No	No	Yes	Yes
N	18492	18268	17919	17647

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.
Average marginal effects from logistic regression. Standard errors in parentheses. Standard errors clustered at Hydrological Unit Code 6-digit level. Region fixed effects at Hydrological Unit Code 6-digit level.
Δ indicates the difference between the value of the variable at the impact site and the most favorable value among available mitigation banks (lowest payback period, lowest land value, highest conservation benefit, or highest wetland potential).
Sample: Impacts to wetlands over 2012–2020 where compensatory mitigation was required and at least one mitigation bank was available.
Data sources: Corps (2020, 2022); Nolte (2020); Taylor and Druckenmiller (2022); US Environmental Protection Agency (2022).

Table B.1 in the Appendix presents results from regressions where the explanatory variables are constructed as the difference between the value of the variables at the impact site and the average value among available mitigation

[21] Some observations are dropped due to there being only a single observation within a Hydrological Unit Code 6-digit region.

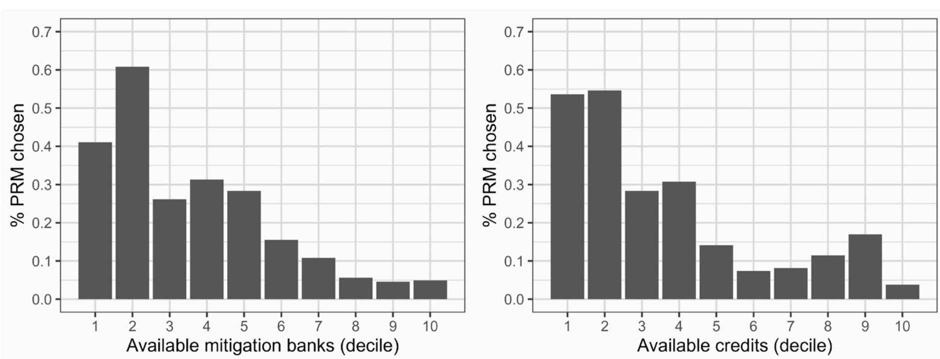
banks (instead of comparing against the most favorable value among available mitigation banks as in Table 2). In these estimates, we observe the expected association only with respect to land value.

In sum, the estimated associations between model variables and the choice of mitigation method, as depicted in Table 2, are in the expected direction and explain well regional differences in the choice of mitigation method. Nonetheless, the model variables offer limited explanatory power when examining within-region variation. In particular, regulator preferences and geography are both important determinants in the choice setting but are absorbed in region fixed effects.

5.4.2 Market size

The effects of the model variables are mediated through supply and demand dynamics and in particular the market entry of mitigation banks. Figure 4 visualizes how the availability of compensation credits strongly predicts the choice of mitigation method. The left panel shows the relationship between the number of credit suppliers and the choice of method. The right panel displays the relationship regarding available credits, adjusting for credit demand. In both panels, it is evident that mitigation banking is the predominant mitigation method in regions where the market is dense and there is an ample supply of compensation credits.

Figure 4: Choice of mitigation method and market size



Sample: Impacts to wetlands over 2012–2020 where compensatory mitigation was required and at least one mitigation bank was available (N = 18,669).

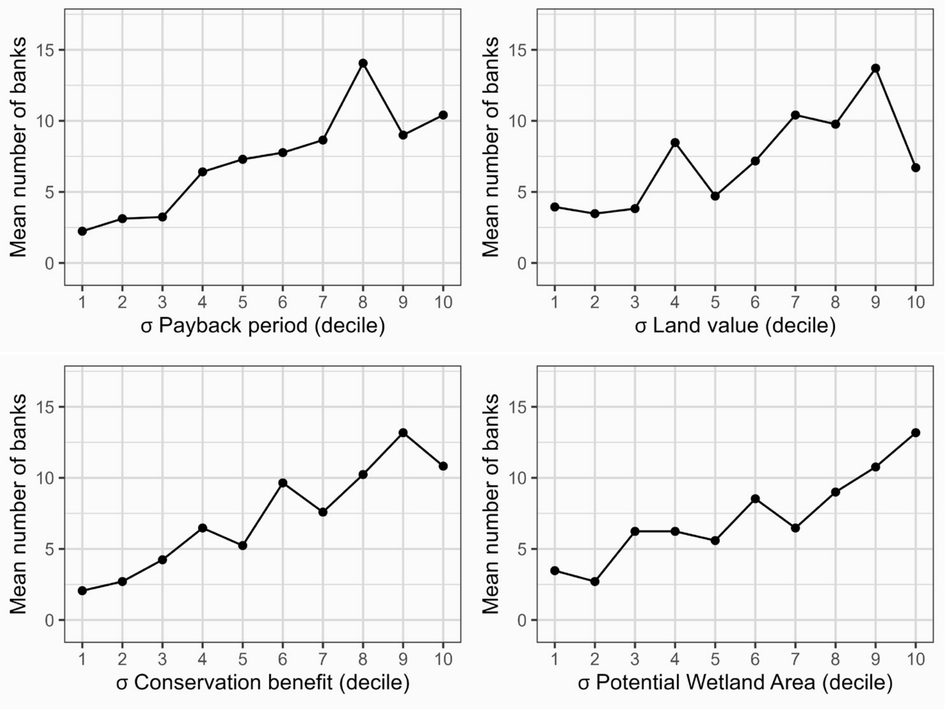
Data sources: (Corps, 2020, 2022)

The stylized finding in Figure 4 may equally well reflect regulator preferences or the fundamentals in our theoretical model. On one hand, banks are more likely to enter the market in regions where the regulator favors mitigation banking. On the other hand, in regions with a large number of banks, there is a stronger likelihood that at least one available mitigation bank dominates on-site mitigation in terms of costs and benefits. This is illustrated in Figure B.1 in the Appendix. The figure shows how each measure becomes more favorable towards mitigation banking as the number of banks increases. While this relationship can be purely mechanic from a statistical perspective (the difference between on-site mitigation and the best available mitigation bank increases as the number of banks increases), it remains a valid question whether regions with more apparent exploitable gains are more likely to attract mitigation banks.

To explore this mechanism, Figure 5 plots the number of mitigation banks against the standard deviation of the model variables as measured within a Hydrological Unit Code 6-digit region. High within-region variability in the characteristics increases the likelihood of there being suitable locations for off-site mitigation that outperform on-site mitigation. Accordingly, the graphs show a positive correlation between the within-region variability in the characteristics and market size. Figures B.2 and B.3 in the Appendix display a similar positive association between the standard deviations of model variables and compensation credit surplus, normalizing for credit demand.

To summarize, the data display patterns where the potential benefits and costs of mitigation activities are correlated with the size of the compensatory mitigation market. Notwithstanding, all data presented in this subsection are summarized over the entire lifespan of the compensatory mitigation program. Further analysis should examine how the characteristics of on-site and off-site mitigation evolve over time, and in particular, if the patterns presented in this section are stable over the course of the program.

Figure 5: Standard deviations of model variables and market size



Mean number of mitigation banks within Hydrological Unit Code 6-digit regions by deciles of the standard deviation of model variables.

Sample: Hydrological Unit Code 6-digit regions with at least one mitigation bank (N = 170).

Data sources: Corps (2022); Nolte (2020); Taylor and Druckenmiller (2022); US Environmental Protection Agency (2022)

6 Summary and conclusions

In this paper, we provide an exploration of the costs and benefits of two of the most common approaches for offsetting biodiversity losses, namely, developer-led offsets and mitigation banking. We investigate the incentives that inform the characteristics of banks participating in a mitigation banking program and compare benefits and costs from either approach.

One commonly invoked advantage of mitigation banking is that it is capable of mainstreaming conservation efforts. The high priority that is put on development and economic prosperity often means that investments in biodiversity conservation do not receive the financial support that they need to succeed (Johnson et al., 2017). The unsystematic, or piecemeal, approach to offsets performed by developers may not gather the necessary support because it fails to capture the gains from scalable, or diffusable, offsets that are provided through a market approach. Although our results confirm that there are cost savings from having a larger number of banks contributing to the same conservation objective, the cost-effectiveness of a market approach is nevertheless mediated by differences in quality of restoration performed by developers and banks. It is not accidental that regulators often favor the use of on-site and in-kind replacement of lost environmental values. Many authors highlight the risk of discontinuity between offsets at impact sites and those at separate mitigation sites, which may lead to a change in the spatial distribution of benefits delivered by environmental amenities (Ben-Dor et al., 2007; Ruhl and Salzman, 2006). This means that restoration by banks may provide an insufficient replacement for lost biodiversity values. In that case, we find that developer-led offsets could be the preferred policy option. However, should quality differentials between banks and developers exist, so that banks would be required to implement proportionately more restoration than developers, our results indicate that the higher cost of meeting NNL through banking could be counterbalanced by having the program be populated by a larger number of firms, due to the cost-reduction effect of market expansion. This result nevertheless hinges on the assumption that environmental benefits from off-site locations are still transferable.

We complemented this analysis by considering the role of heterogeneous quality of restoration, not just between the developer and banks, but also among banks. Although all banks perform restoration off-site, there might be differences in the baseline quality of land held by individual banks (and thus in the marginal value of restoration activities), as well as the type of offsets that are feasible in any given site. Because we assume that restoration effort is distributed in a cost-minimizing way, as long as there is one bank that is capable of performing offsets that adequately replace lost environmental values, and that restoration at that location is of sufficient quality to match that of the developer, then the decentralized approach would always perform better than developer-led offsets. Again, if restoration by banks is of insufficient quality and benefits are transferable, a large enough market could work to decrease the quality requirement of banks, and thus make offsets by banks preferable to those by the developer.

Another frequently cited advantage of mitigation banking is its ability to carry out large-scale restoration. The size of restoration projects has been highlighted as the single most important criterion for the success of offsets (Moreno-Mateos et al., 2012b). In our analysis, we do not examine the role of having one large bank serving multiple developers. Rather, we consider a representative developer and analyze the performance of a market of varying sizes in compensating a fixed amount of damages, given that benefits from restoration may exhibit nonconstant returns to scale. We find that if benefits from restoration exhibit increasing returns to scale, one bank with a large restoration project would incur a lower cost than that accrued from several banks implementing smaller restoration projects. Nevertheless, increasing returns to scale alone is not sufficient to disavow market expansion, since the magnitude of the cost-reduction effect from a larger market might be greater than the cost increase associated with having smaller restoration projects. If the scale effect is more significant, however, then having one firm serving all the offsets could be optimal, provided that the quality of its restoration is commensurate to that of the developer.

Our final set of results relates to the entry decision of banks into the MB market. Because restoration is distributed in a cost-minimizing way, high-quality banks will perform the greatest amount of restoration work. However, because the trading ratio is set according to the rate of replacement between damages and restoration at any given site, and, despite the fact that high-quality banks perform higher restoration and thus face a higher cost, since the trading ratio will be skewed in their favor, high-quality banks stand to gain the most from selling their damage-equivalent restoration credits. That is, the post-entry surplus of high-quality banks is higher than that of low-quality banks. Thus, should all banks face entry costs that are negatively correlated with quality, the market will be populated by banks exhibiting the highest possible quality, which is an outcome that works in favor of mitigation banking. If otherwise, and land with high restoration quality is also that with the greatest value for uses other than restoration (i.e., a higher entry cost), then the market might instead attract firms of low quality.

In our empirical analysis of the US wetland mitigation banking program, we find that wetland conservation benefits are positively correlated with entry costs. A second empirical finding is that while the cost differential between on-site and off-site restoration clearly favors the latter, the benefit/cost comparison, measured as the payback period of wetland conservation, is similar between the two options. In an analysis of choices between on-site and off-site restoration in permit decisions for impacts, we find that the choice is more strongly associated with cost savings than with environmental benefits. Moreover, we find that the potential for reduced costs and increased benefits is positively correlated with the number of credit providers in the mitigation market.

Our analysis provides policy makers with a theoretical structure that informs the choice of developer-led versus market-led offsets. While it is true that in both instances the regulator may be able to set enforceable restoration standards, quality differentials between the developer and banks, as well as entry incentives, dictate the choice of one policy versus the other. By implementing a market approach, the regulator must ensure that banks that self-select into participating in the program are those that offer the greatest potential for meeting the NNL requirement at the least possible cost. And, if land for restoration by banks is of insufficient replacement quality, whether the market will be sufficiently large to compensate for the lack of equivalency so that cost-savings from a market approach could work to decrease the burden of meeting NNL.

A Nonlinear benefit function

Here, we consider the general case where function $B_i(a) = \delta_i f(a)$ may exhibit either increasing ($f'' > 0$) or nonincreasing ($f'' \leq 0$) returns to scale.

As with the linear benefit function, should the MB program only contemplate one firm ($n = 1$), the comparison between MB and PRM relies only on the value of δ_0 of the developer and δ_i of the participating bank. Because the level of abatement employed by either the developer or the bank follows from the constraint $\delta f(a) = D$, where $da/d\delta < 0$, then, given that both share the same variable cost of restoration $c(a)$, the least cost policy will correspond to the one which requires the least amount of abatement. That is, that which involves the higher value of δ. Thus, for $n = 1$, irrespective of scale effects, if $\delta_i \geq \delta_0$, the MB program will fare better than PRM, and worse otherwise.

As for $n > 1$, from the equimarginal principle in (2), abatement by any two firms i and j participating in the MB program will be such that:

$$\frac{c'(a_i^*)}{\delta_i f'(a_i^*)} = \frac{c'(a_j^*)}{\delta_j f'(a_j^*)} \quad \text{for } i \neq j \tag{13}$$

assuming that $c''/c' > f''/f'$, i.e., the cost-reduction effect is more pronounced than the scale effect, so that $\{a_i^*\}$ from (13) is indeed optimal. If so, the relationship between a_i^* and a_j^* is such that if $\delta_i > \delta_j$ then $a_i^* > a_j^*$. That is, the bank with the highest quality will perform the highest level of abatement.[22]

Moreover, irrespective of scale effects, it is always true that higher values of δ decrease the overall burden with meeting NNL. All else equal, the higher the δ, the lower the total abatement necessary to compensate damages D. This observation, combined with the result from Proposition 1 of how costs vary with market size, allows us to establish the following general result regarding the comparison between MB and PRM, from which Results 1 and 2 introduced earlier are special cases:

Proposition 3: *Suppose that $B_i(a_i) = \delta_i f(a_i)$.*

(a) If $n = 1$ and $\delta_i \geq \delta_0$, then MB $\succsim$ PRM.

(b) If $n > 1$ and $c''/c' \geq f''/f'$, the larger the market, the lower the individual restoration values δ_i required for MB $\succsim$ PRM

Proof: Follows from $\partial C/\partial \delta_i = -\lambda^* a_i^* < 0$ and $\partial \lambda^*/\partial n < 0$ by Proposition 1 □

[22]If $c''/c' \leq f''/f'$ then the cost-minimizing allocation of abatement cannot be found from the equimarginal principle. If scale effects are very pronounced, it could be optimal that the regulator forfeit restoration from $n-1$ banks and instead only allow that one bank perform abatement.

If banks exhibit decreasing returns to scale, the fact that (1) costs decrease with higher restoration values and (2) costs decrease with a larger market, implies that a lower level of individual quality would be required for MB to fare better than PRM. As discussed in the linear benefit case, the larger the market, the lower the quality requirements of any individual bank (and in specific that of the more productive bank) to ensure that MB $\succsim$ PRM.

If banks exhibit strong increasing returns to scale, the fact that (1) costs decrease with higher restoration values but (2) costs are not necessarily decreasing with a larger market, might imply that a higher level of individual quality be required for MB to fare better than PRM. We illustrate the role of scale in dictating minimum-quality thresholds with the following example.

Example: Suppose that $c(a) = a^{\beta}$, where $\beta > 1$, and $B_i(a_i) = \delta_i f(a_i)$ for $i \in \{0, 1, \ldots, n\}$, where $f(a) = a^{\alpha}$ and $\alpha \geq 0$. Observe that if $0 \leq \alpha \leq 1$, benefits exhibit nonincreasing returns to scale, and if $\alpha > 1$, benefits are increasing with scale. Further assume that $\delta_i = \delta_j$ for all banks i and j in $\{1, \ldots, n\}$. Whereas restoration by the developer has a value of δ_0, all $n \geq 1$ banks in the program exhibit the same restoration value, which we denote by δ_1.

Under PRM, a_0' is recovered from the constraint $B_0(a_0') = D$, or $a_0' = (D/\delta_0)^{\frac{1}{\alpha}}$. Under MB, and imposing symmetry, abatement a^* is recovered from the constraint $nB_1(a) = D$, or $a^* = (D/n\delta_1)^{1/\alpha}$. Observe that individual abatement a^* by any one bank is always decreasing with n, but the total amount of abatement

$$na^* = n^{\frac{\alpha-1}{\alpha}} (D/\delta_1)^{\frac{1}{\alpha}}$$

is (1) fixed, if benefits exhibit constant returns to scale $\alpha = 1$, (2) increasing in n, if benefits exhibit increasing returns to scale $\alpha > 1$, and (3) decreasing in n, if benefits exhibit decreasing returns to scale $\alpha < 1$.

