

The Value of Biodiversity: Evidence from Migratory Birds

Frederik Noack, Dennis Engist & Ashley Larsen

2025-06-23

Abstract

We estimate the economic value of biodiversity by estimating the value of pest control services provided by migratory birds to U.S. agriculture and forestry. Exploiting exogenous variation in migratory bird returns driven by winter habitat conditions in South America, we find that a 10% decline in avian biodiversity reduces crop revenues by 1.1% and increases forest pest outbreaks by up to 1.2%. We find that the elasticity of substitution among bird species is high in agricultural landscapes and lower in forests, implying that ecosystem service provision in agriculture depends more on the total abundance of birds, while in forests it depends more on the number of species. These findings identify biodiversity as a productive input and highlight cross-border externalities from environmental degradation.

1 Introduction

“That insectivorous birds are of immense value to the farmer and the forester is so well-known that their protection is now believed to be absolutely necessary to the welfare of any country.” (Henshaw, 1907)¹

Biodiversity is essential for ecosystem functioning and a wide range of economic activities that it supports. Yet, the economic consequences of its global decline remain largely unmeasured. While economists have long theorized the value of biodiversity (Weitzman, 1992, 2000; Brock and Xepapadeas, 2003; Polasky et al., 2005; Fenichel et al., 2024; Dasgupta, 2021), empirical estimates of its causal impact on economic production remain scarce. A growing literature has begun to document the economic impacts of species’ decline or recovery (e.g., Raynor et al., 2021; Frank and Sudarshan, 2024; Frank, 2024), but most studies focus on individual species or groups of similar species in isolation. This misses a central insight from ecology: ecosystem services provided by biodiversity depend

¹Henry Henshaw was the head of the U.S. Biological Survey and a proponent of the Weeks-McLean Act for protecting migratory birds. The Act was signed by President Woodrow Wilson in 1913. More than 100 years later, empirical evidence supporting the benefits of birds to farmers and foresters remains elusive.

not just on the presence of individual species, but also on their complementarities in delivering these services (e.g., Cardinale et al., 2012; Tilman et al., 2014). This paper fills this gap by providing a causal estimate of the economic value of biodiversity, taking the complementarities between species explicitly into account. We focus on ecosystem services provided by migratory birds to agriculture and forestry in the United States, addressing a major empirical challenge: biodiversity loss is often driven by economic activity itself (Liang et al., 2021), particularly through agricultural intensification and forest management (Tilman et al., 2017), leading to concerns about endogeneity. To address this, we exploit quasi-random ecological variation: species-specific habitat conditions in their South and Central American winter habitats that affect the return of migratory birds to their North American summer habitats during the subsequent growing season.² While we use birds as an example, the insights are generalizable to other ecological taxa.

Biodiversity is in steep decline, and North American birds are no exception. Since 1970, North America has lost nearly 3 billion birds—about 29% of its wild bird population, with the steepest losses occurring in agricultural landscapes (Rosenberg et al., 2019). Many once-common farmland birds, including swallows and sparrows, have become increasingly rare. Agricultural intensification and the widespread use of pesticides have directly harmed birds through mortality and reduced reproduction, and indirectly by eroding food supplies and habitats (Li et al., 2020; Rigal et al., 2023). The loss of wild biodiversity may have had severe economic consequences as wild species provide essential services: fish contribute to food security, bees pollinate crops, and birds, spiders, and bats suppress pests and disease vectors. Large-scale assessments have attempted to quantify these benefits (MEA, 2005; TEEB, 2010; Watson et al., 2019), but causal evidence linking biodiversity to economic outcomes remains limited. Recent studies have begun to fill this gap by leveraging variation in species populations (Raynor et al., 2021; Springborn et al., 2022; Frank, 2024; Frank and Sudarshan, 2024). These studies demonstrate that the collapse or reintroduction of species can substantially affect human welfare. However, they typically focus on the effects of a single species or homogeneous groups of species, ignoring interactions among species. Yet, such interactions, particularly complementarities and substitution effects (i.e. functional redundancy), are central to understanding the contribution of biodiversity to ecosystem functioning and the services ecosystems provide (Lehman and Tilman, 2000; Loreau and Hector, 2001).

We advance this literature by estimating the value of biodiversity, explicitly accounting for complementarities among species in providing ecosystem services. Specifically, we ask whether having more individuals of a beneficial species improves ecosystem services or whether having more species (species richness) enhances outcomes even when total abundance is held constant. For example, barn swallows feed on adult flying insects, while ground-feeding birds like bobolinks (also known as “armyworm

²Throughout the article, we will refer to Latin America as South America in ecological contexts.

birds”) target the larval stages of pests. Together, they may control insect pests more effectively than either species alone. Although ecologists have studied such complementarities using randomized field experiments (e.g., Tilman et al., 2006; Schnabel et al., 2021), translating these findings into real-world economic outcomes is difficult. In practice, species diversity is not randomly assigned, and ecosystem responses often trigger economic adjustments, such as increased pesticide use (Frank, 2024). These limitations have led to increasing skepticism about applying experimental ecological findings to policy contexts (Dee et al., 2023).

To address the question about the economic consequences of biodiversity loss, we begin by developing a theoretical framework of biodiversity and ecosystem service (ES) provision rooted in ecological principles. We model our simplified ecosystem as a community of interacting species with density-dependent population dynamics. Species affect ES provision both directly (through their own activity) and indirectly (through competition with other species). Using this framework, we show that an increase in a species’ population raises ES provision only if its direct contribution exceeds its competitive suppression of other species. Here, we describe the ES provided by biodiversity using a generalized constant elasticity of substitution (CES) function of species populations. This formulation allows for varying degrees of substitutability or complementarity among species. For example, increasing the number of barn swallows may improve pest control, but combining barn swallows and bobolinks can increase pest control even more, depending on the elasticity of substitution between the two species. Our CES formulation encompasses many commonly used biodiversity metrics, such as the Shannon and Simpson indices (Hill, 1973), and integrates with ES production functions from both the ecological and economic literature (Kremen, 2005; Luck et al., 2009; Giglio et al., 2024).

In our framework, species are migratory, and populations are affected by the environmental conditions in their winter and summer breeding habitats. While environmental conditions in the summer habitat affect all species within that habitat and possibly also the local economic outcomes, winter habitat conditions outside of the summer breeding habitat are species-specific and may or may not affect outcomes in the summer habitats directly. Next, we link the ecological community to economic activity. Economic production is harmed by natural pests, and pests are controlled by insectivorous species. Pesticides exist as a technological substitute for natural pest control. A second insight from our framework is that while pests, pesticide use, and agricultural or forestry outcomes all respond to ES provision, only the direct effects on agricultural and forest yields are welfare-relevant for the farmer and forester. However, compensatory increases in pesticide use may impose social costs beyond the farm and forest. This final step in our theoretical model guides our empirical estimation strategy.

We then use this framework to address the endogeneity challenge by deriving a novel empirical instrument for biodiversity. We exploit species-specific environmental shocks in winter habitats outside

the summer habitat to predict the return of migratory species to their summer habitats, where we then record the agricultural and forest outcomes. Because bird species vary in migratory patterns, and different regions of the US host different bird communities, we obtain spatial and temporal variation in biodiversity based on migratory bird species returns.

We apply this theoretical framework to quantify the economic impact of bird biodiversity in North American agriculture and forestry. Birds are ideal for this analysis: they are major predators of insect pests. Diet studies suggest over half of bird species primarily eat insects (Sekercioglu, 2006) and that they consume an estimated 28 million tons of arthropods annually in agricultural landscapes (Nyffeler et al., 2018). Experiments further show that excluding birds from plots increases pest damage and reduces yields (Martin et al., 2013; Mayne et al., 2023). Meanwhile, pests cause global crop losses of 20–40% (Oerke, 2006; FAO, 2022; Savary et al., 2019) and inflict billions in damages on forests (Aukema et al., 2011; Druckenmiller, 2020). Though pesticides can reduce losses, they entail health risks (Taylor, 2021; Dias et al., 2023; Frank, 2024) and contribute significantly to agricultural and forest production costs. U.S. agricultural pesticide expenditures are projected to reach \$18.1 billion in 2025, exceeding fuel costs.³

Migratory bird species allow us to link environmental shocks in South America to ES provision in North America. About 80% of U.S. bird species are migratory (Rosenberg et al., 2019), with insectivorous species especially dependent on southern habitats during winter. Their survival during migration depends heavily on conditions in their winter habitats (Marra et al., 1998; Dossman et al., 2023; Cooper et al., 2024). Using satellite data, we build an index of winter habitat conditions that predict migratory returns to U.S. summer habitats. Because the mix of migratory species varies across locations, this index generates spatially differentiated variation in ES provision. To illustrate our estimation strategy, consider two counties in South Dakota: one dominated during the summer by cliff swallows, which overwinter in a restricted region in Southern South America, and another dominated by barn swallows, which overwinter throughout South and Central America. A drought in Argentina would more severely reduce cliff swallow populations, affecting the first county's pest control more than the second. By comparing deviations of counties from their normal conditions within the same state and year, we can control for broader economic or climate shocks, isolating the effects of biodiversity-driven ES variation.

Combining breeding bird population data from the USGS Breeding Bird Survey with migration data, crop production records, pesticide use statistics, and forest pest incidence, we estimate the causal impact of ES provided by avian biodiversity on agricultural and forestry outcomes using a two-stage least squares strategy. Our results show that an increased provision of ES also increases crop sales and yields. However, the estimate depends on the underlying elasticity of substitution across

³<https://www.ers.usda.gov/topics/farm-economy/farm-sector-income-finances/farm-sector-income-forecast>

species. The magnitude and precision of the estimate for agricultural outcomes generally increase with the elasticity of substitution, suggesting a high substitutability among insectivorous species in the agricultural landscape. A 10% increase in ES increases yields by 1.3% and revenues by 1.1% in our preferred specification. These estimates differ across crops. While crops with generally low pesticide applications, e.g., due to genetically engineered pest resistance, respond little to variations in ES, other crops with generally high pesticide applications or a lack of genetically engineered pest resistance respond more to ES provision. Our interpretation is supported by our finding that pesticide use decreases with ES, although these results are not statistically significant. We find further that increased bird biodiversity reduces the incidence of forest pests. This impact is significant for forest pests like moths that are easily accessible for insectivorous migratory birds, and low for bark beetles that live below the bark of trees and are therefore mostly inaccessible for migratory bird species. The results for forest pests differ from our agricultural results in that the strongest and most precisely estimated effect of ES on pest incidence is for intermediate underlying elasticities of substitution among species, suggesting that complementarities between species may play a larger role in more natural ecosystems such as forests. Put differently, our results suggest that species complementarities play a larger role in more natural ecosystems, such as forests, whereas agricultural systems depend more on the total abundance of insectivorous birds.