In terms of the comparison between policies, MB fares better than PRM if:

$$W^* - W' = c(a_0') - nc(a^*) \geq 0 \quad \text{or} \quad \delta_1 \geq \frac{\delta_0}{n^{\frac{\beta-\alpha}{\beta}}}. \tag{14}$$

Note that the magnitude of the cost-reduction effect is $c''/c' = \beta$ and that of the scale effect is $f''/f' = \alpha$. It follows from (14) that, if $n = 1$, then $\delta \geq \delta_0$ for MB $\succsim$ PRM, irrespective of α and β; but if $n > 1$, then, by denoting the threshold $\delta_1(n)$ from meeting (14) at equality, for $\delta_1 \geq \delta_1(n)$, we have MB $\succsim$ PRM. Moreover, $\delta'(n) \leq 0$ if $\beta \geq \alpha$ and $\delta'(n) > 0$ if $\beta < \alpha$. Thus, if the magnitude of the scale effect is more pronounced than that of the cost-reduction effect, increasing the size of the market will require that individual quality of any one bank be larger for MB to fare better than PRM.[23]

B Supplementary tables and figures

Table B.1: Choice of mitigation method
Permittee-responsible mitigation vs. the average available mitigation bank

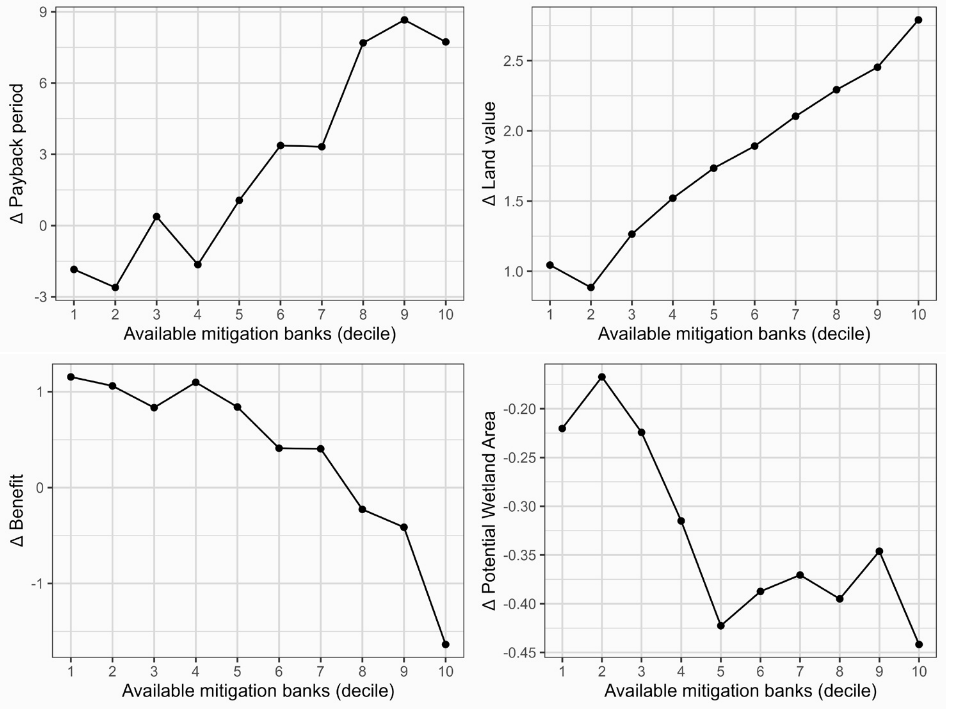
Dependent variable: $\mathbb{I}$(PRM chosen)

	(1)	(2)	(3)	(4)
Δ Payback period	0.00233*		−0.000410	
	(0.00100)		(0.000621)	
Δ Land value		−0.0301*		−0.0173*
		(0.0140)		(0.00681)
Δ Conservation benefit		−0.00260		0.00493
		(0.0119)		(0.00519)
Δ Wetland potential	−0.00248	−0.00594	0.0151	0.0100
	(0.0331)	(0.0326)	(0.0200)	(0.0198)
Region FE	No	No	Yes	Yes
Year FE	No	No	Yes	Yes
N	18492	18268	17919	17647

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.
Average marginal effects from logistic regression. Standard errors in parentheses. Standard errors clustered at Hydrological Unit Code 6-digit level. Region fixed effects at Hydrological Unit Code 6-digit level.
Δ indicates the difference between the impact site and the average available mitigation bank (e.g. the difference between land value at the impact site and the average land value among available mitigation banks).
Sample: Impacts to wetlands over 2012–2020 where compensatory mitigation was required and at least one mitigation bank was available.
Data sources: Corps (2020, 2022); Nolte (2020); Taylor and Druckenmiller (2022)

[23] Note, however, that the symmetric allocation of abatement in case $\alpha > \beta$ is not the cost-minimizing distribution of abatement. In fact, if the regulator were to allow only one bank to perform restoration, so that $\hat{a}_i = D/\delta_1$ and $\hat{a}_j = 0$ for all $j \neq i$, then $nc(a^*) - c(\hat{a}) > 0$.

Figure B.1: Market size and model variables

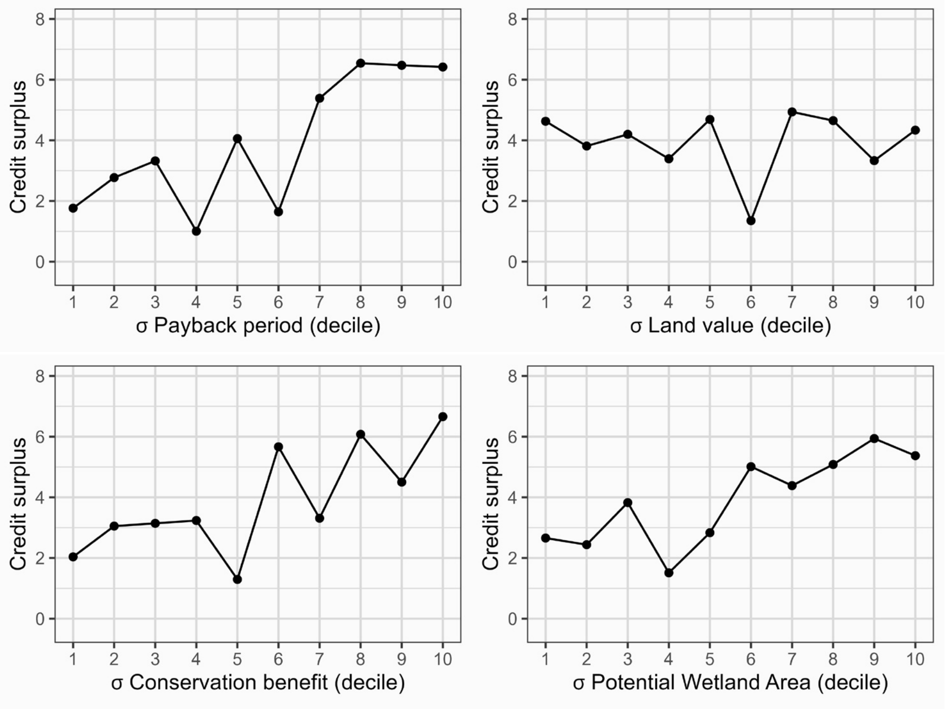


Δ indicates the difference between the value of the variable at the impact site and the most favorable value among available mitigation banks (lowest payback period, lowest land value, highest conservation benefit, or highest wetland potential).

Sample: Impacts to wetlands over 2012–2020 where compensatory mitigation was required and at least one mitigation bank was available (N = 18,669).

Data sources: Corps (2020, 2022); Nolte (2020); Taylor and Druckenmiller (2022)

Figure B.2: Standard deviation of model variables and credit surplus

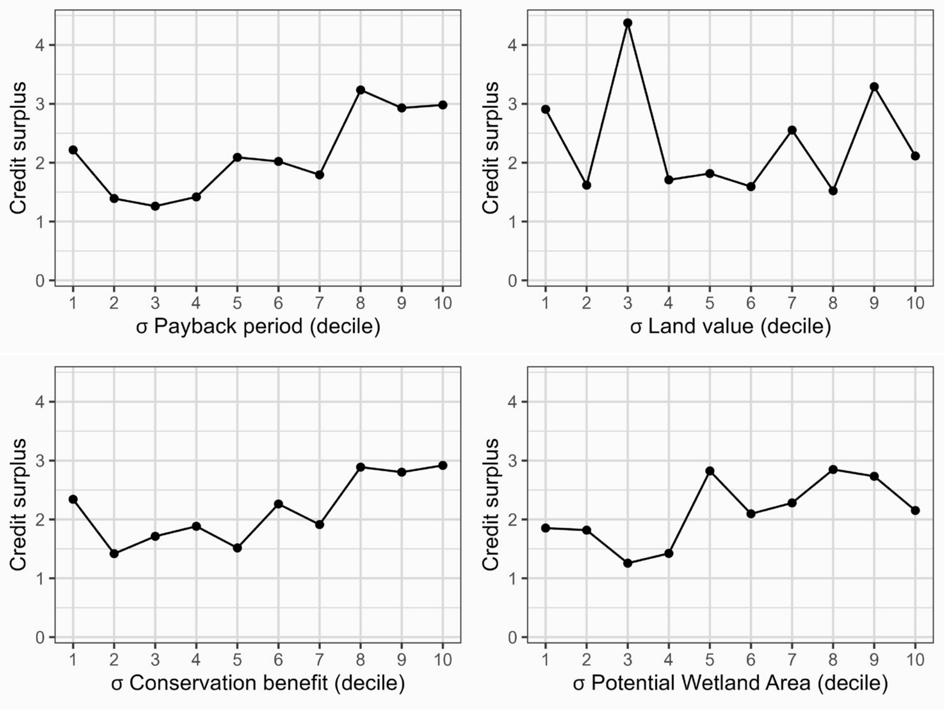


Relationship between standard deviations of model variables and compensation credit surplus within a region.

Sample: Hydrological Unit Code 6-digit regions with at least one mitigation bank (N = 170).

Data sources: Corps (2022); Nolte (2020); Taylor and Druckenmiller (2022); US Environmental Protection Agency (2022)

Figure B.3: Standard deviation of model variables and credit surplus normalized by credit demand



Relationship between standard deviations of model variables and compensation credit surplus within a region divided by credit withdrawals since January 2021.

Sample: Hydrological Unit Code 6-digit regions with at least one mitigation bank (N = 170).

Data sources: Corps (2022); Nolte (2020); Taylor and Druckenmiller (2022)

Chapter 3

The Effect of Water Resource Protection on Construction Employment in the United States

Abstract

This paper examines the labor market effects of federal regulations that protect water resources in the United States. The Clean Water Rule, an executive order enacted in 2015, expanded the scope of waters that are federally protected under the Clean Water Act. I use a difference-in-differences framework to compare construction employment between the 22 states where the Rule was implemented and the 28 states where it was never implemented due to litigation in regional courts. I find that the overall effect of the Rule on construction employment was negligible. However, a negative effect appears in states that had unsuccessfully litigated against the Rule. Furthermore, the decrease in construction activity was most prominent in counties where low-cost compliance options through environmental offset markets were limited.

1 Introduction

The US Clean Water Act (CWA) establishes environmental protections for water resources that fall under its jurisdiction. The exact scope of this jurisdiction has been a contentious political issue since the adoption of the Act in 1972, largely due to the costs imposed on regulated firms. In August 2015, the Obama administration enacted the Clean Water Rule (*CWR*, *the Rule*), an executive order that sought to clarify and expand the scope of federally protected waters. While some states welcomed the extended environmental protections, others had the view that the jurisdictional expansion caused undue costs and development constraints for agriculture, housing, and infrastructure construction *inter alia* (EPA and DA, 2021). Consequently, the Rule faced legal challenges in several states, delaying its implementation. Ultimately, the Rule was implemented in 22 states and DC from August 2018 until December 2019, when the Trump administration repealed it nationwide.

In this paper, I measure the economic costs of the regulatory shift of the Clean Water Rule in the construction industry. The US Environmental Protection Agency (EPA) has determined that the Clean Water Act regulatory program—and, consequently, the Clean Water Rule—has the largest cost implications for this sector (EPA and DA, 2021). To identify effects, I exploit the differential adoption of the Rule across the US. In particular, within a difference-in-differences framework, I compare construction employment in states that implemented the Rule during 2018–2019 against states where the Rule was never implemented.

The short lifespan of the Clean Water Rule precludes the estimation of any long-term economic costs or environmental benefits directly attributable to increased environmental protections. In contrast, an unexpected increase in regulatory stringency, even if temporary, still has economic implications if firms need to adjust or postpone their planned projects. A major challenge in assessing the costs of such regulations is the difficulty of determining where the increased stringency becomes a constraint for firms. Regarding water resources, the differences in geography and state-level regulatory practices are important determinants that are also difficult to assess ex ante (EPA and DA, 2021; Keiser et al., 2022). This ex-post assessment offers insights into the distribution of regulatory costs across regions and informs the ongoing policy debate about the stringency and decentralization of aquatic resource protections.

There are two main findings from the analysis. First, the overall effect of the Clean Water Rule on construction employment was small or negligible in magnitude in the implementing states. Second, non-negligible effects arise in the four states that had unsuccessfully litigated against the implementation of the Rule (Michigan, Ohio, Oklahoma, and Tennessee). In these states, the federal Clean Water Rule surpassed state-level protections for aquatic resources. At the same time, administrative data characterizes these four states as having a high prevalence of water resources subject to the jurisdictional expansion.

Further heterogeneity analysis suggests that the Rule had negative effects in counties where compliance options were limited and where the compliance costs consequently were likely to have been high. The majority of the costs of being regulated under the Clean Water Act come from compensatory mitigation, whereby permittees must compensate for adverse impacts on jurisdictional waters. In the economic analyses from the EPA, compensatory mitigation costs have amounted to over 60% of the total cost of the program (EPA and DA, 2015, 2020, 2021). A unique feature of the CWA compensatory mitigation program is that a market mechanism mediates the majority of the compensation activities. To comply with their CWA permit conditions, permittees purchase compensatory credits from third-party firms (*mitigation banks*) that have generated the credits in advance from conservation activities. The supply of these credits is fixed in the short term—particularly when facing a demand shock such as the jurisdictional expansion in the Clean Water Rule. My estimation results suggest that the decrease in construction employment which did take place is driven by counties where the stock of compensation credits available for purchase was low. Conversely, in counties where the stock of credits was high, there is no estimated decrease in employment. The same pattern emerges regarding the availability of a regulatory mechanism (*in-lieu fee programs*) that provides additional flexibility to the supply of compensation credits.

Two important caveats qualify the findings. First, the estimated effects take place within a 16-month window during which the Clean Water Rule was implemented. This period is short for evaluating firm decisions as the planning and completion of a construction project can take several years. A second and related caveat is that when the Clean Water Rule was finally implemented, the Trump administration had already drafted an executive order that would eventually repeal the Rule. This implies that any observed effects in firm behavior may reflect postponing of construction projects rather than their outright cancellation. Even when considering these caveats, this analysis demonstrates firms' reaction to an unexpected change in the stringency of environmental regulations and how heterogeneity in available compliance options modified that reaction.

The present analysis relates to two distinct strands of literature: the literature on the labor market impacts of environmental regulation and the literature on environmental market design. There have been numerous studies estimating the labor market effects of environmental regulations. (Berman and Bui, 2001; Curtis, 2018, 2020; Ferris and Frank, 2021; Ferris et al., 2014; Greenstone, 2002; Hafstead and Williams III, 2018; Sheriff et al., 2019; Walker, 2013). I follow this literature by examining how the Clean Water Rule impacted the construction industry, which was directly affected by the associated compliance costs. The literature on the design of environmental offset markets is active and growing, with several recent contributions from ecological and economic perspectives (Needham et al., 2019b; Simpson et al., 2021a, 2022, 2021c; zu Ermgassen et al., 2020). However, there are few empirical studies on offset markets, and the existing studies primarily examine the environmental success of offsetting projects (Levrel et al., 2017; Tillman et al., 2022). This analysis is novel in connecting the design of a compensatory credit market to firm behavior and employment outcomes.

By providing an analysis of the economic costs of environmental regulation, my analysis also contributes to the ongoing policy debate regarding the scope and definition of federally protected water resources (Keiser et al., 2021, 2022). Recent studies have discussed and evaluated the economic benefits of aquatic resource protection in the US (Keiser, 2019; Keiser and Shapiro, 2019; Taylor and Druckenmiller, 2022). Although my analysis suggests non-negligible regulatory costs in specific settings, the analysis also demonstrates how regulatory design features can mitigate these costs.

The remainder of this paper proceeds as follows. Section 2 explains the policy framework. Section 3 describes the data used in the analysis. Section 4 describes the empirical strategy and Section 5 presents the estimation results. Section 6 concludes.

2 Policy background

2.1 The Clean Water Act and federal jurisdiction

The Clean Water Act of 1972 (CWA) is the foundation of aquatic resource protection in the US. The federal statute establishes a number of regulatory programs, including provisions on water quality standards (CWA Section 303), oil spills (CWA Section 311), pollutant discharges (CWA Section 402), and dredging and filling activities (CWA Section 404). The implementation of these programs is delegated to the US Environmental Protection Agency and the US Army Corps of Engineers. The purpose of the programs is to restore and maintain the “physical, chemical and biological integrity” of water resources considered as “the Nation’s waters”, also denoted as “the waters of the United States”.[24]

The exact definition of “the waters of the United States” and thus the jurisdictional scope of the federal regulations has been a subject of political and legal debate since the adoption of the Clean Water Act. There is a general agreement that large interstate water bodies, such as the Great Lakes or the Colorado and Mississippi Rivers, as well as river networks and wetlands directly connected to

[24] 33 US Code 1251(a); 33 US Code 1362(7).

them, fall under federal jurisdiction. The debate has focused on whether CWA jurisdiction should cover minor waters that have no direct connection to federal waters but may still indirectly affect their ecology or hydrology. Courts have particularly evaluated the jurisdictional status of isolated wetlands, as well as of intermittent and ephemeral streams.[25] (Keiser et al., 2022)

In May 2015, the Obama administration adopted the Clean Water Rule, an executive order which clarified the definition of "the waters of the United States", in part by codifying existing case law and implementation practices. The politically controversial feature of the CWR was that it categorically expanded federal jurisdiction to previously unprotected water resources, including certain categories of isolated wetlands as well as intermittent and ephemeral streams. As a result, 32 states challenged the Rule in various courts, and the Court of Appeals for the Sixth Circuit granted a stay of the Rule in all of the US. The Supreme Court lifted this stay in January 2018. However, the Trump administration had already adopted an executive order to suspend the Clean Water Rule nationwide. In August 2018, the Supreme Court overturned the suspension and thus enabled the implementation of the Clean Water Rule.[26] With several regional stays still in place, the Rule now became effective in 22 states and the District of Columbia. Finally, in December 2019, the Trump administration repealed the Clean Water Rule, with the intent to adopt its own jurisdictional definitions at a later time.

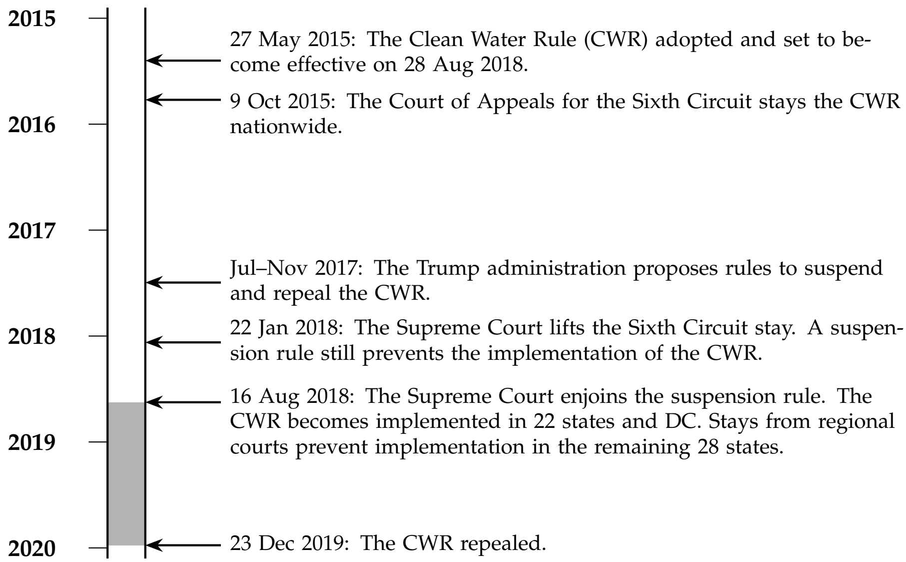
Figure 1a depicts the timeline of these events, and Figure 1b displays the states where the Clean Water Rule was implemented from August 2018 to December 2019. It is important to note that several of these states already had their own regulations in place that partially exceeded the level of protections afforded in pre-2015 implementation practice (Environmental Law Institute, 2013). Still, in several states, the CWR imposed an increase in regulatory stringency. For example, ten out of the 22 states where the CWR was implemented had stringency prohibitions in place. These prohibitions prevented state authorities from enacting regulations that exceed the federal standard unless specific criteria were satisfied. Such prohibitions were also in place in Michigan, Ohio, Oklahoma, and Tennessee, i.e. the states that unsuccessfully litigated against the Rule (Environmental Law Institute, 2013). Regardless, the initiated litigation actions are likely the best indicator of a wide gap either between the federal and state-level protections or between the perceived costs and benefits of the jurisdictional expansion.

[25] See *Solid Waste Agency of Northern Cook County v. U.S. Army Corps of Engineers*, 531 U.S. 159 (2001) and *Rapanos v. United States*, 376 F.3d 629 (6th Circuit 2004). Intermittent streams flow only during a certain time of the year and the water table is above the stream bed only during this time. Ephemeral streams flow only after precipitation events and the water table is below the stream bed year-round.

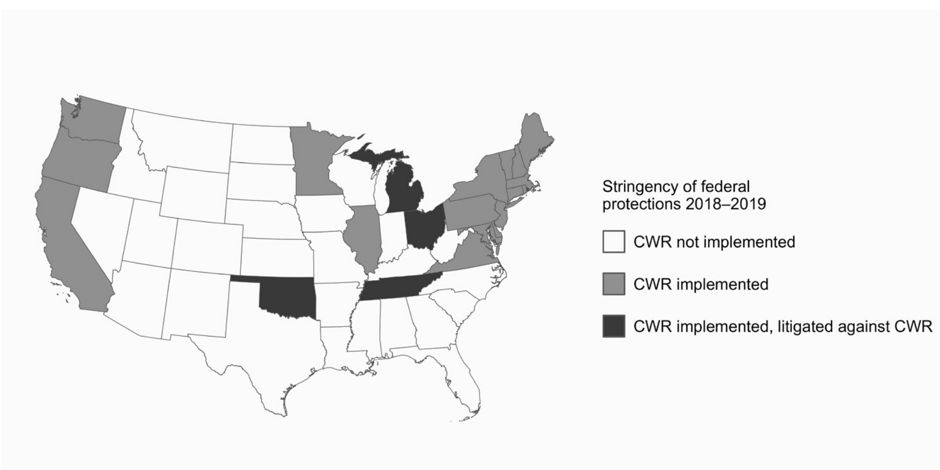
[26] *South Carolina Coastal Conservation League v. US EPA*, 318 F. Supp. 3d 959 (District Court of South Carolina, 2018).

Figure 1: Implementation of the Clean Water Rule

(a) Timeline of events



(b) States that implemented the Clean Water Rule



The map indicates the 22 states where the Clean Water Rule was implemented between 16 August 2018 and 23 December 2019. Out of these states, Michigan, Ohio, Oklahoma, and Tennessee unsuccessfully litigated against the Rule in regional courts.

I take this differential into account in the analysis. In the following section, I discuss the cost implications of expanding the scope of jurisdictional aquatic resources.

2.2 Compliance costs and compensatory mitigation

The majority of the costs related to the Clean Water Rule arise from the Clean Water Act Section 404 regulatory program. The Section 404 program regulates discharges of dredged or fill material into waters of the United States. Activities that typically require a permit under Section 404 include urban development, water resource projects (such as dams and levees), infrastructure development, and mining. In the economic analysis for the Clean Water Rule, the highest total cost estimate across all CWA programs amounted to \$465.0 million per year (fiscal year 2014 dollars), of which costs related to the Section 404 program constituted \$374.9 million (EPA and DA, 2015). In a subsequent economic analysis, the construction sector (NAICS 23) was identified as the most affected industry (EPA and DA, 2021).

For project developers, regulatory costs can be divided to permit process costs and compliance costs. Permit process costs arise from the delineation and survey of jurisdictional waters; identifying possible impacts on these waters; planning the project in a manner that avoids or minimizes these impacts; developing a plan to compensate for the impacts; and finally submitting a complete application. In the economic analysis for the Clean Water Rule, for an individual project, the highest cited estimates of permit process costs were \$62,000 in fixed costs plus \$16,800 per acre of impact (EPA and DA, 2015).

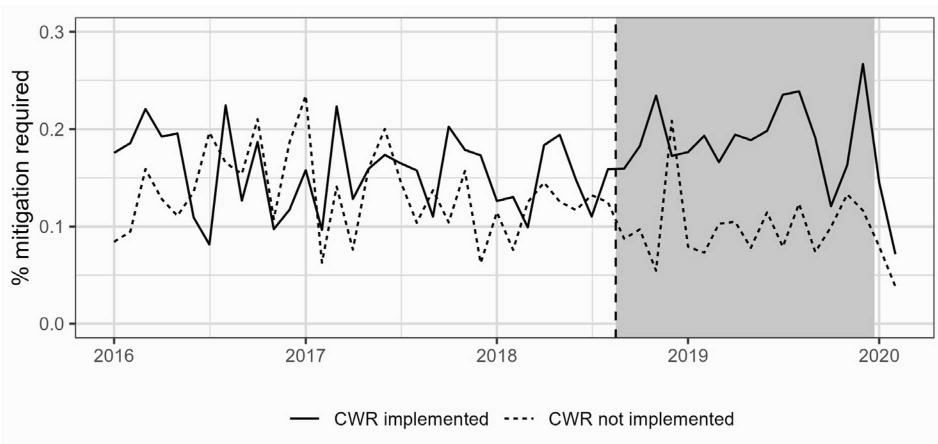
Compliance costs arise particularly from *compensatory mitigation* obligations. Compensatory mitigation entails that every adverse impact on jurisdictional waters is mitigated by replacing or providing a substitute aquatic resource. Three mechanisms are available for compensatory mitigation: (1) mitigation banking, (2) in-lieu fee programs, and (3) permittee-responsible mitigation.

Mitigation banks are sites where third-party firms restore, establish, enhance, or preserve aquatic resources in order to generate compensation credits. The firms sell the credits to CWA permittees who, in turn, use the credits to comply with their compensatory mitigation obligations. An important aspect of mitigation banking is that a compensation project must be successfully completed before any credits are awarded. In other words, the compensation must take place in advance of the impacts. This implies that the credit supply is fixed in the short run. It typically takes from two to five years for a mitigation bank to obtain approval and up to ten years to complete the restoration activities (BenDor and Riggsbee, 2011; Levrel et al., 2017).

In-lieu fee (ILF) programs provide temporal flexibility in the credit supply. ILFs are public or non-profit entities that, similar to mitigation banks, collect funds through sales of compensation credits to CWA Section 404 permittees. The distinctive feature of ILFs is that they can receive and sell credits in advance of the compensation project. As I will demonstrate below, access to ILF programs was lower in the states that had challenged the Clean Water Rule.

In permittee-responsible mitigation (PRM), the permittee itself (or an authorized contractor) undertakes a compensation project. In contrast to mitigation banking and ILFs, the permittee maintains full legal responsibility for the success of the compensation project. In the early years of the compensatory mitigation program, PRM was the most prevalent compensation method. However, several reports characterized PRM as expensive for permittees, ineffective in terms of achieving desired ecological outcomes, and accompanied by a high rate of noncompliance (Council et al., 2001; GAO, 2005). For these reasons, the EPA and Corps generally consider mitigation banking and ILFs preferable to PRM (EPA and DA, 2021). Nonetheless, when mitigation credits are not available, PRM remains the only option for a developer to compensate for impacts.

Figure 2: Compensatory mitigation requirements on water resources affected by the Clean Water Rule



Shares of Clean Water Act Section 404 permit decisions where compensatory mitigation was required for impacts on water resources affected by the Clean Water Rule (isolated wetlands and intermittent and ephemeral streams). The shaded region indicates the implementation of the Rule from 16 August 2018 to 23 December 2019. Data source: Corps (2020)

The economic analysis for the Clean Water Rule identified compensatory mitigation costs as ranging from $41,572 to $111,985 per acre of wetland mitigation and $95 to $1,000 per linear foot of stream mitigation (2014 dollars). Out of the total regulatory costs that project developers are facing, compensatory mitigation costs were estimated to comprise approximately 70–78% (EPA and DA, 2015). Figure 2 shows how the Clean Water Rule resulted in increased compensatory mitigation requirements in Section 404 permits for the water resources it affected. During the implementation of the Rule, the total share of permits requiring mitigation was 19.3% in implementing states and 10.2% in the non-implementing states. Before the implementation of the Rule, the difference between the two groups was 2.3%. As a consequence, project developers in affected states faced higher costs associated with compensatory mitigation. Given the inelastic short-run supply of compensatory credits, this demand shock may have increased compliance costs disproportionately in regions where fewer credits were available. In the regression analysis, I explore the degree to which credit availability modified the labor market effects of the Clean Water Rule.

3 Data

3.1 Data sources

Table 1 lists the data sources used in this study. I analyze employment data from the US Bureau of Labor Statistics available at the county level and at monthly intervals. I delimit the analysis to the years 2012–2019, during which there are no major external factors that would confound the analysis. The construction industry suffered a major downturn after the Great Recession, and the trends in employment began to stabilize by the end of 2012. The repeal of the Clean Water Rule in December 2019 marks the endpoint of the analysis. Although the federal and state policies to protect aquatic resources continued to evolve during 2020, the construction industry experienced another dramatic decline at the onset of the COVID-19 pandemic. The highly asymmetric effect of the pandemic would confound the estimation of policy parameters after March 2020.

To proxy compliance costs, I obtain information about wetland and stream compensatory credit transactions from the Regulatory In-lieu Fee and Bank Information Tracking System (RIBITS) administered by the US Army Corps of Engineers (Corps, 2022). In particular, I use the data in RIBITS to construct a county-level panel of compensatory credit availability. For each county and month, I observe the number of compensatory credits available for purchase and the acres of

Table 1: List of data sources

Data Source	Description	Years
US Bureau of Labor Statistics	Employment statistics, monthly by county	2012–2021
Corps (2022)	Transaction records for the Clean Water Act Section 404 compensatory mitigation program	1982–2022
EPA (2022)	Jurisdictional determinations that identify whether a water resource is regulated under the Clean Water Act	2016–2020
Corps (2020)	Permit decisions on impacts to water resources regulated under the Clean Water Act	2016–2020

water resources that these credits represent. The data contain detailed information on the credit providers, particularly if they are private mitigation banks or in-lieu fee programs.

I use two administrative data sources to ascertain which counties have a higher prevalence of intermittent and ephemeral streams and isolated wetlands, i.e. the water resources that became included within federal jurisdiction under the Clean Water Rule. Firstly, I use data on Approved Jurisdictional Determinations (JD) from the EPA. Jurisdictional determinations, initiated upon the request of a landowner, are official decisions on whether an aquatic resource belongs to federal jurisdiction under Section 404 of the Clean Water Act and whether impacts on the resource are thus subject to a permit requirement.[27] A jurisdictional determination is the first step in the Section 404 permit process, although not all landowners who request a determination proceed with a permit application.

Importantly, prior to the implementation of the Clean Water Rule, the jurisdictional determination data indicate whether an evaluated water resource was classified as isolated or non-permanent and thus non-jurisdictional. The Clean Water Rule targeted these particular resources to be included under federal jurisdiction. In other words, the prior determinations of isolated and non-permanent features are informative of the regions where the Rule would increase the requirements to obtain a federal permit (EPA and DA, 2021). Accordingly, the determinations

[27] 33 US Code 331.2.

should also be informative of the regions where firms faced increased regulatory costs.[28]

Secondly, I obtain similar information on the locations of affected water resources from the CWA Section 404 permit database, administered by the US Army Corps of Engineers (Corps, 2020). The permit data include a detailed account of the impacts on aquatic resources that required a Section 404 permit. However, the information on the legal classification of the water resource is not as precise as in the jurisdictional determination data. The permit data only indicate the ecological Cowardin classification code of the impacted water resource (Cowardin, 1979). For streams, the Cowardin codes for intermittent and ephemeral streams are available (codes R4 and R6). For wetlands, there is no separate class for isolated features. However, in the jurisdictional determination data, it is clear that palustrine emergent wetlands (code PEM) are most likely to be determined as isolated. Accordingly, I use impacts on aquatic features belonging to the Cowardin classes R4, R6, and PEM to indicate counties where the Clean Water Rule would likely have an effect.

3.2 Data description

Figure 3 depicts the mean residual construction employment after partialling out state-specific linear trends and month fixed effects.[29] Panel (a) in Figure 3 shows the residual employment trends in all counties by treatment group. In the treated counties, the mean residual employment drops after the implementation of the Rule, although a slight downward trend may be emerging already before the implementation date. No change in trend is visible in counties where the Rule was never implemented. Panel (b) shows the corresponding trend for counties in the treated states that litigated against the Rule (Michigan, Ohio, Oklahoma, and Tennessee). As explained in Section 2, these states made an unsuccessful attempt to prevent the implementation of the Clean Water Rule in their jurisdictions. The drop in employment during 2019 is more pronounced here than in the full sample of treated counties. Together, these two graphs suggest a negative employment

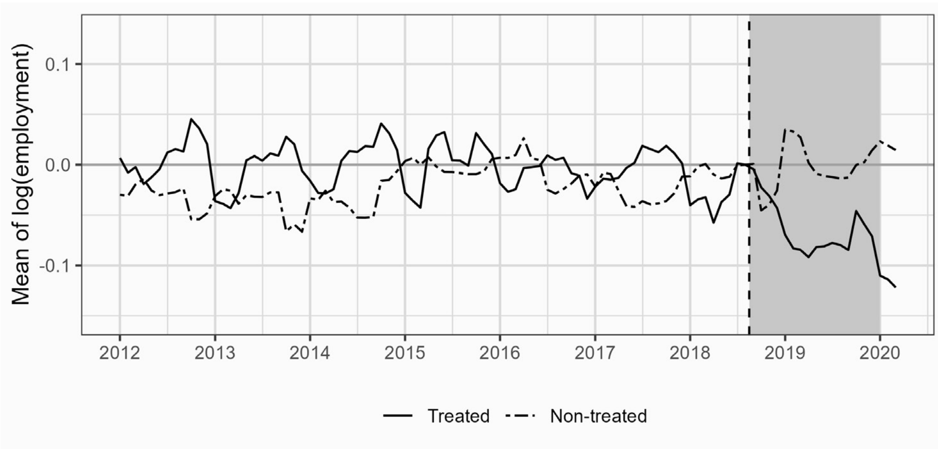
[28]Note that the changes in jurisdictional scope also affect landowners' propensity to request a determination. Thus, changes in the approved jurisdictional determinations themselves cannot be used to directly measure the economic effects of the Clean Water Rule. In addition to expanding the jurisdictional scope as described, the Rule also categorically excluded certain minor features outside federal jurisdiction. For these particular features, there was a stark increase in the number of requested jurisdictional determinations after the implementation of the Rule.