We contribute to the literature on the economics of biodiversity and ecosystem services, which has largely been theoretical (Weitzman, 1992; Polasky and Solow, 1995; Weitzman, 2000; Brock and Xepapadeas, 2003; Fenichel et al., 2024; Dasgupta, 2021). Empirical studies on the economic value of biodiversity are scarce and restricted to correlational studies (Di Falco and Chavas, 2009) or simulations with calibrated models (Bertram and Quaas, 2017). Instead, the empirical literature has focused on the impact of conservation policy on biodiversity outcomes (e.g. Lueck and Michael (2003); Ferraro et al. (2007); Langpap et al. (2018); Hsiang and Sekar (2016)) and ecosystem services (Andam et al., 2008; Alix-Garcia et al., 2015; Jayachandran et al., 2017; Taylor and Druckenmiller, 2022; Assunção et al., 2023). An exception is the study by Giglio et al. (2024) who develop a theory of biodiversity and ES provision. They find support for their theoretical predictions using cross-country evidence from biodiversity degradation and financial markets. We build on and extend their framework by introducing competition among species and estimating causal effects on economic outcomes.

In sum, we demonstrate that biodiversity—not just the presence of individual species, but their composition and complementarity—has measurable economic value. Our findings suggest that the recent decline in bird populations may impose annual agricultural losses of over \$11 billion. These results underscore the urgency of biodiversity conservation and the importance of international coordination: the services ecosystems provide in one region often depend on ecological conditions in

another.

The remainder of this paper is structured as follows. Section 2 introduces the theoretical framework. Sections 3 and 4 describe and visualize the data. Section 5 outlines the empirical strategy. Section 6 presents our results. Section 7 discusses implications and concludes.

2 Theoretical Framework

Before introducing our data and empirical strategy, we develop a theoretical framework as the basis for our analysis. Our theoretical framework describes interacting species that provide ES. Species compete for resources, but they also interact in providing ES. While we apply this framework to a community of bird species and insect pests, nothing hinges on this specific interpretation, as the theoretical insights are generalizable beyond this application.

We use a standard density-dependent Ricker population model to describe population dynamics (e.g. May and McLean (2007) or Otto and Day (2011)). Additionally, we introduce environmental shocks such that the population of species i in location j in $t + 1$ can be described as

$$x_{ijt+1} = x_{ijt} e^{1 - \alpha x_{ijt} + f(w_{jt+1}^{hs}) + g(w_{it}^{hw})}. \quad (1)$$

Here, α equals one over the carrying capacity, and $f()$ and $g()$ measure the impact of the environmental shocks w_{it}^{hw} and w_{jt+1}^{hs} on the population of species i in location j . The difference between both shocks is that w_{it}^{hw} is species-specific, i.e., it affects all individuals of that species in their winter habitats. In contrast, w_{jt+1}^{hs} is community-specific and affects all species within the community during the subsequent breeding season in the summer habitat. Although this distinction is irrelevant to our theoretical insights, we will utilize it in the empirical section of the paper. These shocks could also describe the impact of agriculture along the migration routes and summer habitats on biodiversity. However, we do not focus on this feedback in the following.

Species compete for resources and territories. Introducing competition into the Ricker model, similar to e.g. May (1974) or Moll and Brown (2008) yields

$$x_{ijt+1} = x_{ijt} e^{1 - \sum_{l=1}^N \alpha_l x_{ljt} + f(w_{jt+1}^{hs}) + g(w_{it}^{hw})}, \quad (2)$$

where x_{ljt} is the abundance of species l in location j and N is the universe of species. The term α_l measures the species-specific competition effect, which measures the species' impact on individuals of the same and other species through competition. The parameter α_l plays an important role in the following as it indirectly measures the efficiency of ES provision. Species with larger α need more

resources to produce the same amount of ES.

2.1 Ecosystem Services

Birds provide important ecosystem services (ES) such as pest control. The predation of insect pests is one of the most important contributions of birds to human well-being (Whelan et al., 2015). The provision of ES generally depends on the species characteristics as well as the diversity of species (Kremen, 2005; Cardinale et al., 2012). The control of pest populations may, for example, depend on the effectiveness of individual species as well as the combination of different insectivorous bird species that prey on different pests or different stages of the same pests (e.g., caterpillars and moths). Here, we model ES as a constant elasticity of substitution (CES) production function of individuals and species:

$$ES_{jt} = \bar{x}_{jt}^\iota \left(\sum_{i=1}^N x_{ijt}^{\rho_1} \right)^{\frac{1}{\rho_2}} \quad (3)$$

where ι , ρ_1 , and ρ_2 are parameters, $\bar{x}_{jt} := \sum_{i=1}^N x_{ijt}$ is total abundance of individuals across all species in community j and N is the total number of species. The parameter ι measures the importance of absolute abundance, compared to relative abundance, on which most biodiversity measures are based. The parameters ρ_1 and ρ_2 are related to the elasticity of substitution across species in providing ES. This general functional form includes two important special cases.

Definition 1. Effective species numbers: For $\rho_2 = 1 - \rho_1$, and $\iota = -\frac{\rho_1}{\rho_2}$ the function becomes the effective species numbers function proposed by Hill (1973) which also includes indices such as the Shannon or Simpson diversity indices as special cases, depending the the value of ρ_1 .

Definition 2. Ecosystem service production: For $\iota = 0$, and $\rho := \rho_1 = \rho_2 = \frac{\sigma-1}{\sigma}$ the function equals the constant elasticity of substitution (CES) ecosystem service production function proposed by Giglio et al. (2024). The parameter σ measures the elasticity of substitution among species in producing ecosystem services.

The larger the parameter σ , the higher the elasticity of substitution among species. Generally, ES provision increases in the total number of individuals (abundance \bar{x}_{jt}) and the total number of species (N) if ES provision is described by an ecosystem service production function and $\sigma > 1$. ES provision increases in the total number of species and is independent of the absolute number of individuals per species (scale invariant) if effective species numbers describe ES provision. The proof is given in Appendix A.2. We focus on the ecosystem service production function because the recent empirical literature suggests that population decline leads to reduced ecosystem service provision. In the following, we therefore assume the parameters of Definition 2. We illustrate the impact of the

choice of σ on ES in Section 4.

Two extensions are worth considering. First, species may differ in their contribution to ecosystem service (ES) provision. Fenichel et al. (2024) incorporates this by multiplying each individual population size x_{ijt} by an ES efficiency parameter. While this extension is conceptually intuitive, we retain the simplified version from above for two reasons. First, the efficiency parameter is empirically unobserved, making identification difficult in the empirical analysis below. Second, species-specific efficiency is already captured in our model by the resource use efficiency parameter α . As such, introducing an additional parameter would complicate the notation without yielding new theoretical insights.

Second, we have assumed constant returns to scale in the ES function. A straightforward extension would be to introduce a returns-to-scale parameter in the exponent $\frac{1}{\rho_2}$. However, this modification also offers no additional theoretical insight and would merely rescale the empirical estimates because our empirical strategy relies on a log transformation of the ES function.

Next, we introduce the individual population dynamics into the ES function, such that it becomes:

$$ES_{jt} = \left[\sum_{i \in N} \left(x_{ijt-1} e^{1 - \sum_{l=1}^N \alpha_l x_{ljt-1} + f(w_{jt}^{hs}) + g(w_{it-1}^{hw})} \right)^\rho \right]^{\frac{1}{\rho}} \quad (4)$$

This expression highlights the opposing forces that an increasing population of a species exercises on the overall ES provision. On the one hand, more individuals of a species increase the provision of ES from this species. On the other hand, more individuals of that species also increase the competition with other species that also provide ecosystem services. Whether increasing the population size of a species also increases the provision of ES, therefore, depends on the elasticity of substitution among species (captured by ρ) and the abundance of species in the ecosystem. This insight leads to our first formal result:

Proposition 1. *A population increase of species i at time $t-1$ increases ES in t only if*

$$\frac{x_{ijt}^\rho}{\sum_{l=1}^N x_{ljt}^\rho} > \alpha_i x_{ijt-1}.$$

The proof is in Appendix A.3. It is easy to verify that the condition only holds for some parameter combinations.⁴ The left-hand side of the inequality is an index that measures the contribution of a marginal population increase of species i on ES production. The right-hand side measures the negative impact of its increased abundance on other species (and itself) through competition. Another

⁴Consider, for example, the limiting case of $\sigma \rightarrow 1$ and therefore $\rho \rightarrow 0$. The expression then converges to $(N\alpha_i)^{-1} > x_{ijt-1}$. In this case, the condition is met for a population below its carrying capacity divided by the total number of species, and not met for a population above its carrying capacity divided by the total number of species. Recall that α_i is one over the carrying capacity.

interpretation of α is the efficiency in ES provision. A large α_i of a species i implies that an ecosystem with that species at its carrying capacity can only provide limited ES. In contrast, the same ecosystem with a different species at its carrying capacity can provide more ES if its α is smaller than that of species i . Recall that the carrying capacity is $1/\alpha$. An important implication of this insight is that the decline of a species does not necessarily lead to a reduction in ES if population increases of competing species overcompensate the reduced ES provision by that species. While we cannot test this proposition directly in the empirical part of this paper, we illustrate a case where the increase of a group of beneficial species reduced ES provision through the suppression of another group of more efficient beneficial species in Section 6.9.

2.2 Ecosystem Services and the Control of Agricultural Pests

A major ecosystem service provided by biodiversity is the control of pests. In the following, we describe the relationship between bird populations, agricultural pests, and agricultural production. The model also describes the substitution between ES and their technological alternative.

Probably the most common function in modeling predator-prey interactions is the Lotka-Volterra model, which we adopt in the following with pesticide use and ES as perfect substitutes for pest control. Lichtenberg and Zilberman (1986) suggests that "In pest management, for instance, one would expect to find decreasing marginal productivity of abatement because further reductions in damages tend to decline as pest populations get smaller." We therefore suggest declining marginal productivity of ES and pesticides, and describe the pest dynamics as

$$\dot{Z}_{jt} = Z_{jt} \left(1 - \frac{Z_{jt}}{\kappa} \right) - \gamma Z_{jt} \log(ES_{jt}) - \mu Z_{jt} \log(M_{jt}) \quad (5)$$

where \dot{Z}_{jt} is the derivative of the pest stock with respect to time and M_{jt} is the agricultural pest control effort, e.g., pesticide use.

Insect pest generation time is short compared to the generation time of birds. Further, birds often feed on a variety of insects, such that their population is not closely coupled to pests. We, therefore, make the simplifying assumption that pests reach a steady state immediately, taking the bird population as given:

$$Z_{jt}^* = \kappa (1 - \gamma \log(ES_{jt}) - \mu \log(M_{jt})). \quad (6)$$

The steady-state insect pest population declines linearly in bird populations and pest control effects.

2.3 Agricultural Pests and Agricultural Production

We now introduce economic production into our simplified ecosystem, using agriculture as an example. We describe agricultural profits as a revenue function $p\psi_{jt}d(Z_{jt}^*)$ minus the costs of pest control, $c(M_{jt})$. Here, p is the crop price and ψ_{jt} is the time- and location-specific potential output in the absence of pest losses (Lichtenberg and Zilberman, 1986) but with optimally chosen other production inputs such as fertilizer or irrigation. While the assumption of optimally chosen inputs seems restrictive, we show in Proposition 2 that the adjustment of inputs is welfare irrelevant for the farmer.

The damage function is a function of the pest stock Z_{jt}^* . Recall that the pest stock is also a function of pest control (6) such that the damage function is indirectly a function of pest control and ES. The pest management problem of the farmer, therefore, becomes

$$\max_{M_{jt}} p\psi_{jt}d(Z_{jt}^*) - cM_{jt}. \quad (7)$$

where c is the per-unit cost of pesticide. To describe pest damage, we assume an exponential damage function given by

$$d(Z_{jt}^*) = e^{-Z_{jt}^*}.$$

The advantage of this functional form is that the output approaches zero as the pest stock approaches infinity, while it equals the potential output, ψ_{jt} , if the pest stock becomes extinct.

The solution to the farmer's problem with $\mu\kappa < 1$ is

$$M_{jt}^* = \phi_{jt}ES_{jt}^{\frac{\gamma\kappa}{1-\mu\kappa}} \quad (8)$$

where ϕ_{jt} is a parameter defined in Appendix A.4. The condition $\mu\kappa < 1$ ensures an interior solution to the problem. We provide further details in Appendix A.4. Equation (8) suggests that pesticide use increases with ES provision as a result of their complementarities in the production process. However, we will test this relationship empirically below. Using this result in the expression for agricultural yields gives the optimized yields as a function of ES:

$$Y_{jt}^* = \phi'_{jt}ES_{jt}^{\frac{\gamma\kappa}{1-\mu\kappa}} \quad (9)$$

The definition of the parameter ϕ'_{jt} and further details are provided in Appendix A.4. This expression suggests that yields increase monotonically in the provision of ES, taking the endogenous response of pesticide use into account. Based on this framework, we can derive the welfare implications of increases (or decreases) in ES provision.

Proposition 2. *A marginal change in ES affects farm profit only directly through its impact on crop yields.*

The proof is based on the Envelope theorem and is provided in Appendix A.5. The intuition is that the impact of marginal input adjustments on profits is negligible because the marginal benefit equals the marginal cost of inputs at the optimum.

While this result applies to farm profit, it does not fully describe welfare outcomes if externalities exist. Recent findings show that pesticide use negatively affects human health outcomes (Taylor, 2021; Dias et al., 2023; Frank, 2024; Reynier and Rubin, 2025), suggesting substantial negative externalities. Therefore, a change in ES could have additional welfare-relevant consequences through pesticide adjustments.

Corollary 1. *The welfare consequences of changes in ES are determined by the direct impact of ES on yields and the externalities from compensatory input adjustments.*

This result follows from the fact that externalities are not considered in the farmer's optimization problem. They therefore do not cancel out in the input adjustment process as suggested by the Envelope theorem for optimized profit functions. While these results are specific to agriculture, they similarly apply to other economic production dependent on ES, including forestry, fisheries, hydropower generation, and water purification. We will apply this framework in the empirical section of the paper to outcomes in agriculture and forestry. The next section describes how we take the theoretical insights to the data.

2.4 Testing the Theoretical Predictions

Throughout the empirical approach, the endogeneity of biodiversity is the key challenge. We are especially concerned that agricultural intensification (e.g. increased pesticide use), expansion of agricultural production, and forest operations drive the decline of ES. This relationship could lead to reverse causation, i.e., that economic production determines biodiversity levels. We address this problem using a two-stage least squares (2SLS) approach in which we use shocks to the winter habitats of migratory birds to predict their return to their summer habitats in the US. The largest drawback of this strategy is that shocks could be correlated across continents through large weather phenomena such as El Niño events or price effects from shocks to South American crop production. Fortunately, the spatial variation in the presence of migratory bird species in North American summer habitats allows us to compare areas with migratory bird shocks to similar areas without shocks to migratory birds in the same year and state. The main purpose of this section is to derive the instrument from the theory.

In the following, we assume a declining marginal impact of environmental shocks on population dynamics for two reasons. First, while our estimation strategy does not hinge on the assumption, it simplifies the approach and provides a credible functional form of the estimation equation. Second, the assumption seems plausible for the impact of resource shocks on population growth, which is the relevant case in our empirical specification. Specifically, we assume $f(w_{jt+1}^{hs}) := \log(w_{jt+1}^{hs})$ and $g(w_{it}^{hw}) := \log(w_{it}^{hw})$ which underlies all empirical specifications below.

To predict the return of migratory birds to their summer habitats, we use shocks to their South American winter habitats during their wintering periods. In our model, this is equivalent to $g(w_{it}^{hw})$ while $f(w_{jt+1}^{hs})$ describes the environmental conditions in the North American summer habitats, which may or may not be correlated with w_{it}^{hw} .

Taking logs, the ES provision defined by equation (4) can be written as

$$\log(ES_{jt}) = 1 - \sum_{l=1}^N \alpha_l x_{ljt-1} + \log(\bar{x}_{jt-1}) + \log(w_{jt}^{hs}) + \rho^{-1} \log \left[\sum_{i=1}^N (s_{ijt-1} w_{it-1}^{hw})^\rho \right] \quad (10)$$

where \bar{x}_{jt-1} is the total number of individuals across all species in ecosystem j at time $t-1$. The mathematical details are in Appendix A.6. This is our main first-stage estimation equation, which explains variations in ES provision. The last term of this equation is a winter habitat weather shock index, which we will use as our instrument for ES. This allows us to estimate the causal impact of ES on agricultural output. We further provide estimates on the impact of ES on pest populations and pesticide use to support the underlying mechanisms and to provide evidence for potential welfare-relevant externalities from pesticide adjustments. Estimating the effect of ES on pest populations, pesticide use, and yields is straightforward. We directly estimate equation (6) and log-transformed versions of equations (8) and (9). We describe the details of the estimation strategy after introducing the data.

3 Data

Here, we describe the data that we use to measure winter habitat shocks, ES, pest abundance, pesticide use, crop yields, and growing-season summer habitat weather controls. We begin with the data on bird abundance and migration, as they underlie the calculation of ES and the definition of winter habitats.

3.1 Bird Abundance

We utilize bird abundance data from the North American Breeding Bird Survey (BBS), a citizen science initiative managed by the United States Geological Survey (Ziolkowski et al., 2024), which began in 1966. Every year in the summer breeding season, volunteers count the abundance of all bird species at thousands of observation routes across North America. These observation routes are spread out across the United States. Volunteers drive along the roughly 40km routes and stop 50 times at regular intervals to count birds. At each stop, the volunteer counts all birds that are seen or heard within a 400m radius of the stop. The data is then aggregated at 10-stop intervals (route segments), which is the observation unit for our analysis. We observe abundance for each species-route-segment-year combination. The BBS data contains a separate vector file that precisely lays out the observation routes. The route segments and counting protocol remain consistent over time, allowing observations from the early years of the BBS to be directly comparable to more recent observations at the same route.

3.2 Ecosystem Services

We measure bird ES using the ES production function defined by (3) and Definition 2. The function requires an assumption about the elasticity of substitution parameter $\rho := \frac{\sigma-1}{\sigma}$. The choice of σ is crucial for our analysis since it determines the relative weights of species richness (the number of species) versus the total abundance across all species (the number of individuals) in the provision of ES (see Section 2). Therefore, we calculate ES for a range of σ values between 1.1 and infinity, which covers the range of values it can take (see Section 2).

Here, we focus on species that provide pest control services. We therefore restrict the data to all birds with an invertebrate diet of >25% during the growing season from April to September based on the SAviTraits database (Murphy et al., 2023) before calculating the ES. We call these species "insectivorous" in the following. Note, however, that our measure of ES contains both migrant and non-migrant bird species.

Further, birds that live on farmland provide most of the ES to agriculture. We therefore restrict the bird species to the 75 farmland species listed in Stanton et al. (2018) when quantifying the impact of ES on crop sales, yields, and pesticide use. Similarly, only forest species are likely to provide efficient pest control in forests. We therefore restrict the species to forest species, based on Rosenberg et al. (2019), when quantifying the impact of ES on forest pests. Note that we are unable to use the Rosenberg et al. (2019) dataset to define farmland birds because farmland is a combination of different habitat types not captured by an individual habitat category.

Out of the 65,050,143 observed birds in our data set, 83% are insectivorous, 54% are insectivorous

farmland birds, and 29% are insectivorous forest birds.

3.3 Long-Distance Migratory Species

We construct our instrument using common long-distance migratory bird species. Specifically, we select those species out of the 100 most common species in the dataset whose winter habitats lie predominantly outside the United States (i.e., <10% of their winter range falls within the U.S.). This selection criterion ensures that environmental conditions in the winter habitat do not simultaneously determine bird and agricultural outcomes in the summer habitat. We refer to these species as ‘long-distance migrants’ throughout the paper. A complete list of these species is provided in Appendix A.7.

We define winter habitats based on the seasonal bird habitat data from BirdLife (BirdLife, 2018). Based on this criterion, we select 38 species, of which 37 are insectivorous. We observe 11,810,602 long-distance migratory birds in our data, about one-fifth of all bird observations. Out of those, 55% are insectivorous farmland species and 55% are insectivorous forest species. Note that farmland and forest species are not mutually exclusive categories.

We then determine summering, wintering, and migration months of each bird species using the Bird Migration Explorer platform.⁵ Knowing both the timing and location of the movement of each species, we can determine their exposure to environmental conditions in their winter habitat.

Throughout the paper, we refer to the winter habitats as habitats in South America, although parts of the winter habitat may include parts Central and North America.

3.4 Winter Habitat Conditions

A large fraction of adult bird mortality is explained by the winter habitat conditions (See, for example, Marra et al. (1998), Dossman et al. (2023), and Cooper et al. (2024)). We measure habitat conditions using the satellite-derived normalized difference vegetation index (NDVI) (Pinzon et al., 2023). NDVI is commonly used to measure agricultural productivity (Burke and Lobell, 2017), droughts (Liu et al., 2023), or deforestation (Schultz et al., 2016). The reason for these broad applications is that NDVI generally measures vegetation health, which is influenced by weather, nutrient input, and land use change. NDVI, therefore, captures both habitat loss due to, e.g., deforestation, as well as habitat conditions such as human-induced habitat degradation or changes in habitat conditions due to, e.g., weather events, that can be observed in optical satellite imagery. We calculate the average NDVI in the species-specific winter habitat over the species-specific winter period for each year of the study period. In the regression, we use the NDVI index of the preceding winter period to predict the return of the long-distance migrants to their summer habitats in the subsequent breeding period.

⁵Bird Migration Explorer, <https://explorer.audubon.org/home?layersPanel=expand>

3.5 Breeding Habitats Conditions

Summer habitat weather conditions during the growing season may affect yields, pest abundance, and bird populations simultaneously. Although the selection of species for our 2SLS approach and the state-year fixed effects addresses the concerns arising from correlated large-scale weather patterns, we additionally control for growing season weather following the literature on climate and agriculture (Roberts et al., 2013). Specifically, we describe weather in the summer habitats using growing degree days (GDD), killing degree days (HDD), and precipitation during the growing season from April to September in all specifications as controls. We base their calculation on temperature and precipitation data from NOAA (Durre et al., 2022).