[29]Figure A.1 in the Appendix shows the raw data on construction employment. The downturn in 2018–2019 is visible also in the raw data. However, this downturn was relatively small in magnitude in comparison to seasonal fluctuations. I present the residualized employment data for clarity.

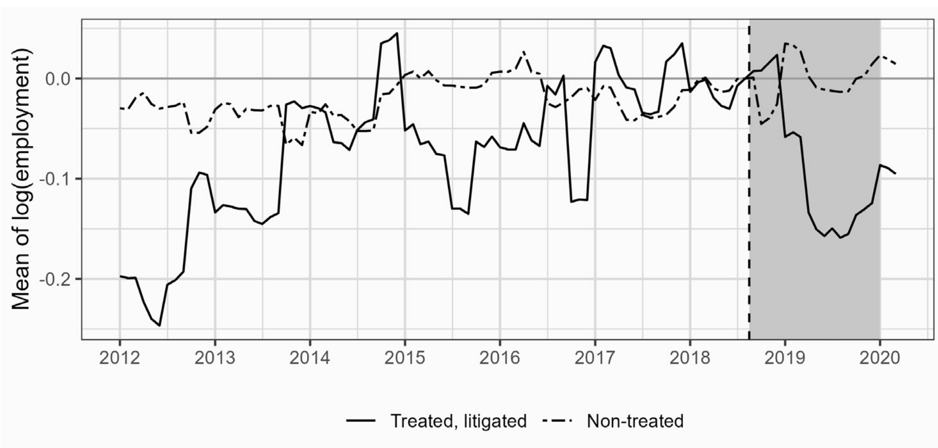
effect of the Clean Water Rule, and that the states that litigated against the Rule drive this effect.

Figure 3: Residualized construction employment

(a) All treated and non-treated states



(b) Treated states that litigated against the Clean Water Rule in the courts



Mean of (log) construction employment, residualized monthly county-level data. August 2018 = 0. Residuals from partialling out state-specific linear trends and month fixed effects.

Treated: Counties where the Clean Water Rule was implemented between August 2018 and December 2019 (shaded area).
Non-treated: Counties where the Clean Water Rule was never implemented.
Treated, litigated: Counties in states that unsuccessfully litigated against the Rule (MI, OH, OK, TN).

Table 2 presents descriptive statistics for three groups: all treated counties, treated counties in states that challenged the Clean Water Rule, and counties where the Rule was never implemented. Overall, the differences between the treated and non-treated counties reflect how the treated states include densely populated coastal regions. In the states that challenged the Clean Water Rule, two features related to compliance costs stand out. First, the average amount of available compensation credits that firms use to comply with their obligations under the Clean Water Act was markedly lower in comparison to the other treated states. (These statistics are measured in August 2018, before the entry into force of the Clean Water Rule.) Likewise, the share of counties that had access to in-lieu fee programs was considerably lower in these states. In contrast to private compensatory credit providers, in-lieu fee programs have the advantage that they may release compensatory credits in advance of the environmental improvement that ultimately awards the credits. Taken together, these factors indicate fewer opportunities for low-cost compliance.

Indicators of the prevalence of affected water resources point in the same direction, although here the contrasts are less stark. In treated states that challenged the Rule, there was a higher proportion of affected water resources in the jurisdictional determinations. Likewise, the Section 404 permit data indicate a higher proportion of affected waters out of the total impacts on all jurisdictional water resources.

Finally, in Section B in the Appendix, I present further description where I consider possible spillover effects, or leakage, from treated counties to non-treated counties. In other words, as construction is mobile, it is possible that firms reacted to the Clean Water Rule by moving their activities to a nearby location with less stringent environmental regulations. Although I find no conclusive evidence of leakage, certain patterns in the data suggest that it possibly took place. Accordingly, I also take possible spillovers into account in the regression analysis.

Table 2: Descriptive statistics

Variable	Treated All	Treated Litigated[a]	Non-treated
Construction employment			
Mean	3,061	1,475	1,500
Std. dev.	8,396	3,528	6,077
Total employment			
Mean	72,509	36,537	29,398
Std. dev.	221,823	92,544	106,682
Available compensation credits (acres)[b]			
Mean	1,249	299	1,157
Std. dev.	3,177	344	2,167
In-lieu fee program available (binary)[c]			
Mean	0.39	0.24	0.39
Std. dev.	0.45	0.38	0.46
Jurisdictional determinations on affected resources (proportion of total count)[d]			
Mean	0.25	0.30	0.23
Std. dev.	0.34	0.37	0.35
Permitted impacts on affected resources (proportion of total acres)[e]			
Mean	0.11	0.24	0.22
Std. dev.	0.27	0.37	0.42
Counties	1,043	260	2,065

County-level data over 2012–2019. Treated: Counties in 22 states and DC where the Clean Water Rule was implemented between 16 August 2018 and 23 December 2019.

[a] Counties in treated states that had litigated against the Clean Water Rule in regional courts (Michigan, Ohio, Oklahoma, and Tennessee). Michigan is included only in the employment statistics due to data availability.

[b] Compensation credits that permittees use to comply with their compensatory mitigation obligations under Section 404 of the Clean Water Act. Availability in August 2018.

[c] Indicator of whether a county is within the service area of an in-lieu fee program. Availability in August 2018.

[d] Approved Jurisdictional Determinations on isolated wetlands or intermittent and ephemeral streams, recorded between January 2015 and July 2018.

[e] Impacts on intermittent and ephemeral streams and palustrine emergent wetlands (Cowardin codes R4, R6, and PEM) that required a CWA Section 404 permit, recorded between January 2016 and July 2018.

Data sources: US Bureau of Labor Statistics; Corps (2020, 2022); EPA (2022).

4 Empirical strategy

I model the effect of the Clean Water Rule on construction employment using an event-study specification as follows:

$$y_{it} = \sum_{s \neq -1} \tau_s D_{its} + x_{it}\beta + \lambda(t \times \text{State}_i) + \alpha_i + \gamma_t + \varepsilon_{it} \tag{1}$$

where y_{it} is the log of employment in the construction sector in county i in month-year t, where D_{its} indicates if period t is s months after the Clean Water Rule was first implemented in county i, x_{it} is the log of employment in other sectors than construction, $t \times \text{State}_i$ is a state-specific linear time trend, α_i is a county fixed effect, and γ_t is a month fixed effect.

The identification of treatment effect parameters in equation (1) requires that, in the post-treatment periods, the counterfactual changes in the treated counties and the observed changes in the control counties are equal in expectation (*parallel trends*). In the raw employment data, the pre-treatment trends are not parallel between the treatment and control groups (see Figure A.1 in the Appendix). However, conditional on state-specific linear trends, the parallel trends assumption becomes plausible (see Figure 3).

Spillover effects from treated counties to non-treated counties potentially violate the stable unit treatment value assumption (*SUTVA*). As mentioned in Section 3.2 and discussed further in Section B in the Appendix, there are indications in the data that construction activity may have moved from counties where the Clean Water Rule was implemented to adjacent counties where the Rule was never implemented (see Figure B.1).

To account for possible spillovers to non-treated counties, I implement an approach suggested in Butts (2021) and Delgado and Florax (2015). This approach includes separate treatment dummies for the spillover group and consequently obtains an unbiased estimate of the policy parameters of interest (τ_s). The model becomes as follows:

$$y_{it} = \sum_{s \neq -1} \tau_s D_{its} + \sum_{s \neq -1} \delta_s \left(S_i \times \mathbb{I}(t - t_0 = s)\right) + x_{it}\beta + \lambda(t \times \text{State}_i) + \alpha_i + \gamma_t + \varepsilon_{it} \tag{2}$$

where S_i indicates if county i is a potential spillover county and $\mathbb{I}(t - t_0 = s)$ indicates if month-year t is s months after the Clean Water Rule was first implemented. In this model, the parameters δ_s are the dynamic spillover effects in the non-treated counties that S_i indicates.

The exposition in eq. (2) makes it clear that when spillovers exist ($\delta_s \neq 0$), omitting the spillover terms $S_i \times \mathbb{I}(t - t_0 = s)$ induces omitted variable bias in the estimator of τ_s. The non-treated units adjacent to treated units are also affected by the treatment. Using these non-treated units as controls would fail to identify the counterfactual trend for the treated units. This also highlights an important assumption: S_i must indicate all non-treated counties that are potentially affected by a spillover effect. If this assumption holds, the estimates of τ_s are unbiased estimates of the dynamic average treatment effects. I restrict the set of potential (non-treated) spillover counties to those counties that are directly adjacent to the treated counties. Considering activities such as suburban development, firms can relocate their projects only within a constrained distance. The inclusion of spillover indicators is further motivated in Section B in the Appendix.

5 Results

5.1 Main results

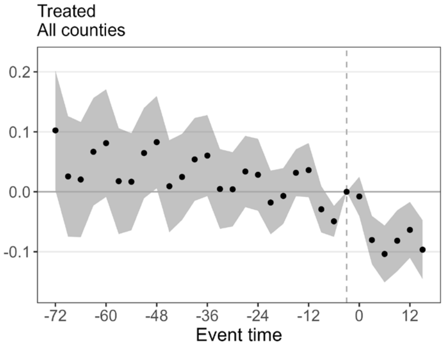
Figure 4 and Table 3 summarize the main estimation results. Panel (a) in Figure 4 reports the dynamic estimates over all treated counties. The average of the post-treatment estimates, reported in column (a) in Table 3, is -0.072 (95% CI: -0.115 to -0.005). However, the trend in the estimates is slightly decreasing prior to treatment. Due to this trend, it is ambiguous if the Clean Water Rule modified the trend in construction employment to any meaningful degree.

Panel (b) in Figure 4 reports estimates from a regression where the treatment effect parameters are estimated separately for counties in states that unsuccessfully litigated against the implementation of the Clean Water Rule. In these counties (left panel), a clear downturn appears after the implementation of the Rule (post-treatment average: -0.122; 95% CI: -0.212 to -0.033). The pre-treatment estimates are close to zero up to three and a half years before the treatment date, although there is greater variance before that, and this variance is comparable to the post-treatment decrease. There is no visible treatment effect in states that did not litigate against the Rule (right panel).

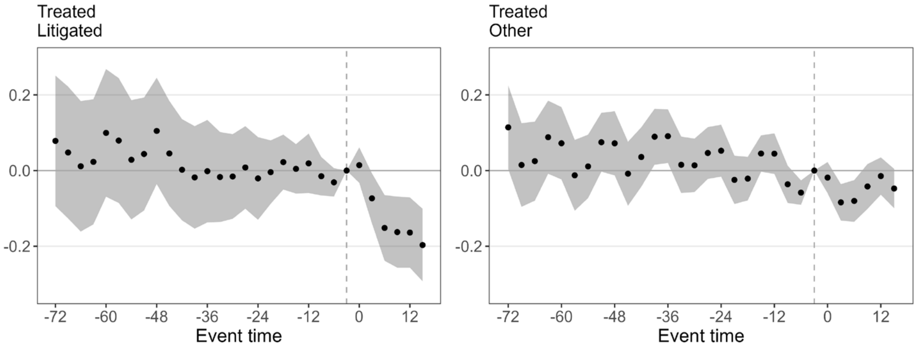
Turning to the estimated spillover effects, Table 3 indicates a positive although imprecise point estimate. However, the pre-treatment trend (reported in Figure B.2 in the Appendix) of these estimates is positive and drives the positive post-treatment estimates. It is still interesting to note that the inclusion of the spillover indicators slightly attenuates the estimated treatment effects (comparison of columns (a) and (b) against (c) and (d). This highlights the importance of accounting for possible spillover effects in policy evaluation.

Figure 4: Main estimation results

(a) All treated counties



(b) Estimates by subgroup



Outcome: Log of construction employment (NAICS 23).
Treatment: Implementation of the Clean Water Rule.

Estimates and 95% confidence intervals of dynamic treatment effects (τ_s in eq. (2)), aggregated to quarterly estimates.

Panel (a): Estimates for all treated counties.
Panel (b): Estimates separately for two subgroups: (i) Counties in treated states that unsuccessfully litigated against the implementation of the Clean Water Rule (MI, OH, OK, and TN); (ii) counties in the remaining treated states. See Figure 1b for an illustration of these groups.

Regression specifications as in eq. (2) controlling for: log of non-construction employment, spillover effects to non-treated counties, state-specific linear trends, month fixed effects, and county fixed effects.
Standard errors clustered by county (N = 3,108). Number of observations: 273,476.

Data source: US Bureau of Labor Statistics.

Table 3: Estimation results

	(a)	(b)	(c)	(d)
Treated × Post	−0.072		−0.077	
	(0.024)		(0.025)	
Treated × Post × Litigated		−0.122		−0.127
		(0.046)		(0.045)
Treated × Post × Other		−0.048		−0.052
		(0.027)		(0.027)
Spillover county × Post	0.062	0.062		
	(0.044)	(0.044)		
Log non-construction employment	−0.312	−0.313	−0.312	−0.313
	(0.180)	(0.180)	(0.180)	(0.180)
State × trend	✓	✓	✓	✓
Month FEs	✓	✓	✓	✓
N	273,476	273,476	273,476	273,476
N clusters	3,108	3,108	3,108	3,108

Outcome: Log of construction employment (NAICS 23).
Treatment: Implementation of the Clean Water Rule from August 2018 to December 2019.

Mean of dynamic treatment effect estimates after treatment, i.e. the mean over estimates of τ_s, $s \geq 0$ in eq. (2). For models (a) and (b), the full paths of dynamic estimates are reported in Figure 4. Standard errors clustered by county in parentheses.

County-level data at monthly intervals. Data source: US Bureau of Labor Statistics

In summary, the regression results show that the overall effect of the Clean Water Rule on construction employment was small or negligible in magnitude. However, there appears to be a clear negative effect on the construction industry in treated states that litigated against the implementation of the Rule. In Section 3.2 I described with administrative data how the compliance costs were likely higher in these states than in the remaining treated states that were advocating the implementation of the Rule. In the following section, I present a heterogeneity analysis to determine whether the estimates vary according to various indicators of compliance costs.

5.2 Heterogeneity

As discussed in Section 2.2, the majority of the compliance costs related to the Clean Water Rule arise from the obligation of permittees to compensate for adverse impacts on aquatic resources. In the majority of cases, permittees meet this obligation by purchasing credits that third-party entrepreneurial firms have generated from conservation activities. Importantly, a conservation firm must complete its conservation project before it can sell any compensation credits. Consequently, the compensation credit supply is inelastic in the short term. The only exception is publicly funded in-lieu fee programs that are allowed to release credits prior to the completion of a conservation project.

Figure 5 describes how compensation credit availability and supply inelasticity modified the effects of the Clean Water Rule. To this end, I employ two indicators that proxy for the supply constraint in the compensation credit market. The first indicator is the compensatory credit surplus relative to credit demand. In specific, I divide the amount of credits that are available for purchase in a county in August 2018 by the amount of credits that were purchased in the previous 12 months[30]. A second indicator is the availability of in-lieu free programs that are able to provide compensatory credits in advance of a conservation project, thus expediting the credit supply.[31]

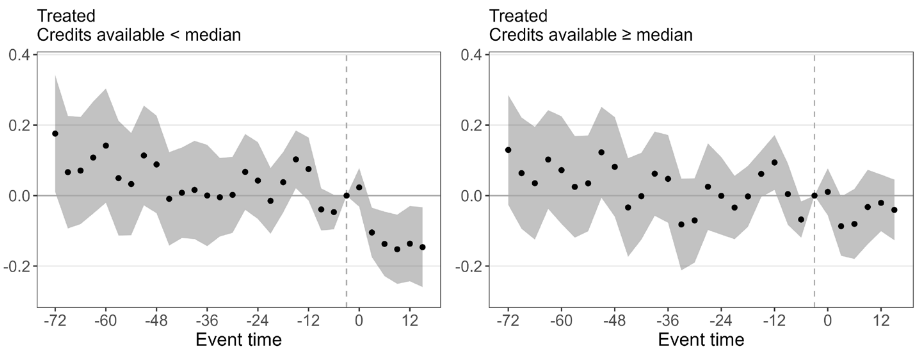
There are two sets of dynamics estimates plotted in Figure 5, each corresponding to one compliance cost indicator. Starting with panel (a), in counties where the amount of available compensatory credits was low, there is a downturn in construction employment after the Clean Water Rule was implemented (mean of

[30] In constructing the indicator, the credits are measured in acres of the compensated aquatic resource. This choice permits comparability of the records across regions, as the credit-per-acre ratios vary considerably across different districts of the US Army Corps of Engineers.