3.6 Forest Pests and Forest Cover

Data on pests in agriculture are largely absent. However, the US Forest Service provides detailed geospatial data on forest pests in its Insect and Disease Detection Surveys (IDS). Forest pests are a particularly interesting application because 1) forest pests, such as spongy moths and pine beetles, cause billions of dollars in damage annually (Lovett et al., 2016), and 2) the forest industry still relies heavily on natural pest control (Marini et al., 2022). The IDS provides shapefiles of thousands of pest outbreaks every year that it maps using aerial surveys across the US. The data are available from 1997 to 2022. We calculate the area of the pest outbreak polygons that overlap with buffer areas around each bird route section. We use 1, 5 and 10 km buffers with 5 km buffers in our main specification. For the analysis, we restrict the observations to those with some forest cover in the year 2000 based on land cover data from the European Space Agency (CCI, 2017).

3.7 Pesticides

Pesticide use may also respond to ES as predicted by our theoretical framework. Specifically, we use data from the USGS Pesticide National Synthesis Project. These data provide survey-based estimates of county-level pesticide use by compound from 1992 to 2019. We classify these substances into the broad categories of herbicides, insecticides, and fungicides, following, e.g., Frank (2024) and Engist et al. (2024). We focus on insecticides because we expect them to respond directly to changes in ES related to insect pests. We use the "high" estimates that interpolate missing data instead of setting them to zero.

3.8 Crops

We use county-level crop area and production data from the USDA Agricultural Census spanning 1969 to 2022. Because not all variables are reported in every census year, temporal coverage varies by crop. Major field crops such as corn and wheat have consistent area and production data throughout the period, while data for some fruits, including apple production, are unavailable after 1997. In some counties, data are suppressed due to confidentiality concerns when only a single or very small number of producers are present.

Area is reported in acres, and production is recorded in crop-specific units (e.g., bushels, tons, hundredweights, or pounds). We construct yield measures as production per acre harvested. In our analysis, we focus on yield and sales data for a broad set of crops, including corn, cotton, rice, sorghum, soybeans, sugarbeets, wheat, almonds, apples, cherries, grapes, oranges, strawberries, walnuts, beans, blueberries, pistachios, potatoes, and sugarcane.

To account for heterogeneity across crops, we include crop-by-year fixed effects in our joint yield regressions. We also estimate crop-specific regressions for corn, soybeans, wheat, and apples—four economically significant crops that are widely grown across the U.S. and vary in their dependence on ecosystem services. While corn and soybeans are largely genetically modified (GM) crops in the US with partial pest resistance (Noack et al., 2024), wheat and apples are non-GM crops and may respond differently to ES. Further, previous studies have shown that apple yields respond to the ES provided by insectivorous birds (García et al., 2021). We therefore add apples as a benchmark case. The dataset is structured at the county-by-census-year level, covering the years 1969, 1974, 1978, 1982, 1987, 1992, 1997, 2002, 2007, 2012, 2017, and 2022.

3.9 Final Datasets

The datasets differ in their spatial resolution and temporal coverage. The availability of NDVI data from 1983 to 2022 defines our overall study period. Agricultural outcomes are observed only in agricultural census years, while data on forest pest outbreaks are available from 1997 to 2022. These data availability constraints define the specific study periods for our analyses of agriculture and forests, respectively. For the forest analysis, we further restrict the sample to route segments with forest cover, as the absence of observed pests may otherwise reflect either healthy forests or a lack of forested land. A similar restriction is not required for the agricultural dataset, since yield and sales data are reported only when crops are actually produced. BBS observation routes often span several counties. Our unit of observation is the bird route segment. We cluster standard errors at both the bird route and county levels to account for potential correlation in errors arising from repeated crop outcomes within counties and the clustered sampling design at the route level (see discussion below).

4 Descriptive Statistics

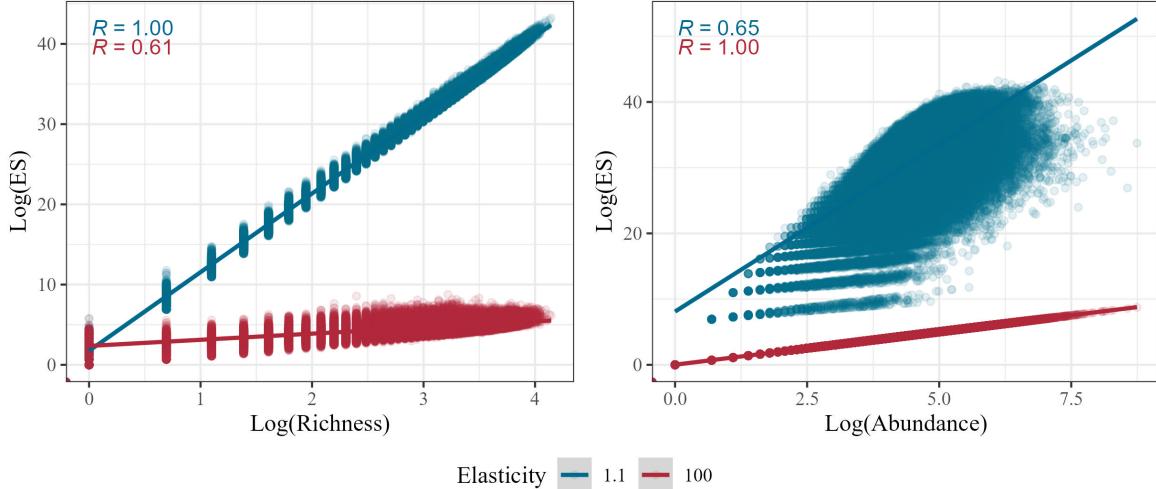
Here we visualize the data, focusing on biodiversity and the ecosystem services it provides. We also show the variation of the data that compose our instrument to provide support for the instrument's validity. The summary statistics of the data are provided in Appendix A.8.

4.1 Biodiversity and Ecosystem Services

We begin by examining the relationship between our ecosystem services (ES) measures and standard biodiversity metrics, specifically species abundance and species diversity. At one extreme is species richness, which captures only the number of distinct species and disregards their relative abundance. Species richness is highly correlated with the ES measure when the elasticity of substitution across species is low, but this correlation weakens as the elasticity increases (Figure 1).

At the other extreme is species abundance, which aggregates the total number of individuals across all species without accounting for species diversity. The ES measure is more strongly correlated with species abundance when the elasticity of substitution is high, and less so when the elasticity is low.

Figure 1: Species Richness, Total Abundance, and the Elasticity of Substitution in Ecosystem Service Provision



Notes: Correlation between Log(Species Richness) and Log(ES) (left panel) and Log(Total Abundance) and Log(ES) (right panel) for high (100) and low (1.1) elasticities of substitution to calculate ES. The data include all insectivorous birds. We report the Pearson correlation coefficient (R) in the top left corner of each panel.

ES vary over time, independent of whether we assume a low or high elasticity of substitution across species. We report the aggregate and individual dynamics in Appendix A.9.

4.2 Migration Pattern and Winter Habitat Conditions

We construct our instrumental variable based on variation in bird migration patterns. These patterns differ substantially across species, leading to differential exposure to winter habitat conditions. The instrument leverages three sources of variation: (i) differences in migration routes across species, (ii) variation in the timing of migration, and (iii) temporal fluctuations in winter habitat conditions. We discuss each source of variation in more detail below.

Figure 2 illustrates the summer and winter habitats of the five most common long-distance migratory species in our sample. We added the bobolink because we used it as an example throughout the introduction. While summer habitats exhibit considerable overlap across species, winter habitats differ markedly, often with limited geographic overlap.

These species differ not only in their winter ranges but also in the timing of their migration. While some species arrive late in their winter habitats and depart early, others stay for extended periods. These different migration behaviors result in significant differences in winter habitats among species. Additionally, winter habitat conditions vary over time. Land use changes, as well as weather conditions, affect NDVI within the same location over time.

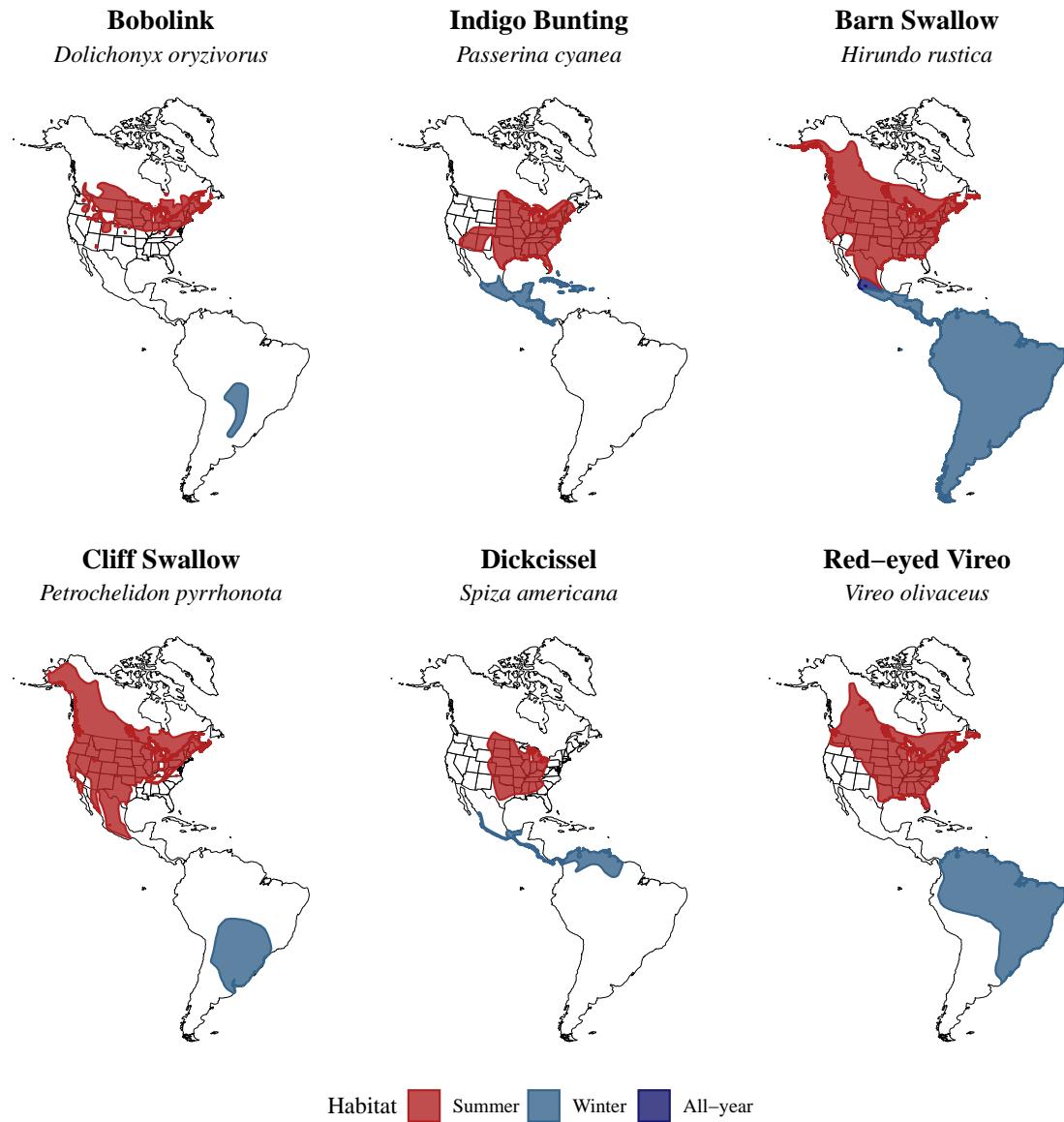
The combination of these two sources of variation creates fluctuation in winter habitat conditions that vary across species and over time. Figure 3 shows NDVI dynamics for all long-distance migratory species over time. Despite large-scale deforestation in the Amazon and other land use changes, the short-term fluctuations in NDVI seem to dominate the long-term changes.

The last source of variation for the instrument comes from the different distributions of birds in their North American summer habitats. The share of migratory species varies across space (Figure 4 left panel), largely independently of the level of ES (Figure 4 right panel). This variation allows us to compare counties with high and low shares of migratory species within the same states in years with good and bad South American winter habitat conditions. In addition to these spatial variations, there is variation in the species composition of long-distance migrants. In other words, two areas with the same share of migratory species could be exposed to different winter habitat shocks because of their different species composition. We show a map of the dominant long-distance migrant for each county in Appendix A.1.

4.3 Agriculture and Forestry

The primary outcomes of our empirical analysis relate to agricultural and forest production. In the agricultural sector, we focus on crop yields and revenues as our main outcome variables. We pay particular attention to corn, soybeans, and wheat, which are among the most economically significant and widely cultivated crops in the United States. We also include apples in the analysis, as they

Figure 2: Summer and Winter Distribution of Common Long-Distance Migratory Species

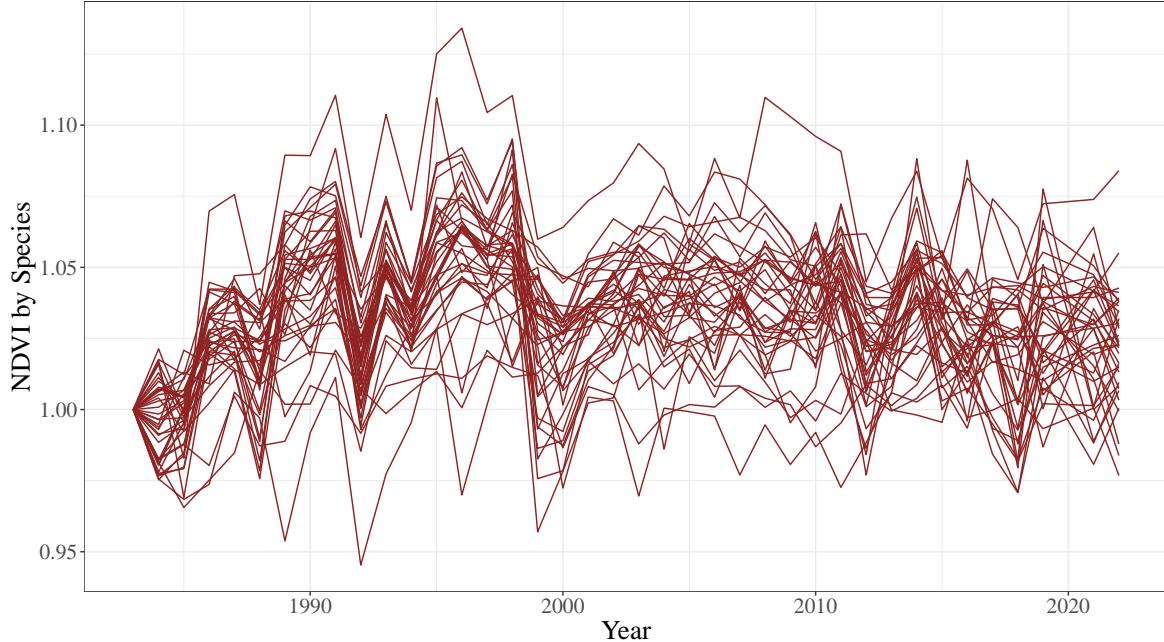


Notes: Winter (blue) and summer (red) habitat of six common long-distance migratory species.

are the most commonly grown fruit tree. However, apple production is more geographically limited compared to field crop production, with more than 100 acres of production observed in only 460 counties (Figure 5).

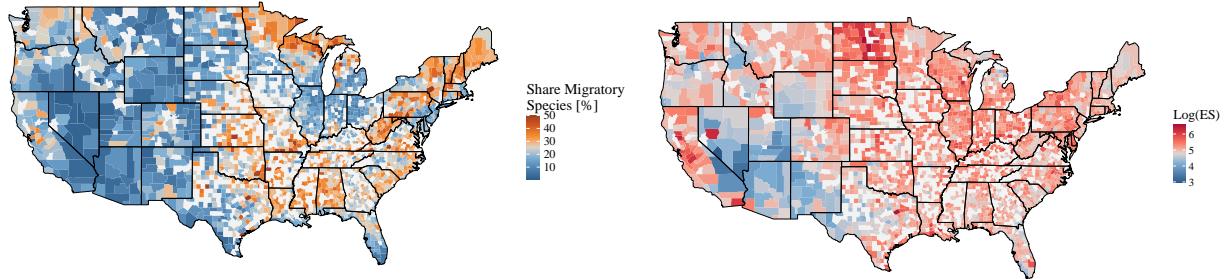
In the context of forests, we focus on insect pests, primarily bark beetles and moths, which are by far the most prevalent and damaging forest pests (see Appendix A.10). Other pests, such as thrips and leaf miners, are less common and are typically not targeted by birds due to their small size.

Figure 3: Winter Habitat Conditions



Notes: Each line shows the species-year-level average winter habitat NDVI relative to the species-level NDVI in 1983.

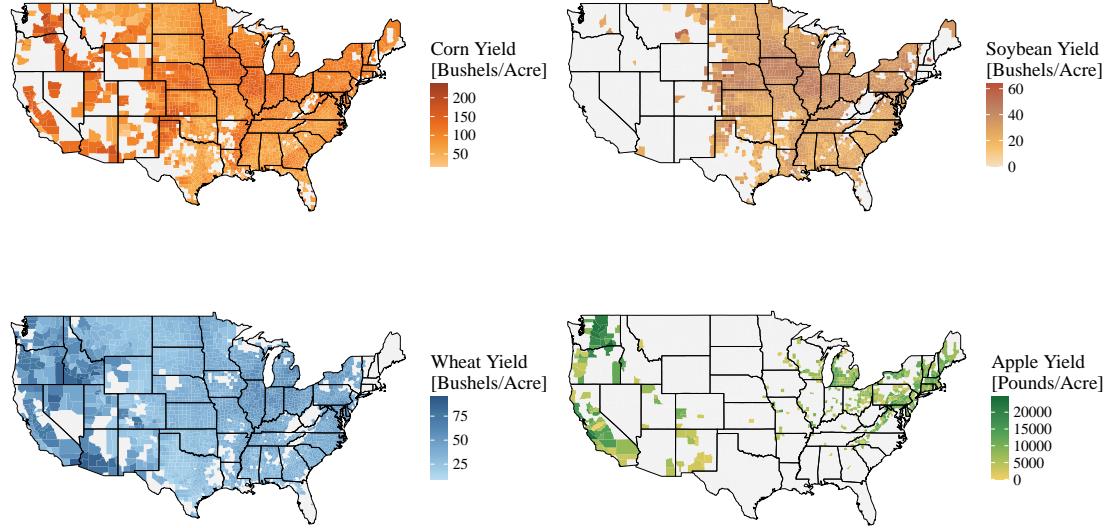
Figure 4: Share of Long-Distance Migrants and Ecosystem Service Levels



Notes: This figure reports the mean share of long-distance migrants in the total bird population (left panel) and the mean ecosystem service level based on a $\sigma = 10$. Means are computed over the entire study period and averaged within each county across all route segments. Missing values are in white.

We do not attempt to quantify the impact of ecosystem services (ES) on forest production directly. This is because pest outbreaks often induce shifts in harvest behavior: landowners may increase timber harvests in the short term to mitigate damage, leading to temporarily higher revenues, but also altered harvest dynamics over a longer horizon. These dynamics complicate the identification of a causal effect on forest output.

Figure 5: Crop production



Notes: The shading represents mean crop yields throughout the study period for areas exceeding 100 acres of production. Gray-shaded areas indicate that the crops are not grown.

5 Empirical Approach

The main challenge in estimating the impact of ES on agricultural and forest outcomes is that agricultural and forest outcomes also determine the levels of ES provision. Our main empirical strategy, therefore, uses variations in the species-specific South American winter habitat conditions to predict the return of migratory birds to their North American summer habitat. We develop the estimation equations based on our theory sections 2. We begin by translating the theoretical insights into estimable equations for our first stage, before describing the second-stage estimation.

5.1 Winter Habitat Condition and the Return of Migratory Species

We begin by estimating how winter habitat conditions affect the return of individual bird species to their summer habitats using a panel of species-level data. We take logs of equation (2), incorporating species (ν_i), route-segment (η_j), and year (θ_t) fixed effects:

$$\log(x_{ijt}) = \log(x_{ijt-1}) - \sum_{l \in N} \alpha_l x_{ljt-1} + f(w_{jt}^{hs}) + g(w_{it-1}^{hw}) + \nu_i + \eta_j + \theta_t + \varepsilon_{ijt}. \quad (11)$$

Here, we cannot include year-species fixed effects because it would absorb the species-level winter habitat shocks. The term $f(w_{jt}^{hs})$ represents an inverse hyperbolic sine (ihs) transformation of growing degree days, killing degree days, and precipitation, describing the summer habitat conditions that are (mostly) exogenous to local land use. In addition, $g(w_{it-1}^{hw}) = \text{ihs}(\text{NDVI})$ captures winter habitat quality as defined in section 3 in the preceding winter. Although the ihs transformation may introduce bias in the estimation (Chen and Roth, 2024), this specification only serves to illustrate the mechanism that connects winter habitats in South America to North American summer habitats. We therefore focus on the qualitative results for this specification.

The lagged dependent variable, $\log(x_{ijt-1})$, captures the positive effect of population size on reproduction, reflecting that larger populations tend to produce more offspring. The competition term, $\sum_{l=1}^N \alpha_l x_{ljt-1}$, is expected to be negative, reflecting density-dependent effects. Since the competition parameters α_l are unknown, we set them equal to one in the baseline specification. However, to address potential endogeneity biases, for example, from long-term agricultural trends, we omit both lagged population terms in our main estimation.

A key empirical challenge lies in constructing an appropriate counterfactual, as we cannot distinguish between winter habitat conditions across different populations within a single species. In other words, all individuals of a species are exposed to the same winter habitat conditions. To address this, we estimate the effects jointly across species. In essence, species experiencing relatively favorable winter habitat conditions serve as a counterfactual for those facing less favorable conditions, conditional on species, route segment, and year or year-by-state fixed effects. Our identification strategy thus relies exclusively on within-year variation in winter habitat shocks across species, as illustrated in Figure 3.