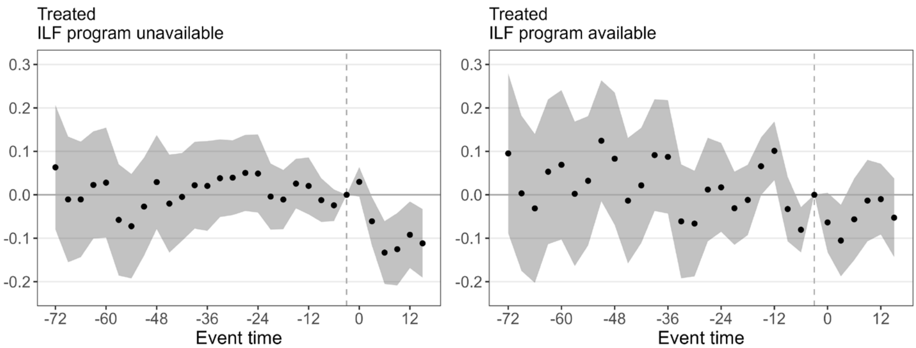
[31] Michigan is excluded from the regressions due to missing data on these indicators. Michigan has assumed the implementation of the Section 404 program, and it is not included in the federally administered datasets on compensatory credit transactions.

Figure 5: Estimates by compliance cost indicators

(a) Compensatory credit availability relative to demand when the Clean Water Rule was implemented
Subsample: Counties with compensatory credits ever available.



(b) In-lieu fee program availability when the Clean Water Rule was implemented
Subsample: Counties with compensatory credits ever available.



Outcome: Log of construction employment (NAICS 23).
Treatment: Implementation of the Clean Water Rule.

Estimates and 95% confidence intervals of dynamic treatment effects (τ_s in eq. (2)), aggregated to quarterly estimates. Estimates separately for subgroups of treated counties.

Panel (a): Subgroup division based on the availability of compensation credits in the Clean Water Act Section 404 compensatory mitigation program. Credits available = credit surplus in August 2018 divided by credits purchased in the twelve preceding months.
Panel (b): Subgroup division based on the availability of in-lieu fee programs in August 2018.

Regression specifications as in eq. (2) controlling for: log of non-construction employment, spillover effects to non-treated counties, state-specific linear trends, month fixed effects, and county fixed effects.

Standard errors clustered by county (N = 1,731). Number of observations: 152,328.

post-treatment estimates: -0.127; 95% CI: -0.202 to -0.052). Although seasonal variability persists in the estimates, the pre-treatment estimates show no trend in the preceding three years. In the counties with high credit availability, there is no visible effect. In panel (b), a similar pattern emerges. In counties where in-lieu fee programs were unavailable, there is a clear downturn in construction employment following the Clean Water Rule (mean of post-treatment estimates: -0.081; 95% CI: -0.157 to -0.005). The deviation from the pre-treatment estimates is clear. In counties where an ILF program was available, there is no visible response to the Clean Water Rule.

Figure 6 shows further heterogeneity analysis using indicators for the presence of water resources affected by the Clean Water Rule. There are no systematic findings from this analysis. Although the available data indicate that the treated states that litigated against the Rule have a higher concentration of affected water resources (see Table 2), this heterogeneity does not modify the estimated treatment effects in the full sample of treated counties.

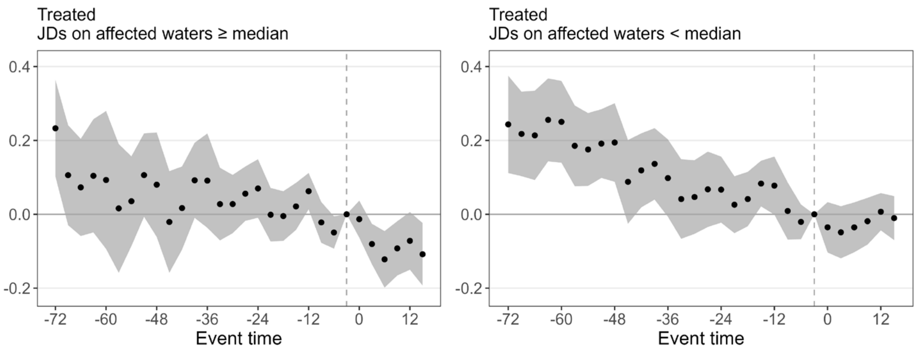
6 Discussion

In this paper, I examined how an expansion of environmental protections under the Clean Water Act affected construction employment in the US. The overall estimated effect of the increased protections on construction employment is negligible in magnitude. A negative effect becomes apparent in states that had litigated against the jurisdictional expansion in regional courts. In these states, the federal policy change represented an increase in stringency compared to state-level policies in these states. Further analysis suggests that limited or expensive compliance options characterize the counties where construction employment decreased after environmental regulations became more stringent.

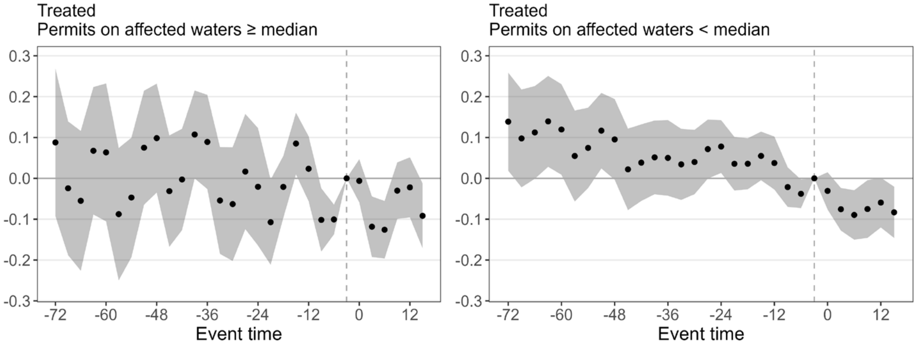
The estimated decrease in construction employment averages -0.122 (95% CI: -0.212 to -0.033) in log scale in the states that litigated against the Clean Water Rule. However, in the full sample of counties subject to the Rule, there is no discernible effect. A further qualification of the estimates is that they only represent short-run effects that occurred during the 16 months when the Rule was in force. In particular, in August 2018 when the new regulations were implemented, the Trump administration had already expressed its intent to revert the policy change. Since firms were aware of this intent, it is likely that the estimated negative effects only represent postponed construction projects rather than canceled projects. This would imply that any estimated negative employment effects are hardly informative of welfare effects. Unfortunately, the onset of the COVID-19

Figure 6: Estimates by indicators for affected water resources

(a) Approved jurisdictional determinations on affected water resources.
Subsample: Counties with JDs ever recorded (N = 1,569).



(b) Permitted impacts on affected water resources.
Subsample: Counties with CWA permits ever recorded (N = 1,700).



Outcome: Log of construction employment (NAICS 23).
Treatment: Implementation of the Clean Water Rule.

Estimates and 95% confidence intervals of dynamic treatment effects (τ_s in eq. (2)), aggregated to quarterly estimates. Estimates separately for subgroups of treated counties.

Panel (a): Subgroup division based on the share of approved jurisdictional determinations on affected water resources (intermittent and ephemeral streams, isolated wetlands) out of total determinations in a county. Number of observations: 138,044.
Panel (b): Subgroup division based on the share of permitted impacts, measured in acres, to affected water resources (intermittent and ephemeral streams, isolated wetlands) out of total permitted impacts in a county. Acres for stream impacts calculated as the linear extent of the impact multiplied by a 50 feet buffer. Number of observations: 149,600.

Regression specifications as in eq. (2) controlling for: log of non-construction employment, spillover effects to non-treated counties, state-specific linear trends, month fixed effects, and county fixed effects.

Standard errors clustered by county.

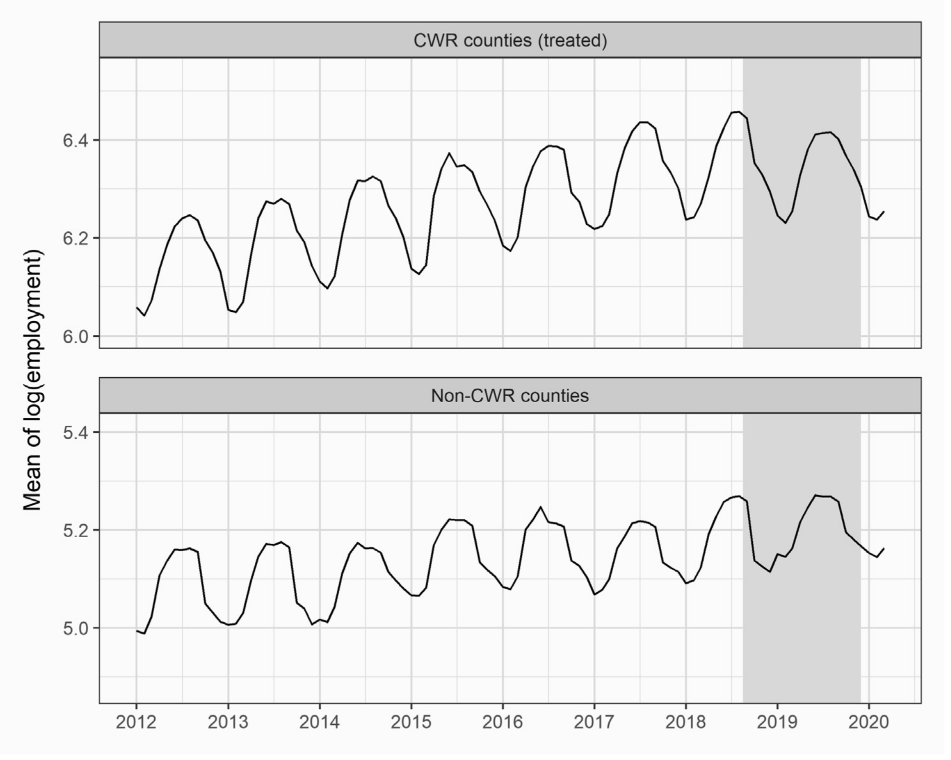
pandemic and subsequent changes in state and federal policies make it difficult to reliably evaluate the long-run effects of the policies in question. In any case, this analysis does not evaluate the overall welfare implications of the policy reform, as this evaluation would require quantification of benefits specifically from the aquatic resources that the reform targeted.

Notwithstanding its caveats, two main findings arise from the analysis. First, non-negligible effects arise in states that litigated against the Clean Water Rule. Second, the heterogeneity analysis suggests that the design and implementation of the compliance scheme modified the effects of the regulatory change. As discussed in Section 2, the majority of the costs for firms arise from compensatory mitigation, i.e. the obligation to compensate for adverse impacts on water resources by providing a substitute resource elsewhere. The majority of this compensation is mediated through markets where permittees buy compensation credits from third-party conservation firms. The effects of the jurisdictional expansion were pronounced in counties where few compensation credits were available. Moreover, these counties did not have any available public providers (in-lieu fee programs) that would have been able to supply credits in a short time frame after the policy change. This suggests that the inelastic supply in the compensatory credit market had implications for firms that would be subject to increased compensation requirements under the new regulations. In regions where private compensatory credit providers are few (e.g. due to insufficient demand), establishing public providers could resolve uncertainties regarding credit availability.

In conclusion, the analysis in this paper shows how design features in a market-based compliance scheme can modify the trade-off between economic activity and environmental protection. Further research on the topic, particularly an analysis of outcomes in the long term, is required to more comprehensively understand the welfare implications of aquatic resource protection under the Clean Water Act. Another specific challenge relates to understanding the geographic distribution of the targeted water resources, i.e. isolated wetlands and intermittent and ephemeral streams. Due to their very nature, mapping these resources is challenging and they are not sufficiently surveyed in available national inventories. Having this information available could provide a better understanding of their benefits as well as the impact of environmental protections on employment and the economy. Finally, the uncertainty surrounding the jurisdictional scope of the Clean Water Act as well as the interplay between state and federal regulations merit their own analyses.

A Supplementary figures

Figure A.1: Construction employment trends



CWR counties: Counties where the Clean Water Rule was implemented between August 2018 and December 2019 (shaded area).

The data depicted in this figure are the raw data underlying the residualized employment data in Figure 3.

B Spillovers to non-treated counties

In this section, I describe the patterns in the data that suggest a spillover effect or leakage where construction activity shifted from treated counties to adjacent non-treated counties. I also describe and implement an approach to account for such spillovers in estimation in order to identify the labor market impacts of the Clean Water Rule.

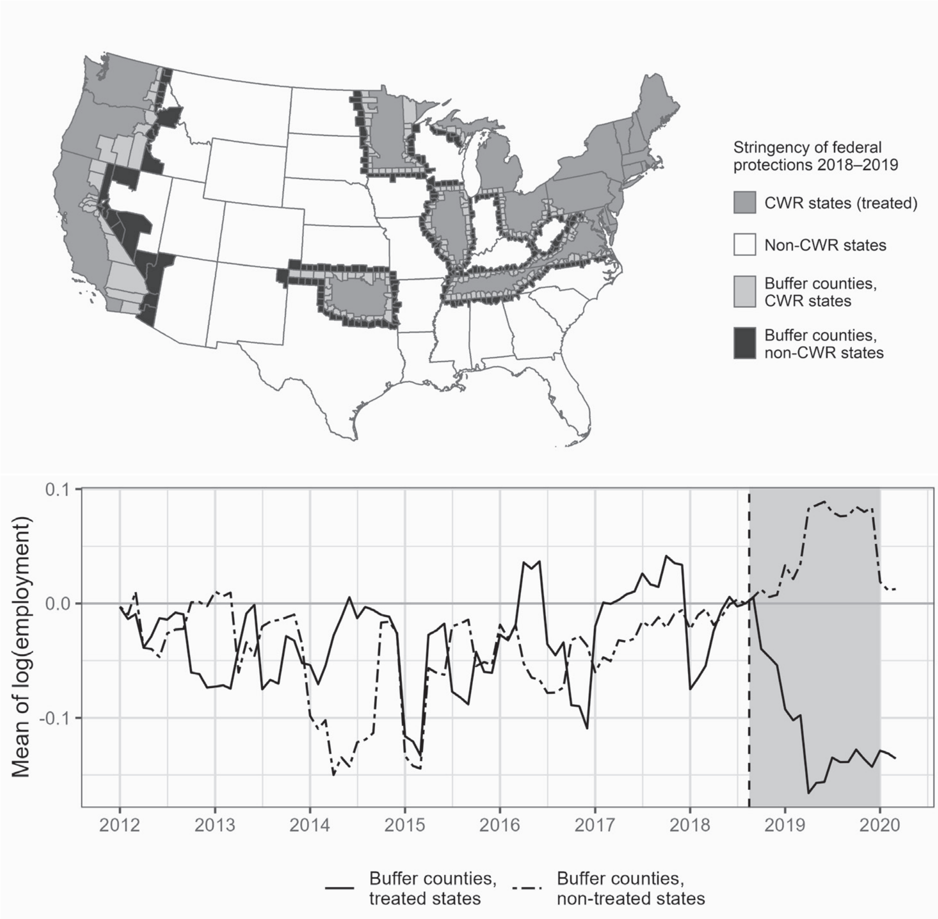
Considering the high compliance costs associated with the Clean Water Act, construction firms may have moved their activities to nearby counties where the geographic scope of federal protections did not expand. For example, in a metropolitan area that includes both treated and non-treated counties, suburban or industrial development may become more attractive in the non-treated counties. To account for such spillover effects, I analyze the outcomes in the treated and non-treated counties that are adjacent to one another, i.e. counties that share a border with a county that has a different treatment status (*buffer counties*).

Figure B.1 depicts the buffer counties and shows their trends of residualized construction employment. Prior to the implementation of the Clean Water Rule, the trends follow each other closely. However, after the implementation of the Rule, the trends diverge. There is a clear downturn in the treated counties, whereas the trend is increasing in the non-treated counties. Considering the increasing pre-treatment trend and the overall variation in the time series, it is ambiguous if this increase can be interpreted as a spillover effect. Nonetheless, given that the downturn is so pronounced in the treated buffer counties, the estimation strategy should account for the possibility of a spillover effect.

A spillover effect as explained above violates the stable unit treatment value assumption (SUTVA). As described in Section 4, I adopt an approach as suggested in Butts (2021) and Delgado and Florax (2015) where separate treatment indicators are included in the regression model for the spillover recipients in the non-treated units.

Figure B.2 reports the estimated spillover effects (estimates of δ_s in eq. (2)). Panel (a) shows the estimates for all buffer counties. Here, a pre-treatment trend clearly drives the positive aggregate coefficient reported in Table 3. Panel (b) shows the estimates for buffer counties adjacent to the treated states that litigated against the Clean Water Rule. Here, there is no pre-treatment trend immediately before the implementation of the Rule. However, noting the pre-treatment patterns at 24 months and earlier makes it ambiguous if the post-treatment estimates merely reflect the usual variation in the time series. In any case, despite their sign and pattern, the post-treatment estimates are imprecise.

Figure B.1: Residualized construction employment in buffer counties

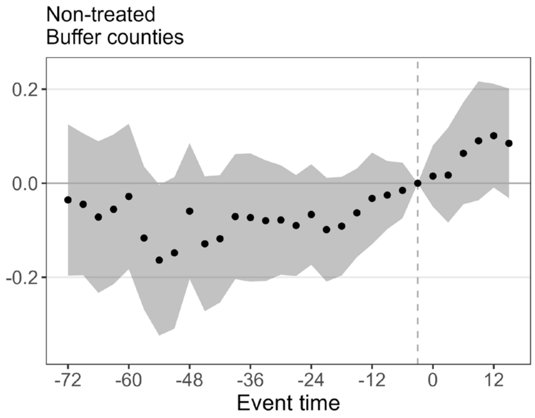


Mean of (log) construction employment, residualized monthly county-level data. August 2018 = 0. Residuals from partialling out month fixed effects and state-specific linear trends.

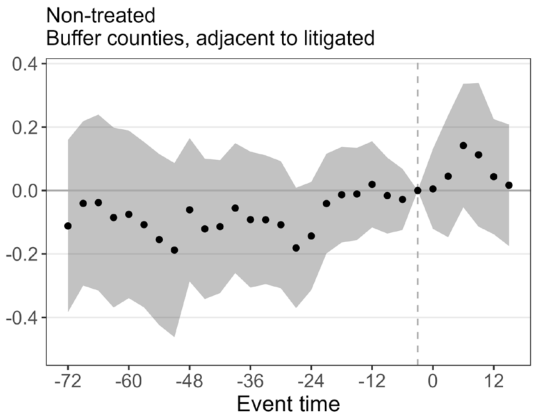
CWR states (treated): States where the Clean Water Rule was implemented between August 2018 and December 2019.
Buffer counties: Counties that are adjacent to a county that is in a different treatment group.

Figure B.2: Spillover effects

(a) All buffer counties



(b) Buffer counties adjacent to treated states that litigated against the Clean Water Rule



Outcome: Log of construction employment (NAICS 23).
Treatment: Implementation of the Clean Water Rule.

Estimates and 95% confidence intervals of dynamic spillover effects in non-treated counties (δ_s in eq. (2)), aggregated to quarterly estimates.

Panel (a): Estimates of spillover effects in model (a) as reported in Figure 4, panel (a) and Table 3, column (a). See Figure B.1 for illustration of non-treated buffer counties included in the spillover group. Panel (b): Estimates of spillover effects in counties adjacent to treated states that litigated against the Clean Water Rule.

For notes and full specification, see Figure 4 and Table 3.

Bibliography

BenDor, T., Brozovic, N., and Pallathucheril, V. G. (2007). Assessing the socioeconomic impacts of wetland mitigation in the Chicago region. Journal of the American Planning Association, 73(3):263.

BenDor, T. K., Guo, T., and Yates, A. J. (2014). Optimal advanced credit releases in ecosystem service markets. Environmental Management, 53:496–509.

BenDor, T. K. and Riggsbee, J. A. (2011). A survey of entrepreneurial risk in US wetland and stream compensatory mitigation markets. Environmental Science & Policy, 14(3):301–314.

Berman, E. and Bui, L. T. (2001). Environmental regulation and productivity: Evidence from oil refineries. Review of Economics and Statistics, 83(3):498–510.

Binmore, K., Rubinstein, A., and Wolinsky, A. (1986). The Nash bargaining solution in economic modelling. The RAND Journal of Economics, pages 176–188.

Bonds, M. H. and Pompe, J. J. (2003). Calculating wetland mitigation banking credits: adjusting for wetland function and location. Natural Resources Journal, pages 961–977.

Borusyak, K., Jaravel, X., and Spiess, J. (2021). Revisiting event study designs: Robust and efficient estimation. arXiv preprint arXiv:2108.12419.

Boyd, J. and Banzhaf, S. (2007). What are ecosystem services? The need for standardized environmental accounting units. Ecological Economics, 63(2-3):616–626.

Brondizio, E. S., Settele, J., Díaz, S., and Ngo, H. T. (2019). Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. IPBES secretariat, Bonn, Germany.

Bull, J. W. and Milner-Gulland, E.-J. (2020). Choosing prevention or cure when mitigating biodiversity loss: Trade-offs under 'no net loss' policies. Journal of Applied Ecology, 57(2):354–366.

Bull, J. W. and Strange, N. (2018). The global extent of biodiversity offset implementation under no net loss policies. Nature Sustainability, 1(12):790–798.

Butts, K. (2021). Difference-in-differences estimation with spatial spillovers. arXiv preprint arXiv:2105.03737.

Butts, K. and Gardner, J. (2021). did2s: Two-stage difference-in-differences. arXiv preprint arXiv:2109.05913.

Callaway, B. and Sant'Anna, P. H. (2021). Difference-in-differences with multiple time periods. Journal of Econometrics, 225(2):200–230.

Coggan, A., Buitelaar, E., Whitten, S., and Bennett, J. (2013). Factors that influence transaction costs in development offsets: Who bears what and why? Ecological Economics, 88:222–231.

Corps (2020). [US Army Corps of Engineers]. Operations and Maintenance Business Information Link Regulatory Module (ORM). (Obtained under the Freedom of Information Act from the US Army Corps of Engineers).

Corps (2022). [US Army Corps of Engineers]. Regulatory In-Lieu Fee and Bank Information Tracking System. `https://ribits.ops.usace.army.mil/` (Accessed: 2022-09-30).

Corps and EPA (1990). [US Army Corps of Engineers and the Environmental Protection Agency]. Memorandum of Agreement between the Environmental Protection Agency and the Department of the Army concerning the Determination of Mitigation under the Clean Water Act Section 404(b)(1) Guidelines.

Corps and EPA (1995). [US Army Corps of Engineers and the Environmental Protection Agency]. Federal Guidance for the Establishment, Use and Operation of Mitigation Banks. Federal Register, 60 Fed. Reg., 228:58605–58614.

Corps and EPA (2008). [US Army Corps of Engineers and the Environmental Protection Agency]. Compensatory Mitigation for Losses of Aquatic Resources; Final Rule. Federal Register, 73 Fed. Reg., 70:19593–19705.

Council, N. R. et al. (2001). Compensating for wetland losses under the Clean Water Act. National Academies Press, Washington D.C.

Cowardin, L. M. (1979). Classification of wetlands and deepwater habitats of the United States. Fish and Wildlife Service, US Department of the Interior.

Curtis, E. M. (2018). Who loses under cap-and-trade programs? The labor market effects of the NOx budget trading program. Review of Economics and Statistics, 100(1):151–166.

Curtis, E. M. (2020). Reevaluating the ozone nonattainment standards: Evidence from the 2004 expansion. Journal of Environmental Economics and Management, 99:102261.

Dahl, T. E. (1990). Wetlands losses in the United States, 1780s to 1980s. US Department of the Interior, Fish and Wildlife Service.

Dasgupta, P. (2021). The economics of biodiversity: The Dasgupta review. HM Treasury.

De Chaisemartin, C. and d'Haultfoeuille, X. (2020). Two-way fixed effects estimators with heterogeneous treatment effects. American Economic Review, 110(9):2964–96.

Delgado, M. S. and Florax, R. J. (2015). Difference-in-differences techniques for spatial data: Local autocorrelation and spatial interaction. Economics Letters, 137:123–126.

Deutz, A., Heal, G. M., Niu, R., Swanson, E., Townshend, T., Zhu, L., Delmar, A., Meghji, A., Sethi, S. A., and Tobin-de la Puente, J. (2020). Financing nature: Closing the global biodiversity financing gap. The Paulson Institute, The Nature Conservancy, and the Cornell Atkinson Center for Sustainability.

Doyle, M. W. and Yates, A. J. (2010). Stream ecosystem service markets under no-net-loss regulation. Ecological Economics, 69(4):820–827.

Drechsler, M. (2022). On the cost-effective temporal allocation of credits in conservation offsets when habitat restoration takes time and is uncertain. Environmental and Resource Economics, 82(2):437–459.

Drechsler, M. and Hartig, F. (2011). Conserving biodiversity with tradable permits under changing conservation costs and habitat restoration time lags. Ecological Economics, 70(3):533–541.

Drechsler, M. and Wätzold, F. (2009). Applying tradable permits to biodiversity conservation: Effects of space-dependent conservation benefits and cost heterogeneity on habitat allocation. Ecological Economics, 68(4):1083–1092.

Dwyer, J. L., Roy, D. P., Sauer, B., Jenkerson, C. B., Zhang, H. K., and Lymburner, L. (2018). Analysis ready data: Enabling analysis of the Landsat archive. Remote Sensing, 10(9):1363.

Environmental Law Institute (2013). State constraints: State imposed limitations on the authority of agencies to regulate waters beyond the scope of the federal Clean Water Act.

EPA (2022). [US Environmental Protection Agency]. Clean Water Act Approved Jurisdictional Determinations. `https://watersgeo.epa.gov/cwa/CWA-JDs/` (Accessed: 2022-09-30).

EPA and DA (2015). [US Environmental Protection Agency and the Department of the Army]. Economic Analysis of the EPA-Army Clean Water Rule.

EPA and DA (2020). [US Environmental Protection Agency and the Department of the Army]. Economic Analysis for the Navigable Waters Protection Rule: Definition of "Waters of the United States".

EPA and DA (2021). [US Environmental Protection Agency and the Department of the Army]. Economic Analysis for the Proposed "Revised Definition of 'Waters of the United States"' Rule.

Fernandez, L. and Karp, L. (1998). Restoring wetlands through wetlands mitigation banks. Environmental and Resource Economics, 12(3):323–344.

Ferris, A. E. and Frank, E. G. (2021). Labor market impacts of land protection: The Northern Spotted Owl. Journal of Environmental Economics and Management, 109:102480.

Ferris, A. E., Shadbegian, R. J., and Wolverton, A. (2014). The effect of environmental regulation on power sector employment: Phase I of the Title IV SO2 trading program. Journal of the Association of Environmental and Resource Economists, 1(4):521–553.

GAO (2005). [Government Accountability Office, United States]. wetlands Protection: Corps of Engineers does not have an effective oversight approach to ensure that compensatory mitigation is occurring: Report to the Ranking Democratic Member, Committee on Transportation and Infrastructure, House of Representatives. `https://www.gao.gov/products/gao-05-898` (Accessed: 2022-09-30).

Gardner, J. (2022). Two-stage differences in differences. arXiv preprint arXiv:2207.05943.

GIBOP (2019). Global Inventory of Biodiversity Offset Policies. https://portals.iucn.org/offsetpolicy/ (Accessed: 2022-09-30).

Goodman-Bacon, A. (2021). Difference-in-differences with variation in treatment timing. Journal of Econometrics, 225(2):254–277.

Greenstone, M. (2002). The impacts of environmental regulations on industrial activity: Evidence from the 1970 and 1977 Clean Air Act amendments and the census of manufactures. Journal of Political Economy, 110(6):1175–1219.

Griffin, R. K. and Dahl, T. E. (2016). Restoration outcomes and reporting: An assessment of wetland area gains in Wisconsin, USA. Ecological Restoration, 34(3):191–199.

Hafstead, M. A. and Williams III, R. C. (2018). Unemployment and environmental regulation in general equilibrium. Journal of Public Economics, 160:50–65.

Heimlich, R. E. (1998). Wetlands and agriculture: Private interests and public benefits. Number 765. US Department of Agriculture, Economic Research Service.

Holmes, T. P., Bergstrom, J. C., Huszar, E., Kask, S. B., and Orr III, F. (2004). Contingent valuation, net marginal benefits, and the scale of riparian ecosystem restoration. Ecological Economics, 49(1):19–30.

Institute for Water Resources (2015). The mitigation rule retrospective: A review of the 2008 regulations governing compensatory mitigation for losses of aquatic resources.

Johnson, C. N., Balmford, A., Brook, B. W., Buettel, J. C., Galetti, M., Guangchun, L., and Wilmshurst, J. M. (2017). Biodiversity losses and conservation responses in the anthropocene. Science, 356(6335):270–275.

Kangas, J. and Ollikainen, M. (2019). Economic insights in ecological compensations: Market analysis with an empirical application to the Finnish economy. Ecological Economics, 159:54–67.

Keiser, D. A. (2019). The missing benefits of clean water and the role of mismeasured pollution. Journal of the Association of Environmental and Resource Economists, 6(4):669–707.

Keiser, D. A., Olmstead, S. M., Boyle, K. J., Flatt, V. B., Keeler, B. L., Kling, C. L., Phaneuf, D. J., Shapiro, J. S., and Shimshack, J. P. (2021). A water rule that turns a blind eye to transboundary pollution. Science, 372(6539):241–243.

Keiser, D. A., Olmstead, S. M., Boyle, K. J., Flatt, V. B., Keeler, B. L., Phaneuf, D. J., Shapiro, J. S., and Shimshack, J. P. (2022). The Evolution of the "Waters of the United States" and the Role of Economics. Review of Environmental Economics and Policy, 16(1):000–000.

Keiser, D. A. and Shapiro, J. S. (2019). Consequences of the Clean Water Act and the demand for water quality. The Quarterly Journal of Economics, 134(1):349–396.

Koh, N. S., Hahn, T., and Boonstra, W. J. (2019). How much of a market is involved in a biodiversity offset? A typology of biodiversity offset policies. Journal of Environmental Management, 232:679–691.

Levrel, H., Scemama, P., and Vaissière, A.-C. (2017). Should we be wary of mitigation banking? Evidence regarding the risks associated with this wetland offset arrangement in Florida. Ecological Economics, 135:136–149.

Madsen, B. (2011). State of biodiversity markets report: Offset and compensation programs worldwide. `http://www.ecosystemmarketplace.com/documents/acrobat/sbdmr.pdf` (Accessed: 2022-09-30).

Mahdavi, S., Salehi, B., Granger, J., Amani, M., Brisco, B., and Huang, W. (2018). Remote sensing for wetland classification: A comprehensive review. GIScience & Remote Sensing, 55(5):623–658.

Malani, A. and Reif, J. (2015). Interpreting pre-trends as anticipation: Impact on estimated treatment effects from tort reform. Journal of Public Economics, 124:1–17.

Marsh, L. L., Porter, D. R., Salvesen, D. A., and McDonald, L. A. (1997). Mitigation Banking: Theory and Practice. In Natural Resources Forum, volume 21, pages 221–228. London: Butterworths.

Mitsch, W. J. and Gosselink, J. G. (2000). The value of wetlands: importance of scale and landscape setting. Ecological economics, 35(1):25–33.

Moene, K. O., Wallerstein, M., Hoel, M., et al. (1992). Bargaining structure and economic performance. Working paper series 218. Institute of Industrial Relations. University of California, Los Angeles.

Moreno-Mateos, D., Power, M. E., Comín, F. A., and Yockteng, R. (2012a). Structural and functional loss in restored wetland ecosystems. PLoS biology, 10(1):e1001247.

Moreno-Mateos, D., Power, M. E., Comín, F. A., and Yockteng, R. (2012b). Structural and functional loss in restored wetland ecosystems. PLoS Biology, 10(1):e1001247.

Nash, J. (1953). Two-person cooperative games. Econometrica: Journal of the Econometric Society, pages 128–140.

Needham, K., de Vries, F. P., Armsworth, P. R., and Hanley, N. (2019a). Designing markets for biodiversity offsets: Lessons from tradable pollution permits. Journal of Applied Ecology, 56(6):1429–1435.

Needham, K., de Vries, F. P., Armsworth, P. R., and Hanley, N. (2019b). Designing markets for biodiversity offsets: Lessons from tradable pollution permits. Journal of Applied Ecology, 56(6):1429–1435.

Nolte, C. (2020). High-resolution land value maps reveal underestimation of conservation costs in the United States. Proceedings of the National Academy of Sciences, 117(47):29577–29583.

OECD (2016). Biodiversity offsets: Effective Design and Implementation. OECD Publishing: Paris. `https://doi.org/10.1787/9789264222519-en` (Accessed: 2022-09-30).

OECD (2017). The Political Economy of Biodiversity Policy Reform. OECD Publishing: Paris. `https://doi.org/10.1787/9789264269545-en` (Accessed: 2022-09-30).

Rambachan, A. and Roth, J. (2019). An honest approach to parallel trends. Working paper, Harvard University.

Roth, J. and Sant'Anna, P. H. (2021). Efficient estimation for staggered rollout designs. arXiv preprint arXiv:2102.01291.

Ruhl, J. and Salzman, J. E. (2006). The effects of wetland mitigation banking on people. National Wetlands Newsletter, 28(2).

Sheriff, G., Ferris, A. E., and Shadbegian, R. J. (2019). How did air quality standards affect employment at US power plants? The importance of timing, geography, and stringency. Journal of the Association of Environmental and Resource Economists, 6(1):111–149.

Simpson, K., Hanley, N., Armsworth, P., de Vries, F., and Dallimer, M. (2021a). Incentivising biodiversity net gain with an offset market. Q Open, 1(1):qoab004.

Simpson, K., Hanley, N., and Bull, J. (2022). An analysis of the empirical evidence on markets for biodiversity offsets. Working paper.

Simpson, K. H., de Vries, F., Dallimer, M., Armsworth, P. R., and Hanley, N. (2021b). Understanding the performance of biodiversity offset markets: Evidence from an integrated ecological-economic model. Land Economics, 97(4):836–857.

Simpson, K. H., de Vries, F., Dallimer, M., Armsworth, P. R., and Hanley, N. (2021c). Understanding the performance of biodiversity offset markets: Evidence from an integrated ecological-economic model. Land Economics, 97(4):836–857.