A note on notation: for brevity, we use a generalized notation for fixed effects throughout and do not distinguish between different regression specifications. While this is not mathematically precise, it simplifies exposition and reduces notational clutter.

5.2 Predicting Ecosystem Services with Winter Habitat Conditions

The main focus of this paper is, however, the impact of ES on agricultural and forest production, using an index of winter habitat condition as an instrument for ES. Our first-stage estimation of ES is based on equation (10). While the theory section develops winter habitat conditions as a plausible instrument for ES in North America, unobserved heterogeneity across time and space may bias the estimation and threaten the instrument's validity. We therefore include route-segment and year fixed effects in our baseline specification and additional year-by-state fixed effects in a robustness test. These specifications absorb fluctuations caused by large-scale weather phenomena or agricultural price

shocks, which may connect habitat conditions in the winter and summer habitats. A second concern is that the lagged total population terms capture local dynamics, such as agricultural intensification or climate change, which could affect bird populations as well as agricultural and forest outcomes in subsequent years. To address these challenges, we drop both lagged population variables in our main specification. With these modifications, the estimation equation for the impact of winter habitat shocks on ES provided by migratory birds described by equation (10) becomes

$$\log(ES_{jt}) = \text{ih}(w_{jt}^{hs}) + \beta_e \text{ih} \left[\sum_{i=1}^N (s_{ijt-1} w_{it-1}^{hw})^\rho \right] + \text{ih} \left(\sum_{i=1}^N s_{ijt-1}^\rho \right) + \eta_j + \theta_t + \varepsilon_{jt}. \quad (12)$$

Here, $\log(ES_{jt})$ are log transformed ES, w_{jt}^{hs} are the current growing season (April to September) weather conditions in the summer habitat, including growing degree days, killing degree days, and precipitation. $s_{ijt-1} w_{it-1}^{hw}$ are the past population share weighted winter habitat condition measured in the previous wintering season starting in $t - 1$. Lastly, η_j are route section fixed effects, and θ_t are year or state-by-year fixed effects, depending on the regression specification. While β_e has a structural interpretation (see equation (10)), we focus on the elasticity interpretation in the following.

We are less concerned about the endogeneity of the lagged population shares as opposed to the endogeneity of lagged population levels for two reasons. First, these individual species' shares do not indicate the bird population levels. The shares of species remain constant if all species decline similarly, due to e.g., ecosystem degradation. Therefore, the shares are uncorrelated with ES or agricultural productivity (see Section 4). Second, even if they were correlated with the outcomes or ES levels, they would be unable to predict long-term trends because our instrument is composed of many and temporarily uncorrelated NDVI shocks (see Section 4). As a precautionary measure, we additionally control for the share of migratory species i.e. $\sum_{i=1}^N s_{ijt-1}^\rho$, in all regressions in line with the suggestion of the shift-share literature (Borusyak et al., 2025). Since we use non-linear transformations of the NDVI winter habitat shocks, we use the same transformations of the shares as controls, thus isolating the variation of the shock from the variation of the shares. We discuss further instrument validity tests in Section 5.4.

Estimates from Two-Way Fixed Effects models with staggered treatments are potentially biased when treatment effects are heterogeneous (Callaway and Sant'Anna, 2021; Goodman-Bacon, 2021). While estimators for our case have not yet been developed, we are also less concerned about potential biases since observations move randomly in and out of treatment, leaving the control group unchanged on average. This is in stark contrast to the Two-Way Fixed Effects literature, which focuses on cases in which the treatment is absorbing, such that the control group changes monotonically over time.

While ES_{jt} is zero in less than 1% of observations (e.g., 0.17% for agricultural birds in corn-

producing counties or 0.92% for forest birds in forested areas), zeros are more common for the winter habitat NDVI index (e.g., 8.7% in corn-producing counties and 5% in forested areas). We therefore use inverse hyperbolic sine transformations for the NDVI index instead of logs. For all other variables, including ES, yields, and pesticides, we use log transformations and drop observations with zero ES values. Note that the second-stage outcomes are never zero in our case. Since we do not interpret the size of the coefficient of this first-stage regression but only use the prediction of ES in the second stage, we are not concerned about potential biases in the first-stage parameter estimate (Chen and Roth, 2024).

5.3 Ecosystem Services and Economic Outcomes

Here, we combine the prediction of ES from the previous section with a second stage in which we estimate the impact of ES on yields in a 2SLS approach. Taking logs of equations (8) and (9) and adding the fixed effects and controls from the previous section, the second stage estimation equation is

$$\log(Y_{jt}^*) = \beta_y \widehat{\log(ES_{jt})} + \text{lhs}(w_{jt}^{hs}) + \text{lhs}\left(\sum_{i=1}^N s_{ijt-1}^\rho\right) + \eta_j + \theta_t + \varepsilon_{jt} \quad (13)$$

The outcome variables in our main specification are aggregate crop sales and individual crop yields, adding crop fixed effects and crop times year fixed effects for the latter outcome. In addition, we present results for the three most common field crops (corn, soybeans, and wheat) and for the most common fruit crop (apples). Lastly, we estimate the same equation but with insecticide use as an outcome.

Our outcomes and the main independent variable are log-transformed. Log transformations with zeros can cause biases in the coefficient estimates if alternative transformations such as inverse hyperbolic sine transformations are used (Chen and Roth, 2024). However, our outcomes are never zero, and ES is zero in less than 1% of all cases. We omit these observations and focus on the intensive margin of our estimation (see discussion above).

We cluster standard errors at the county and BBS route levels to account for clustered sampling (Abadie et al., 2023) and the correlation of errors from potentially repeated observations of the crop outcomes across route segments within the same county. Both clusters do not necessarily coincide because routes can span several counties, while counties could include several routes. In a robustness test, we also present results with errors robust to spatial correlation (Conley, 1999), addressing both correlations simultaneously.

We suggest that the pest control provided by birds is the main mechanism linking ES to yields. To further support this mechanism, we provide evidence from forest pests. The IDS maps forest pests

from airplanes. Pests are always present, but the survey only detects pests if their population exceeds a detection threshold (\bar{Z}). We therefore estimate a linear probability model using a two-stage least squares (2SLS) approach. Our first stage is defined as above. The second stage is given by (6), but adding the same controls and fixed effects as in equation (13):

$$P(Z_{jt}^* > \bar{Z}) = \beta_f \log(\widehat{ES}_{jt}) + \text{lhs}(w_{jt}^{hs}) + \text{lhs}\left(\sum_{i=1}^N s_{ijt-1}^\rho\right) + \eta_j + \theta_t + \varepsilon_{jt} \quad (14)$$

Here, $P(Z_{jt}^* > \bar{Z})$ measures the probability that the pest stock exceeds the detection threshold, w_{jt}^{hs} are growing season weather controls including growing degree days, killing degree days, and precipitation in the summer habitats. The term $\text{lhs}\left(\sum_{i=1}^N s_{ijt-1}^\rho\right)$ is the population share control and η_j and θ_t are route segment and state by year fixed effects. We estimate the equation for moths and bark beetles separately.

In a robustness test, we estimate a two-stage residual inclusion logit model (Terza et al., 2008; Wooldridge, 2015). While this approach is more appropriate for binary outcomes, it drops many observations with no variation in the outcome, which can introduce selection bias and complicate inference when using instrumental variables with many fixed effects. For these reasons, we present the linear probability model as our preferred specification and report the logit results in the appendix.

5.4 Instrument Validity

Our instrumental variable combines plausibly exogenous variation in winter habitat conditions with the share of long-distance migratory birds observed in summer habitats. As discussed in the previous section, shocks to winter habitats consist of multiple, imperfectly correlated disturbances across species and over time (see Figure 3). We argue that, conditional on state-by-year fixed effects, these shocks satisfy the criteria for a valid shift component under the “many shocks” condition outlined in Borusyak et al. (2025).

The second component of the instrument, the share of long-distance migratory birds, serves as the exposure variable. We argue that this share is additionally exogenous with respect to the outcomes of interest. Figures 4 and 5 provide visual evidence that the spatial distribution of migratory bird shares is uncorrelated with crop yields. We formalize this intuition by estimating a regression of log crop yields and sales in year t on the share of long-distance migratory birds in year $t+1$. This approach tests whether the instrument’s exposure term is correlated with outcomes through channels other than the winter habitat shocks. Importantly, we use the share of migratory birds in $t+1$ to construct this placebo test: if ecosystem service (ES) shocks in winter habitats affect the migratory bird share, then future bird shares should not influence current outcomes unless there is reverse causality or

omitted variable bias. The results of this placebo regression, presented in Table 1, show no significant association between crop yields or sales in year t and the share of long-distance migratory birds in year $t + 1$, conditional on the same set of fixed effects employed in our main specifications. This supports the validity of the exclusion restriction underlying our instrument.

Table 1: Instrument Validity: Current Yields and Long-Distance Migrant Shares

	Log(Sales)	Log(Yield)
	(1)	(2)
Share of Long-Distance Migrants ($t+1$)	0.017 (0.017)	-0.012 (0.016)
Route FE	Yes	Yes
Year-State FE	Yes	Yes
Year-item FE		Yes
Observations	44,093	143,373

Notes: Standard errors are clustered at the county and route level. *, **, and *** indicate significance at the 10%, 5%, and 1% level. All specifications control for ihs transformed growing degree days, killing degree days, and precipitation in the current year.

6 Results

We start by presenting our results on the impact of South American winter habitat conditions on the return of migratory species and the ecosystem services that they provide. We then report our main results on the impact of ecosystem services on agricultural yields and sales. We discuss the elasticity of substitution implied by our results. Lastly, we support these main results with evidence on the impact of ES on pesticide use and pest outbreaks in forests.

6.1 Winter Habitat Conditions and the Return of Migratory Birds

Here we present our results on the impact of winter habitat conditions on the return of individual migratory species to their summer habitats. We start with the species-level result. In this specification, we compare the return of long-distance migrants to their summer habitats across species that are more or less exposed to winter habitat shocks within a given year or year-state combination. The results show that a higher NDVI in the winter habitats during the species-specific wintering period increases the abundance of that species in the next year's summer habitats (Table 2). As predicted by our model, more individuals of a species in the previous year increase the population in the current year through a reproduction effect. In contrast, more individuals across all species reduce the abundance of a species in the current year through a competition effect (see Section 2). Dropping both lagged population terms increases the NDVI estimates. The high number of observations stems from the fact

that the observations are at the species-route segment level, rather than the route segment level, as is the case for the rest of the analysis.

Table 2: First Stage Results: Winter Habitat Conditions and the Return of Migratory Birds

	Ihs(Species Abundance)			
	(1)	(2)	(3)	(4)
Ihs(NDVI)	0.046*** (0.013)	0.046*** (0.013)	0.143*** (0.020)	0.143*** (0.020)
Lag Total Abundance	-0.068*** (0.006)	-0.071*** (0.006)		
Ihs(Lag Species Abundance)	0.719*** (0.002)	0.719*** (0.002)		
Route-Segment FE	Yes	Yes	Yes	Yes
Species FE	Yes	Yes	Yes	Yes
Year FE	Yes		Yes	
Year-State FE		Yes		Yes
Observations	9,991,720	9,991,720	12,125,420	12,125,420

Notes: ‘Ihs(Lag Species Abundance)’ is the ihs transformed abundance of species i in year $t - 1$. ‘Lag Total Abundance’ is the total number of individuals across all species expressed in 1000 individuals (total number of individuals/1000) in year $t-1$. ‘Ihs(NDVI)’ is the ihs transformed winter habitat NDVI. Standard errors are clustered at the route level. *, **, and *** indicate significance at the 10%, 5%, and 1% level. All specifications control ihs transformed growing degree days, killing degree days and precipitation in the current year.