Sonter, L. J., Barnes, M., Matthews, J. W., and Maron, M. (2019). Quantifying habitat losses and gains made by US Species Conservation Banks to improve compensation policies and avoid perverse outcomes. Conservation Letters, 12(3):e12629.

Stehman, S. and Wickham, J. (2020). A guide for evaluating and reporting map data quality: Affirming Shao et al."Overselling overall map accuracy misinforms about research reliability". Landscape Ecology, 35(6):1263–1267.

Stehman, S. V., Pengra, B. W., Horton, J. A., and Wellington, D. F. (2021). Validation of the US Geological Survey's Land Change Monitoring, Assessment and Projection (LCMAP) Collection 1.0 annual land cover products 1985–2017. Remote Sensing of Environment, 265:112646.

Sun, L. and Abraham, S. (2021). Estimating dynamic treatment effects in event studies with heterogeneous treatment effects. Journal of Econometrics, 225(2):175–199.

Taylor, C. A. and Druckenmiller, H. (2022). Wetlands, flooding, and the Clean Water Act. American Economic Review, 112(4):1334–63.

Theis, S. and Poesch, M. S. (2022). Assessing conservation and mitigation banking practices and associated gains and losses in the United States. Sustainability, 14(11):6652.

Thornton, M., Shrestha, R., Wei, Y., Thornton, P., Kao, S., and Wilson, B. (2020). Daymet: Annual Climate Summaries on a 1-km Grid for North America, Version 4. `https://doi.org/10.3334/ORNLDAAC/1852` (Accessed: 2022-09-30).

Tillman, S. C., Spyreas, G., Olnas, A., and Matthews, J. W. (2022). Plant communities in wetland mitigation banks surpass the quality of those in the most degraded, naturally occurring wetlands, but fall short of high-quality wetlands. Ecological Engineering, 176:106526.

Turner, R. K., Van Den Bergh, J. C., Söderqvist, T., Barendregt, A., Van Der Straaten, J., Maltby, E., and Van Ierland, E. C. (2000). Ecological-economic analysis of wetlands: Scientific integration for management and policy. Ecological Economics, 35(1):7–23.

US Environmental Protection Agency (2022). [US Environmental Protection Agency]. Potential Wetland Areas. `https://www.epa.gov/enviroatlas/enviroatlas-data` (Accessed: 2022-09-30).

USGS (2022). LCMAP collection 1.3 science products, Earth Resources Observation and Science (EROS) center. `https://doi.org/10.5066/P9W1TO6E` (Accessed: 2022-09-30).

Van den Bosch, K. and Matthews, J. W. (2017). An assessment of long-term compliance with performance standards in compensatory mitigation wetlands. Environmental Management, 59(4):546–556.

Walker, W. R. (2013). The transitional costs of sectoral reallocation: Evidence from the Clean Air Act and the workforce. The Quarterly Journal of Economics, 128(4):1787–1835.

Wickham, J., Stehman, S. V., Sorenson, D. G., Gass, L., and Dewitz, J. A. (2021). Thematic accuracy assessment of the NLCD 2016 land cover for the conterminous United States. Remote Sensing of Environment, 257:112357.

Winkler, K., Fuchs, R., Rounsevell, M., and Herold, M. (2021). Global land use changes are four times greater than previously estimated. Nature Communications, 12(1):2501.

Woodward, R. T. and Wui, Y.-S. (2001). The economic value of wetland services: A meta-analysis. Ecological Economics, 37(2):257–270.

Xian, G. Z., Smith, K., Wellington, D., Horton, J., Zhou, Q., Li, C., Auch, R., Brown, J. F., Zhu, Z., and Reker, R. R. (2022). Implementation of the CCDC algorithm to produce the LCMAP collection 1.0 annual land surface change product. Earth System Science Data, 14(1):143–162.

Yang, W., Chang, J., Xu, B., Peng, C., and Ge, Y. (2008). Ecosystem service value assessment for constructed wetlands: A case study in hangzhou, china. Ecological Economics, 68(1-2):116–125.

Zhu, Z. and Woodcock, C. E. (2014). Continuous change detection and classification of land cover using all available Landsat data. Remote Sensing of Environment, 144:152–171.

zu Ermgassen, S. O., Baker, J., Griffiths, R. A., Strange, N., Struebig, M. J., and Bull, J. W. (2019). The ecological outcomes of biodiversity offsets under "no net loss" policies: A global review. Conservation Letters, 12(6):e12664.

zu Ermgassen, S. O., Maron, M., Walker, C. M. C., Gordon, A., Simmonds, J. S., Strange, N., Robertson, M., and Bull, J. W. (2020). The hidden biodiversity risks of increasing flexibility in biodiversity offset trades. Biological Conservation, 252:108861.

Previous doctoral theses in the Department of Economics, Gothenburg

Avhandlingar publicerade innan serien Ekonomiska Studier startades
(Theses published before the series Ekonomiska Studier was started):

Östman, Hugo (1911), Norrlands ekonomiska utveckling
Moritz, Marcus (1911), Den svenska tobaksindustrien
Sundbom, I. (1933), Prisbildning och ändamålsenlighet
Gerhard, I. (1948), Problem rörande Sveriges utrikeshandel 1936/38
Hegeland, Hugo (1951), The Quantity Theory of Money
Mattsson, Bengt (1970), Cost-Benefit analys
Rosengren, Björn (1975), Valutareglering och nationell ekonomisk politik
Hjalmarsson, Lennart (1975), Studies in a Dynamic Theory of Production and its Applications
Örtendahl, Per-Anders (1975), Substitutionsaspekter på produktionsprocessen vid massaframställning
Anderson, Arne M. (1976), Produktion, kapacitet och kostnader vid ett helautomatiskt emballageglasbruk
Ohlsson, Olle (1976), Substitution och odelbarheter i produktionsprocessen vid massaframställning
Gunnarsson, Jan (1976), Produktionssystem och tätortshierarki – om sambandet mellan rumslig och ekonomisk struktur
Köstner, Evert (1976), Optimal allokering av tid mellan utbildning och arbete
Wigren, Rune (1976), Analys av regionala effektivitetsskillnader inom industribranscher
Wästlund, Jan (1976), Skattning och analys av regionala effektivitetsskillnader inom industribranscher
Flöjstad, Gunnar (1976), Studies in Distortions, Trade and Allocation Problems
Sandelin, Bo (1977), Prisutveckling och kapitalvinster på bostadsfastigheter
Dahlberg, Lars (1977), Empirical Studies in Public Planning
Lönnroth, Johan (1977), Marxism som matematisk ekonomi
Johansson, Börje (1978), Contributions to Sequential Analysis of Oligopolistic Competition

<u>Ekonomiska Studier</u>, utgivna av Nationalekonomiska institutionen vid Göteborgs Universitet. Nr 1 och 4 var inte doktorsavhandlingar. (The contributions to the department series ’Ekonomiska Studier’ where no. 1 and 4 were no doctoral theses):

2. **Ambjörn, Erik** (1959), Svenskt importberoende 1926-1956: en ekonomisk-statistisk kartläggning med kommentarer
3. **Landgren, K-G.** (1960), Den ”Nya ekonomien” i Sverige: J.M. Keynes, E. Wigfors och utecklingen 1927-39
5. **Bigsten, Arne** (1979), Regional Inequality and Development: A Case Study of Kenya
6. **Andersson, Lars** (1979), Statens styrning av de kommunala budgetarnas struktur (Central Government Influence on the Structure of the Municipal Budget)
7. **Gustafsson, Björn** (1979), Inkomst- och uppväxtförhållanden (Income and Family Background)
8. **Granholm, Arne** (1981), Interregional Planning Models for the Allocation of Private and Public Investments
9. **Lundborg, Per** (1982), Trade Policy and Development: Income Distributional Effects in the Less Developed Countries of the US and EEC Policies for Agricultural

Commodities

10. **Juås, Birgitta** (1982), Värdering av risken för personskador. En jämförande studie av implicita och explicita värden. (Valuation of Personal Injuries. A comparison of Explicit and Implicit Values)
11. **Bergendahl, Per-Anders** (1982), Energi och ekonomi - tillämpningar av input-output analys (Energy and the Economy - Applications of Input-Output Analysis)
12. **Blomström, Magnus** (1983), Foreign Investment, Technical Efficiency and Structural Change - Evidence from the Mexican Manufacturing Industry
13. **Larsson, Lars-Göran** (1983), Comparative Statics on the Basis of Optimization Methods
14. **Persson, Håkan** (1983), Theory and Applications of Multisectoral Growth Models
15. **Sterner, Thomas** (1986), Energy Use in Mexican Industry.
16. **Flood, Lennart** (1986), On the Application of Time Use and Expenditure Allocation Models.
17. **Schuller, Bernd-Joachim** (1986), Ekonomi och kriminalitet - en empirisk undersökning av brottsligheten i Sverige (Economics of crime - an empirical analysis of crime in Sweden)
18. **Walfridson, Bo** (1987), Dynamic Models of Factor Demand. An Application to Swedish Industry.
19. **Stålhammar, Nils-Olov** (1987), Strukturomvandling, företagsbeteende och förväntningsbildning inom den svenska tillverkningsindustrin (Structural Change, Firm Behaviour and Expectation Formation in Swedish Manufactury)
20. **Anxo, Dominique** (1988), Sysselsättningseffekter av en allmän arbetstidsförkortning (Employment effects of a general shortage of the working time)
21. **Mbelle, Ammon** (1988), Foreign Exchange and Industrial Development: A Study of Tanzania.
22. **Ongaro, Wilfred** (1988), Adoption of New Farming Technology: A Case Study of Maize Production in Western Kenya.
23. **Zejan, Mario** (1988), Studies in the Behavior of Swedish Multinationals.
24. **Görling, Anders** (1988), Ekonomisk tillväxt och miljö. Förorenings-struktur och ekonomiska effekter av olika miljövårdsprogram. (Economic Growth and Environment. Pollution Structure and Economic Effects of Some Environmental Programs).
25. **Aguilar, Renato** (1988), Efficiency in Production: Theory and an Application on Kenyan Smallholders.
26. **Kayizzi-Mugerwa, Steve** (1988), External Shocks and Adjustment in Zambia.
27. **Bornmalm-Jardelöw, Gunilla** (1988), Högre utbildning och arbetsmarknad (Higher Education and the Labour Market)
28. **Tansini, Ruben** (1989), Technology Transfer: Dairy Industries in Sweden and Uruguay.
29. **Andersson, Irene** (1989), Familjebeskattning, konsumtion och arbetsutbud - En ekonometrisk analys av löne- och inkomstelasticiteter samt policysimuleringar för svenska hushåll (Family Taxation, Consumption and Labour Supply - An Econometric Analysis of Wage and Income Elasticities and Policy Simulations for Swedish Households)
30. **Henrekson, Magnus** (1990), An Economic Analysis of Swedish Government Expenditure
31. **Sjöö, Boo** (1990), Monetary Policy in a Continuous Time Dynamic Model for Sweden

32. **Rosén, Åsa** (1991), Contributions to the Theory of Labour Contracts.
33. **Loureiro, Joao M. de Matos** (1992), Foreign Exchange Intervention, Sterilization and Credibility in the EMS: An Empirical Study
34. **Irandoust, Manuchehr** (1993), Essays on the Behavior and Performance of the Car Industry
35. **Tasiran, Ali Cevat** (1993), Wage and Income Effects on the Timing and Spacing of Births in Sweden and the United States
36. **Milopoulos, Christos** (1993), Investment Behaviour under Uncertainty: An Econometric Analysis of Swedish Panel Data
37. **Andersson, Per-Åke** (1993), Labour Market Structure in a Controlled Economy: The Case of Zambia
38. **Storrie, Donald W.** (1993), The Anatomy of a Large Swedish Plant Closure
39. **Semboja, Haji Hatibu Haji** (1993), Energy and Development in Kenya
40. **Makonnen, Negatu** (1993), Labor Supply and the Distribution of Economic Well-Being: A Case Study of Lesotho
41. **Julin, Eva** (1993), Structural Change in Rural Kenya
42. **Durevall, Dick** (1993), Essays on Chronic Inflation: The Brazilian Experience
43. **Veiderpass, Ann** (1993), Swedish Retail Electricity Distribution: A Non-Parametric Approach to Efficiency and Productivity Change
44. **Odeck, James** (1993), Measuring Productivity Growth and Efficiency with Data Envelopment Analysis: An Application on the Norwegian Road Sector
45. **Mwenda, Abraham** (1993), Credit Rationing and Investment Behaviour under Market Imperfections: Evidence from Commercial Agriculture in Zambia
46. **Mlambo, Kupukile** (1993), Total Factor Productivity Growth: An Empirical Analysis of Zimbabwe's Manufacturing Sector Based on Factor Demand Modelling
47. **Ndung'u, Njuguna** (1993), Dynamics of the Inflationary Process in Kenya
48. **Modén, Karl-Markus** (1993), Tax Incentives of Corporate Mergers and Foreign Direct Investments
49. **Franzén, Mikael** (1994), Gasoline Demand - A Comparison of Models
50. **Heshmati, Almas** (1994), Estimating Technical Efficiency, Productivity Growth And Selectivity Bias Using Rotating Panel Data: An Application to Swedish Agriculture
51. **Salas, Osvaldo** (1994), Efficiency and Productivity Change: A Micro Data Case Study of the Colombian Cement Industry
52. **Bjurek, Hans** (1994), Essays on Efficiency and Productivity Change with Applications to Public Service Production
53. **Cabezas Vega, Luis** (1994), Factor Substitution, Capacity Utilization and Total Factor Productivity Growth in the Peruvian Manufacturing Industry
54. **Katz, Katarina** (1994), Gender Differentiation and Discrimination. A Study of Soviet Wages
55. **Asal, Maher** (1995), Real Exchange Rate Determination and the Adjustment Process: An Empirical Study in the Cases of Sweden and Egypt
56. **Kjulin, Urban** (1995), Economic Perspectives on Child Care
57. **Andersson, Göran** (1995), Volatility Forecasting and Efficiency of the Swedish Call Options Market
58. **Forteza, Alvaro** (1996), Credibility, Inflation and Incentive Distortions in the Welfare State
59. **Locking, Håkan** (1996), Essays on Swedish Wage Formation
60. **Välilä, Timo** (1996), Essays on the Credibility of Central Bank Independence

61. **Yilma, Mulugeta** (1996), Measuring Smallholder Efficiency: Ugandan Coffee and Food-Crop Production
62. **Mabugu, Ramos E.** (1996), Tax Policy Analysis in Zimbabwe Applying General Equilibrium Models
63. **Johansson, Olof** (1996), Welfare, Externalities, and Taxation; Theory and Some Road Transport Applications.
64. **Chitiga, Margaret** (1996), Computable General Equilibrium Analysis of Income Distribution Policies in Zimbabwe
65. **Leander, Per** (1996), Foreign Exchange Market Behavior Expectations and Chaos
66. **Hansen, Jörgen** (1997), Essays on Earnings and Labor Supply
67. **Cotfas, Mihai** (1997), Essays on Productivity and Efficiency in the Romanian Cement Industry
68. **Horgby, Per-Johan** (1997), Essays on Sharing, Management and Evaluation of Health Risks
69. **Nafar, Nosratollah** (1997), Efficiency and Productivity in Iranian Manufacturing Industries
70. **Zheng, Jinghai** (1997), Essays on Industrial Structure, Technical Change, Employment Adjustment, and Technical Efficiency
71. **Isaksson, Anders** (1997), Essays on Financial Liberalisation in Developing Countries: Capital mobility, price stability, and savings
72. **Gerdin, Anders** (1997), On Productivity and Growth in Kenya, 1964-94
73. **Sharifi, Alimorad** (1998), The Electricity Supply Industry in Iran: Organization, performance and future development
74. **Zamanian, Max** (1997), Methods for Mutual Fund Portfolio Evaluation: An application to the Swedish market
75. **Manda, Damiano Kulundu** (1997), Labour Supply, Returns to Education, and the Effect of Firm Size on Wages: The case of Kenya
76. **Holmén, Martin** (1998), Essays on Corporate Acquisitions and Stock Market Introductions
77. **Pan, Kelvin** (1998), Essays on Enforcement in Money and Banking
78. **Rogat, Jorge** (1998), The Value of Improved Air Quality in Santiago de Chile
79. **Peterson, Stefan** (1998), Essays on Large Shareholders and Corporate Control
80. **Belhaj, Mohammed** (1998), Energy, Transportation and Urban Environment in Africa: The Case of Rabat-Salé, Morocco
81. **Mekonnen, Alemu** (1998), Rural Energy and Afforestation: Case Studies from Ethiopia
82. **Johansson, Anders** (1998), Empirical Essays on Financial and Real Investment Behavior
83. **Köhlin, Gunnar** (1998), The Value of Social Forestry in Orissa, India
84. **Levin, Jörgen** (1998), Structural Adjustment and Poverty: The Case of Kenya
85. **Ncube, Mkhululi** (1998), Analysis of Employment Behaviour in Zimbabwe
86. **Mwansa, Ladslous** (1998), Determinants of Inflation in Zambia
87. **Agnarsson, Sveinn** (1998), Of Men and Machines: Essays in Applied Labour and Production Economics
88. **Kadenge, Phineas** (1998), Essays on Macroeconomic Adjustment in Zimbabwe: Inflation, Money Demand, and the Real Exchange Rate
89. **Nyman, Håkan** (1998), An Economic Analysis of Lone Motherhood in Sweden
90. **Carlsson, Fredrik** (1999), Essays on Externalities and Transport
91. **Johansson, Mats** (1999), Empirical Studies of Income Distribution