6.2 Winter Habitat Conditions and Ecosystem Services

We present our results on the impact of winter habitat conditions on the return of migratory species and the ecosystem services they provide (Table 3). In this estimation, we compare bird communities that are more or less affected by migratory bird shocks within the same year and state. While the previous results only included migratory birds, these results report the impact of a migratory bird shock on the ES provided by all insectivorous birds (insectivores) within an ecosystem. Specifically, we restrict our ES measure to all insectivorous birds (columns 1 to 2), insectivorous birds on farm-land (columns 3 to 4), and insectivorous forest birds (columns 5 to 6). We restrict the sample in the last four columns to farm and forest birds because we use this restricted sample in our second stage for agricultural and forest outcomes, respectively. Further, the estimates are based on different assumptions about the elasticities of substitution among species to calculate the ES. We report the underlying elasticities in the second row of headers. Here, we use elasticities of 2 and 10 as they reflect the range often used in the economic literature. However, we discuss the impact of the underlying elasticities for our main results across a wider range of elasticities in the next section. Overall, the impact of a positive NDVI shock in the South American winter habitats leads to more provision of ES during the following growing season in the North American summer habitats. The estimates

vary non-systematically across different sample restrictions and elasticities. Again, we refrain from a quantitative interpretation because of potential biases from the ihs transformation.

Table 3: First Stage Results: Winter Habitat Conditions and the Provision of Ecosystem Services

Sample	Log(ES)					
	(1) All Insectivorous Birds		(3) Insectivorous Farm Birds		(5) Insectivorous Forest Birds	
	2	10	2	10	2	10
Ihs(NDVI Index)	0.014*** (0.002)	0.009*** (0.001)	0.018*** (0.001)	0.012*** (0.0009)	0.015*** (0.002)	0.018*** (0.001)
Route-Segment FE	Yes	Yes	Yes	Yes	Yes	Yes
Year-State FE	Yes	Yes	Yes	Yes	Yes	Yes
Observations	314,524	314,524	311,791	311,791	288,830	288,830

Notes: ‘Ihs(NDVI Index)’ is the winter habitat NDVI index defined by equation (12). Standard errors are clustered at the route level. *, **, and *** indicate significance at the 10%, 5%, and 1% level. All specifications control for the share of migratory birds, $Ihs\left(\sum_{i=1}^N s_{ijt-1}^\rho\right)$ and ihs transformed growing degree days, killing degree days and precipitation in the current year.

The different number of observations comes from the fact that the outcome variable is log-transformed. About 0.9% of our observations have no farmland species, and 8% have no forest species, such that the ES variable is zero. This restriction is, however, irrelevant for our second stage because we only observe agricultural outcomes on farmland and forest outcomes in the forest landscape, where all observations contain farmland and forest species, respectively.

6.3 Ecosystem Services and Agricultural Outcomes

Here we present our main results on the impact of ES on agricultural outcomes using the first-stage from the previous section to predict ES using NDVI in the winter habitats. While we use shocks to long-distance migrants as an instrument, the ES measure combines insectivorous migrants and non-migrant farmland birds. We present results for log-transformed aggregate crop sales (columns 1 to 4) and simultaneously estimated individual log crop yields (Columns 5 to 8). The columns within the same outcome differ in the underlying elasticity of substitution among species and the level of fixed effects. Here, we present results for elasticities of 2 and 10, but we discuss the impact of the whole range of possible elasticities in the next subsection. We use route segment fixed effects in all specifications, and additionally, year fixed effects for Log(Sales) and year-by-crop fixed effects when using Log(Yield) as an outcome. In a robustness test, we additionally include state-by-year fixed effects in columns (3) to (4) and (7) to (8). The first stage is generally strong, with a Kleibergen-Paap Wald test between 42 and 142.

The results suggest that a 10 percent increase in ES provided by insectivorous birds increases Log(Sales) by 0.7 to 1.4 percent, depending on the assumed elasticity of substitution among species and the fixed effects. Similarly, a 10 percent increase in ES provided by insectivorous birds increases Log(Yields) by 0.6 to 1.7 percent, again depending on the assumed elasticities and fixed effects. We will discuss the elasticities below as well as heterogeneity across crop yields.

We provide the OLS results in Appendix A.11. However, since agricultural development is the primary driver of biodiversity decline, the results should be interpreted with caution.

Table 4: Main Results: Ecosystem Services and Agricultural Outcomes

Outcome Elasticity	Log(Sales)				Log(Yield)			
	2	10	2	10	2	10	2	10
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Log(ES)	0.095** (0.037)	0.143** (0.065)	0.070* (0.037)	0.113* (0.066)	0.082* (0.044)	0.171** (0.080)	0.058 (0.043)	0.149* (0.082)
Route-Segment FE	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Year FE	Yes	Yes						
Year-State FE			Yes	Yes			Yes	Yes
Crop-Year FE					Yes	Yes		
Crop FE							Yes	Yes
Observations	62,917	62,917	62,917	62,917	241,638	241,638	241,638	241,638
Kleibergen-Paap	142.0	94.6	119.1	76.3	97.3	58.0	76.3	41.5

Notes: Standard errors are clustered at the county and route level. *, **, and *** indicate significance at the 10%, 5%, and 1% level. All specifications control for the share of migratory birds, $Ihs\left(\sum_{i=1}^N s_{ijt-1}^\rho\right)$, and lhs transformed growing degree days, killing degree days and precipitation in the current year. ‘Kleibergen-Paap’ is the first stage Kleibergen-Paap Wald Statistic.

The number of observations differs from Table 3 and varies across columns in Table 4 for several reasons. First, agricultural outcomes are only observed in the eight census years that fall within our study period. Second, the yield regressions are estimated at the crop-segment-year level, whereas the sales regressions are estimated at the segment-year level. Moreover, yield data are only available for crops that are actively produced in a given county and year. As a result, the total number of observations in the yield regressions does not equal the number of sales observations multiplied by the number of crops in our sample. The results using spatially robust standard errors (Conley, 1999) are presented in Appendix A.12. The statistical significance of the estimates remains mostly unchanged.

6.4 Elasticity of Substitution Among Species in Agriculture

Across all specifications reported in Table 4, the estimated effect of ES on crop production increases with the assumed elasticity of substitution among species. Because the estimated coefficients are themselves elasticities, they are invariant with respect to the scale of the ES measure. Differences in

coefficient magnitudes, therefore, reflect differences in how responsive crop production is to the various ways ES are quantified. Larger estimates under low elasticity of substitution would suggest that species identity (i.e., species richness) matters more than abundance. Conversely, larger estimates under high elasticity assumptions would imply that species are more substitutable and that total abundance plays a greater role. An alternative but complementary interpretation is that a true, underlying measure of biodiversity drives agricultural outcomes, while each ES measure serves as a noisy proxy for this latent variable. In this view, higher coefficient magnitudes indicate better approximations of the true measure, while poorer proxies attenuate the estimated effects toward zero (Wooldridge, 2010, Ch. 4).

The results presented in Figure 11 suggest that the true elasticity of substitution is high. The estimated impact of ES on crop production generally increases with the underlying elasticity of substitution among species in the ES measure. In the limiting case of an infinite elasticity of substitution, species identity becomes irrelevant, and the ES measure reduces to total bird abundance across all species. Estimates are generally highest under this measure. We present results with Conley standard errors in Appendix A.12. The statistical significance of the estimates remains, however, largely unchanged.

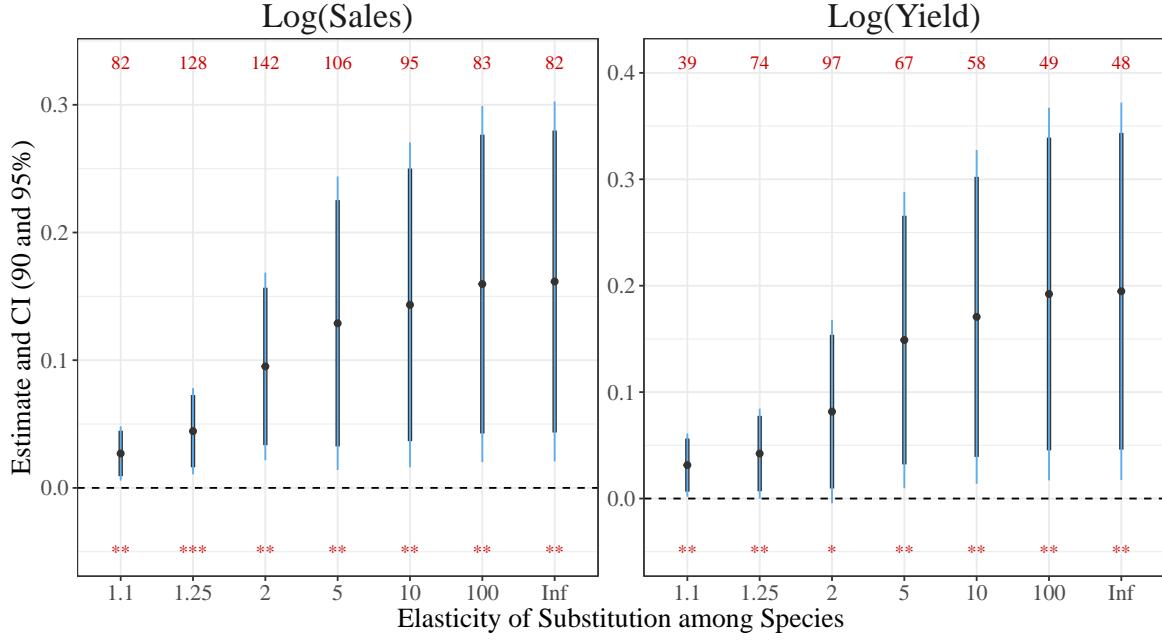
6.5 Heterogeneity Across Crops

Crops may respond differently to ecosystem services (ES) for several reasons. First, many field crops, especially corn, are genetically modified to possess intrinsic resistance to insect pests, reducing their reliance on natural pest control (Noack et al., 2024). Second, crops like corn, soybeans, and wheat typically require less insecticide use than fruits and vegetables (Fernandez-Cornejo et al., 2014), and may therefore exhibit weaker responses to changes in pest pressure. Third, some crops, most notably fruits such as apples, but also field crops like soybeans, depend partially or fully on insect pollination (Garibaldi et al., 2021). Insectivorous birds may inadvertently reduce pollinator populations by preying on insects, potentially offsetting the benefits of pest control. Fourth, fruit orchards often provide direct habitat for insectivorous birds, making these systems more exposed to changes in avian-provided ES.