92. **Alemu, Tekie** (1999), Land Tenure and Soil Conservation: Evidence from Ethiopia
93. **Lundvall, Karl** (1999), Essays on Manufacturing Production in a Developing Economy: Kenya 1992-94
94. **Zhang, Jianhua** (1999), Essays on Emerging Market Finance
95. **Mlima, Aziz Ponary** (1999), Four Essays on Efficiency and Productivity in Swedish Banking
96. **Davidsen, Björn-Ivar** (2000), Bidrag til den økonomisk-metodologiske tenkningen (Contributions to the Economic Methodological Thinking)
97. **Ericson, Peter** (2000), Essays on Labor Supply
98. **Söderbom, Måns** (2000), Investment in African Manufacturing: A Microeconomic Analysis
99. **Höglund, Lena** (2000), Essays on Environmental Regulation with Applications to Sweden
100. **Olsson, Ola** (2000), Perspectives on Knowledge and Growth
101. **Meuller, Lars** (2000), Essays on Money and Credit
102. **Österberg, Torun** (2000), Economic Perspectives on Immigrants and Intergenerational Transmissions
103. **Kalinda Mkenda, Beatrice** (2001), Essays on Purchasing Power Parity, RealExchange Rate, and Optimum Currency Areas
104. **Nerhagen, Lena** (2001), Travel Demand and Value of Time - Towards an Understanding of Individuals Choice Behavior
105. **Mkenda, Adolf** (2001), Fishery Resources and Welfare in Rural Zanzibar
106. **Eggert, Håkan** (2001), Essays on Fisheries Economics
107. **Andrén, Daniela** (2001), Work, Sickness, Earnings, and Early Exits from the Labor Market. An Empirical Analysis Using Swedish Longitudinal Data
108. **Nivorozhkin, Eugene** (2001), Essays on Capital Structure
109. **Hammar, Henrik** (2001), Essays on Policy Instruments: Applications to Smoking and the Environment
110. **Nannyonjo, Justine** (2002), Financial Sector Reforms in Uganda (1990-2000): Interest Rate Spreads, Market Structure, Bank Performance and Monetary Policy
111. **Wu, Hong** (2002), Essays on Insurance Economics
112. **Linde-Rahr, Martin** (2002), Household Economics of Agriculture and Forestry in Rural Vienam
113. **Maneschiöld, Per-Ola** (2002), Essays on Exchange Rates and Central Bank Credibility
114. **Andrén, Thomas** (2002), Essays on Training, Welfare and Labor Supply
115. **Granér, Mats** (2002), Essays on Trade and Productivity: Case Studies of Manufacturing in Chile and Kenya
116. **Jaldell, Henrik** (2002), Essays on the Performance of Fire and Rescue Services
117. **Alpizar, Francisco, R.** (2002), Essays on Environmental Policy-Making in Developing Countries: Applications to Costa Rica
118. **Wahlberg, Roger** (2002), Essays on Discrimination, Welfare and Labor Supply
119. **Piculescu, Violeta** (2002), Studies on the Post-Communist Transition
120. **Pylkkänen, Elina** (2003), Studies on Household Labor Supply and Home Production
121. **Löfgren, Åsa** (2003), Environmental Taxation – Empirical and Theoretical Applications
122. **Ivaschenko, Oleksiy** (2003), Essays on Poverty, Income Inequality and Health in Transition Economies

123. **Lundström, Susanna** (2003), On Institutions, Economic Growth and the Environment
124. **Wambugu, Anthony** (2003), Essays on Earnings and Human Capital in Kenya
125. **Adler, Johan** (2003), Aspects of Macroeconomic Saving
126. **Erlandsson, Mattias** (2003), On Monetary Integration and Macroeconomic Policy
127. **Brink, Anna** (2003), On the Political Economy of Municipality Break-Ups
128. **Ljungwall, Christer** (2003), Essays on China's Economic Performance During the Reform Period
129. **Chifamba, Ronald** (2003), Analysis of Mining Investments in Zimbabwe
130. **Muchapondwa, Edwin** (2003), The Economics of Community-Based Wildlife Conservation in Zimbabwe
131. **Hammes, Klaus** (2003), Essays on Capital Structure and Trade Financing
132. **Abou-Ali, Hala** (2003), Water and Health in Egypt: An Empirical Analysis
133. **Simatele, Munacinga** (2004), Financial Sector Reforms and Monetary Policy in Zambia
134. **Tezic, Kerem** (2004), Essays on Immigrants' Economic Integration
135. INSTÄLLD
136. **Gjirja, Matilda** (2004), Efficiency and Productivity in Swedish Banking
137. **Andersson, Jessica** (2004), Welfare Environment and Tourism in Developing Countries
138. **Chen, Yinghong** (2004), Essays on Voting Power, Corporate Governance and Capital Structure
139. **Yesuf, Mahmud** (2004), Risk, Time and Land Management under Market Imperfections: Applications to Ethiopia
140. **Kateregga, Eseza** (2005), Essays on the Infestation of Lake Victoria by the Water Hyacinth
141. **Edvardsen, Dag Fjeld** (2004), Four Essays on the Measurement of Productive Efficiency
142. **Lidén, Erik** (2005), Essays on Information and Conflicts of Interest in Stock Recommendations
143. **Dieden, Sten** (2005), Income Generation in the African and Coloured Population – Three Essays on the Origins of Household Incomes in South Africa
144. **Eliasson, Marcus** (2005), Individual and Family Consequences of Involuntary Job Loss
145. **Mahmud, Minhaj** (2005), Measuring Trust and the Value of Statistical Lives: Evidence from Bangladesh
146. **Lokina, Razack Bakari** (2005), Efficiency, Risk and Regulation Compliance: Applications to Lake Victoria Fisheries in Tanzania
147. **Jussila Hammes, Johanna** (2005), Essays on the Political Economy of Land Use Change
148. **Nyangena, Wilfred** (2006), Essays on Soil Conservation, Social Capital and Technology Adoption
149. **Nivorozhkin, Anton** (2006), Essays on Unemployment Duration and Programme Evaluation
150. **Sandén, Klas** (2006), Essays on the Skill Premium
151. **Deng, Daniel** (2006), Three Essays on Electricity Spot and Financial Derivative Prices at the Nordic Power Exchange
152. **Gebreeyesus, Mulu** (2006), Essays on Firm Turnover, Growth, and Investment Behavior in Ethiopian Manufacturing

153. **Islam, Nizamul Md.** (2006), Essays on Labor Supply and Poverty: A Microeconometric Application
154. **Kjaer, Mats** (2006), Pricing of Some Path-Dependent Options on Equities and Commodities
155. **Shimeles, Abebe** (2006), Essays on Poverty, Risk and Consumption Dynamics in Ethiopia
156. **Larsson, Jan** (2006), Four Essays on Technology, Productivity and Environment
157. **Congdon Fors, Heather** (2006), Essays in Institutional and Development Economics
158. **Akpalu, Wisdom** (2006), Essays on Economics of Natural Resource Management and Experiments
159. **Daruvala, Dinky** (2006), Experimental Studies on Risk, Inequality and Relative Standing
160. **García, Jorge** (2007), Essays on Asymmetric Information and Environmental Regulation through Disclosure
161. **Bezabih, Mintewab** (2007), Essays on Land Lease Markets, Productivity, Biodiversity, and Environmental Variability
162. **Visser, Martine** (2007), Fairness, Reciprocity and Inequality: Experimental Evidence from South Africa
163. **Holm, Louise** (2007), A Non-Stationary Perspective on the European and Swedish Business Cycle
164. **Herbertsson, Alexander** (2007), Pricing Portfolio Credit Derivatives
165. **Johansson, Anders C.** (2007), Essays in Empirical Finance: Volatility, Interdependencies, and Risk in Emerging Markets
166. **Ibáñez Díaz, Marcela** (2007), Social Dilemmas: The Role of Incentives, Norms and Institutions
167. **Ekbom, Anders** (2007), Economic Analysis of Soil Capital, Land Use and Agricultural Production in Kenya
168. **Sjöberg, Pål** (2007), Essays on Performance and Growth in Swedish Banking
169. **Palma Aguirre, Grisha Alexis** (2008), Explaining Earnings and Income Inequality in Chile
170. **Akay, Alpaslan** (2008), Essays on Microeconometrics and Immigrant Assimilation
171. **Carlsson, Evert** (2008), After Work – Investing for Retirement
172. **Munshi, Farzana** (2008), Essays on Globalization and Occupational Wages
173. **Tsakas, Elias** (2008), Essays on Epistemology and Evolutionary Game Theory
174. **Erlandzon, Karl** (2008), Retirement Planning: Portfolio Choice for Long-Term Investors
175. **Lampi, Elina** (2008), Individual Preferences, Choices, and Risk Perceptions – Survey Based Evidence
176. **Mitrut, Andreea** (2008), Four Essays on Interhousehold Transfers and Institutions in Post-Communist Romania
177. **Hansson, Gustav** (2008), Essays on Social Distance, Institutions, and Economic Growth
178. **Zikhali, Precious** (2008), Land Reform, Trust and Natural Resource Management in Africa
179. **Tengstam, Sven** (2008), Essays on Smallholder Diversification, Industry Location, Debt Relief, and Disability and Utility
180. **Boman, Anders** (2009), Geographic Labour Mobility – Causes and Consequences
181. **Qin, Ping** (2009), Risk, Relative Standing and Property Rights: Rural Household Decision-Making in China

182. **Wei, Jiegen** (2009), Essays in Climate Change and Forest Management
183. **Belu, Constantin** (2009), Essays on Efficiency Measurement and Corporate Social Responsibility
184. **Ahlerup, Pelle** (2009), Essays on Conflict, Institutions, and Ethnic Diversity
185. **Quiroga, Miguel** (2009), Microeconomic Policy for Development: Essays on Trade and Environment, Poverty and Education
186. **Zerfu, Daniel** (2010), Essays on Institutions and Economic Outcomes
187. **Wollbrant, Conny** (2010), Self-Control and Altruism
188. **Villegas Palacio, Clara** (2010), Formal and Informal Regulations: Enforcement and Compliance
189. **Maican, Florin** (2010), Essays in Industry Dynamics on Imperfectly Competitive Markets
190. **Jakobsson, Niklas** (2010), Laws, Attitudes and Public Policy
191. **Manescu, Cristiana** (2010), Economic Implications of Corporate Social Responsibility and Responsible Investments
192. **He, Haoran** (2010), Environmental and Behavioral Economics – Applications to China
193. **Andersson, Fredrik W.** (2011), Essays on Social Comparison
194. **Isaksson, Ann-Sofie** (2011), Essays on Institutions, Inequality and Development
195. **Pham, Khanh Nam** (2011), Prosocial Behavior, Social Interaction and Development: Experimental Evidence from Vietnam
196. **Lindskog, Annika** (2011), Essays on Economic Behaviour: HIV/AIDS, Schooling, and Inequality
197. **Kotsadam, Andreas** (2011), Gender, Work, and Attitudes
198. **Alem, Yonas** (2011), Essays on Shocks, Welfare, and Poverty Dynamics: Microeconometric Evidence from Ethiopia
199. **Köksal-Ayhan, Miyase Yesim** (2011), Parallel Trade, Reference Pricing and Competition in the Pharmaceutical Market: Theory and Evidence
200. **Vondolia, Godwin Kofi** (2011), Essays on Natural Resource Economics
201. **Widerberg, Anna** (2011), Essays on Energy and Climate Policy – Green Certificates, Emissions Trading and Electricity Prices
202. **Siba, Eyerusalem** (2011), Essays on Industrial Development and Political Economy of Africa
203. **Orth, Matilda** (2012), Entry, Competition and Productivity in Retail
204. **Nerman, Måns** (2012), Essays on Development: Household Income, Education, and Female Participation and Representation
205. **Wicks, Rick** (2012), The Place of Conventional Economics in a World with Communities and Social Goods
206. **Sato, Yoshihiro** (2012), Dynamic Investment Models, Employment Generation and Productivity – Evidence from Swedish Data
207. **Valsecchi, Michele** (2012), Essays in Political Economy of Development
208. **Teklewold Belayneh, Hailemariam** (2012), Essays on the Economics of Sustainable Agricultural Technologies in Ethiopia
209. **Wagura Ndiritu, Simon** (2013), Essays on Gender Issues, Food Security, and Technology Adoption in East Africa
210. **Ruist, Joakim** (2013), Immigration, Work, and Welfare
211. **Nordén, Anna** (2013), Essays on Behavioral Economics and Policies for Provision of Ecosystem Services
212. **Yang, Xiaojun** (2013), Household Decision Making, Time Preferences, and

Positional Concern: Experimental Evidence from Rural China

213. **Bonilla Londoño, Jorge Alexander** (2013), Essays on the Economics of Air Quality Control
214. **Mohlin, Kristina** (2013), Essays on Environmental Taxation and Climate Policy
215. **Medhin, Haileselassie** (2013), The Poor and Their Neighbors: Essays on Behavioral and Experimental Economics
216. **Andersson, Lisa** (2013), Essays on Development and Experimental Economics: Migration, Discrimination and Positional Concerns
217. **Weng, Qian** (2014), Essays on Team Cooperation and Firm Performance
218. **Zhang, Xiao-Bing** (2015), Cooperation and Paradoxes in Climate Economics
219. **Jaime Torres, Monica Marcela** (2015) Essays on Behavioral Economics and Policy Design
220. **Bejenariu, Simona** (2015) Determinants of Health Capital at Birth: Evidence from Policy Interventions
221. **Nguyen, Van Diem** (2015) Essays on Takeovers and Executive Compensation
222. **Tolonen, Anja** (2015) Mining Booms in Africa and Local Welfare Effects: Labor Markets, Women's Empowerment and Criminality
223. **Hassen, Sied** (2015) On the Adoption and Dis-adoption of Household Energy and Farm Technologies
224. **Moursli, Mohamed-Reda** (2015) Corporate Governance and the Design of Board of Directors
225. **Borcan, Oana** (2015) Economic Determinants and Consequences of Political Institutions
226. **Ruhinduka, Remidius Denis** (2015) Essays on Field Experiments and Impact Evaluation
227. **Persson, Emil** (2016) Essays on Behavioral and Experimental Economics, Cooperation, Emotions and Health.
228. **Martinangeli, Andrea** (2017) Bitter divisions: inequality, identity and cooperation
229. **Björk, Lisa** (2017) Essays on Behavioral Economics and Fisheries: Coordination and Cooperation
230. **Chegere, Martin Julius** (2017) Post-Harvest Losses, Intimate Partner Violence and Food Security in Tanzania
231. **Villalobos-Fiatt, Laura** (2017) Essays on forest conservation policies, weather and school attendance
232. **Yi, Yuanyuan** (2017) Incentives and Forest Reform: Evidence from China
233. **Kurz, Verena** (2017) Essays on behavioral economics: Nudges, food consumption and procedural fairness
234. **Yashodha, Yashodha** (2017) Contract Choice and Trust in Informal Groundwater Markets
235. **Meles, Tensay Hadus** (2017) Power Outages, Increasing Block Tariffs and Billing Knowledge
236. **Mühlrad, Hanna** (2018) The Impact of Reproductive and Birth Technologies on Health, Fertility and Labor Outcomes
237. **Gakii Gatua, Josephine** (2018) Primary Health Care Interventions and Social Ties in Kenya
238. **Felgendreher, Simon** (2018) Essays on behavioral economics and the effects of the colonial rule on Java
239. **Demeke, Eyoual** (2019) Essays on Environmental and Behavioral Economics
240. **Kiss, Tamas** (2019) Predictability in Equity Markets: Estimation and Inference
241. **Khomenko, Maksym** (2019) Essays on the Design of Public Policies and .

Regulations

242. **Mukanjari, Samson** (2019) Climate Policy and Financial Markets
243. **Schürz, Simon** (2020) Economic and Intergenerational Decision-Making in Families
244. **Lau, Debbie** (2020) Empirical Essays on Education and Health Policy Evaluation
245. **Ho, Hoang-Anh** (2020) Essays on Culture, Institutions, and Economic Development
246. **Rubio Ramos, Melissa** (2020) The Economics of Coercive Institutions, Conflict, and Development
247. **Lindahl, Anna** (2021) Empirical tests of exchange rate and stock return models
248. **Sjöholm, Carolin** (2021) Essays on public policy in the informal sector context
249. **Tian, Ruijie** (2021) Impacts of Climate Policy and Natural Disasters: Evidence from China
250. **Larsson, Sebastian** (2021) Providers and Profiteers: Essays on Profits and Competition in the Provision of Public Services
251. **Norrgren, Lisa** (2022) Carpe Diem or Seize your Health? The Economics of Time Preferences, Health, and Education
252. **Andersson, Lina** (2022) Emotions in Game Theory: Fear, friendliness and hostility
253. **Enlund, Jakob** (2022) Three essays in applied economics
254. **Berggren, Andrea** (2023) Choices among Doctors, Students and Primary Care Providers: Empirical Evidence from Sweden
255. **Hansson, Magnus** (2023) Decentralized Finance and Central Bank Communication
256. **Bilén, David** (2023) Sustainable Consumption and Prosocial Actions
257. **Inkinen, Ville** (2023) Wetland Mitigation Banking in the United States