Given these factors, we expect the strongest responses to bird-provided ES from non-genetically modified crops such as wheat and from pollination-dependent fruits such as apples, and the weakest responses from genetically modified crops such as corn and from field crops like soybeans.

Table 5 presents the crop-level regression estimates. We focus on the high elasticity case (elasticity = 10) based on the discussion in the previous section. We find the largest effects of ES on yields for wheat and apples, though the apple estimate is not statistically significant. Soybeans and corn exhibit small, statistically insignificant responses to ecosystem services (ES), consistent with our expectations.

Figure 6: Agricultural Outcomes and the Elasticity of Substitution among Species.



Notes: Point estimates (black dots), 90% confidence intervals (black lines), 95% confidence intervals (blue lines), p-values (red asterisks at the bottom), and first-stage Kleibergen-Paap Wald statistic (red numbers on top) for our main regression results based on equation (13) and different underlying elasticities of substitution among species (x-axis). An infinite elasticity of substitution is equivalent to the total abundance of individuals across all species. All specifications control for the share of migratory birds, $Ihs\left(\sum_{i=1}^N s_{ijt-1}^o\right)$, and ihs transformed growing degree days, killing degree days, and precipitation in the current year. Additionally, all specifications include route segment and year fixed effects in the Sales regression and route segment and crop-by-year fixed effects in the Yield regressions. Standard errors are clustered at the county and route levels. *, **, and *** indicate significance at the 10%, 5%, and 1% level.

However, the weakness of the first-stage regressions limits further interpretation of these results. This is especially true for apples, which are only produced in a limited number of countries and for which data are only recorded for some census years (see Section 4). We provide the OLS results in Appendix A.13.

Table 5: Ecosystem Services and Crop-Level Yields

	Log(Yield)							
	Corn		Wheat		Soybean		Apples	
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Log(ES)	-0.110 (0.169)	-0.041 (0.179)	0.165* (0.091)	0.139* (0.083)	-0.015 (0.162)	-0.103 (0.155)	0.855 (0.983)	1.41 (1.00)
Route-Segment FE	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Year FE	Yes		Yes		Yes		Yes	
Year-State FE		Yes		Yes		Yes		Yes
Observations	45,123	45,123	43,957	43,957	31,979	31,979	8,109	8,109
Kleibergen-Paap	31.0	20.5	46.0	37.5	21.6	15.5	2.08	3.00

Notes: Standard errors are clustered at the county and route level. *, **, and *** indicate significance at the 10%, 5%, and 1% level. All specifications control for the share of migratory birds, $Ihs\left(\sum_{i=1}^N s_{ijt-1}^\rho\right)$, and ihs transformed growing degree days, killing degree days, and precipitation in the current year.

‘Kleibergen-Paap’ is the first stage Kleibergen-Paap Wald Statistic.

6.6 Ecosystem Services and Insecticide Use

Pesticides, particularly insecticides, are a technological substitute for the ecosystem services (ES) provided by insectivorous birds. While prior studies document increased pesticide use following the decline of insectivorous bats (Frank, 2024), our framework allows for complementarity between pesticides and avian ES (e.g., equation (8)), especially when ES and pesticides target different pests. In this case, greater ES can raise the marginal benefit of pesticides by shifting the marginal yield curve upward. We test the effect of ES on insecticide use using 2SLS regressions with year-by-state fixed effects, estimating models under high and low elasticities of substitution among species. While the results suggest that more ES tends to reduce insecticide use, the estimates are generally imprecise (Table 6). We provide the OLS results in Appendix A.14.

Table 6: Ecosystem Services and Insecticide Use

Elasticity	Log(Insecticide Use)			
	2		10	
	(1)	(2)	(3)	(4)
Log(ES)	-0.139 (0.096)	-0.155 (0.170)	-0.113 (0.091)	-0.166 (0.161)
Route-Segment FE	Yes	Yes	Yes	Yes
Year FE	Yes	Yes		
Year-State FE			Yes	Yes
Observations	218,757	218,757	218,757	218,757
Kleibergen-Paap	165.4	106.6	152.5	95.2

Notes: Standard errors are clustered at the county and route level. *, **, and *** indicate significance at the 10%, 5%, and 1% level. All specifications control for the share of migratory birds Ihs ($\sum_{i=1}^N s_{ijt-1}^\rho$) and ihs transformed growing degree days, killing degree days and precipitation in the current year.
 ‘Kleibergen-Paap’ is the first stage Kleibergen-Paap Wald Statistic.

6.7 Ecosystem Services and Forest Pests

So far, our results suggest that more ES provided by insectivorous birds increases yields and weakly reduces pesticide use. The main mechanism suggested by the theory section is the reduction of pest populations. While we cannot observe pest populations in agriculture, here we provide evidence from pest outbreaks in forests. Table 7 reports the results from a 2SLS linear probability model for different elasticities of substitution and with different levels of fixed effects. We find that ES provided by insectivorous birds reduce the outbreak of moth species such as spongy moths or spruce budworms, but has no or a weakly positive but imprecisely estimated impact on bark beetles such as the mountain pine beetles. In the next section, we discuss the impact of the underlying elasticities of substitution among species on the results.

Table 7: Pest Outbreaks and Ecosystem Services

Pest Elasticity	Bark Beetle				Moth			
	2	10	2	10	2	10	2	10
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Log(ES)	0.059 (0.062)	0.077 (0.079)	0.057 (0.060)	0.088 (0.077)	-0.069** (0.032)	-0.103** (0.043)	-0.091*** (0.034)	-0.115*** (0.044)
Route-Segment FE	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Year FE	Yes	Yes			Yes	Yes		
Year-State FE			Yes	Yes			Yes	Yes
Observations	143,452	143,452	143,452	143,452	143,452	143,452	143,452	143,452
Dependent variable mean	0.123	0.123	0.123	0.123	0.062	0.062	0.062	0.062
Kleibergen-Paap	22.9	34.5	22.7	32.7	22.9	34.5	22.7	32.7

Notes: Standard errors are clustered at the county and route level. *, **, and *** indicate significance at the 10%, 5%, and 1% level. All specifications control for the share of migratory birds Ihs ($\sum_{i=1}^N s_{ijt-1}^\rho$) and ihs transformed growing degree days, killing degree days and precipitation in the current year.
 ‘Kleibergen-Paap’ is the first stage Kleibergen-Paap Wald Statistic.

We present the OLS results as well as 2SLS results for different bands in which we record forest pests on both sides of the routes in the Appendix A.15 and A.16, respectively. The estimates for 10 km bandwidth are slightly larger but otherwise similar; the results for 1km bandwidth have low precision, possibly due to the strongly reduced number of positive pest detections in the smaller bandwidth. Further, we report the results using a two-stage residual inclusion logit model in Appendix A.17. While the size of the marginal effects is not directly comparable, the results are qualitatively similar.

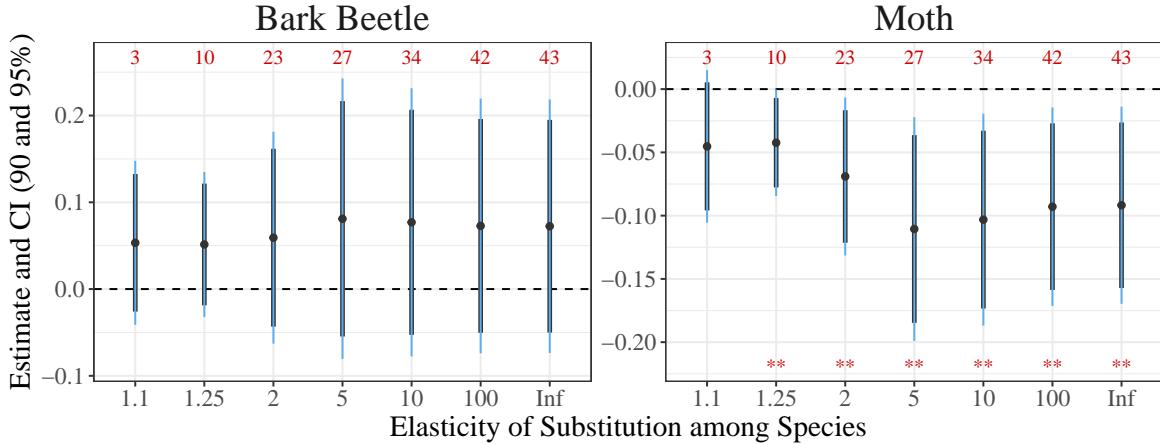
6.8 Elasticity of Substitution Among Species in Forests

Here we present results on the impact of the underlying elasticity of substitution among insectivorous bird species on the probability of pest outbreak. Similar to the previous sections, we interpret the magnitude of the estimate as indicative of the true elasticity of substitution. Unlike our results on agriculture, Figure 7 suggests that an ES measure based on intermediate elasticities of substitution explains more variation in forest pest outbreaks. In other words, complementarities among species are possibly more important in semi-natural ecosystems such as forests compared to highly transformed ecosystems such as agriculture.

6.9 Heterogeneous Response of Forest Pests and Species Interactions

Our results indicate that moths and bark beetles respond differently to ecosystem services (ES) provided by birds. While increased ES reduces the probability of moth outbreaks, it has a weakly positive effect on bark beetles. These contrasting effects can be explained by differences in the species’ ecology.

Figure 7: Forest Pest Outbreaks, Ecosystem Services and the Elasticity of Substitution among Species.



Notes: Point estimates (black dots), 90% confidence intervals (black lines), 95% confidence intervals (blue lines), p-values (red asterisks at the bottom), and first-stage Kleibergen-Paap Wald statistic (red numbers on top) for our main regression results based on equation (14) and different underlying elasticities of substitution among species (x-axis). An infinite elasticity of substitution is equivalent to the total abundance of individuals across all species. Estimates represent elasticities. All specifications control for the share of migratory birds, Ihs ($\sum_{i=1}^N s_{ijt-1}^\rho$), and ihs transformed growing degree days, killing degree days, and precipitation in the current year. Additionally, all specifications include year and route-segment fixed effects. Standard errors are clustered at the county and route levels. *, **, and *** indicate significance at the 10%, 5%, and 1% level. Additionally, the yield regression includes crop-by-year fixed effects.

Moths are flying insects whose larvae (caterpillars) are often exposed and accessible to birds, making birds important predators of both life stages. In contrast, bark beetles spend most of their life cycle, both as adults and larvae, under the bark, where they are largely inaccessible to birds.⁶ Instead, their primary natural enemies are specialized predators, most notably clerid beetles such as the American bark beetle destroyer (*Thanasimus dubius*), which can substantially reduce bark beetle populations (Reeve, 1997). However, these clerid beetles are themselves preyed upon by birds, potentially explaining the observed weakly positive effect of bird-provided ES on bark beetle outbreaks. This qualitative example illustrates Proposition 1, which suggests that increases of beneficial species do not necessarily increase the provision of ES. Instead, these increases may reduce ES if the negative effect on other beneficial species outweighs their direct benefits.

7 Discussion and Conclusion

This paper investigates the role of biodiversity in economic production, with a focus on the ecosystem services (ES) provided by insectivorous birds. Conceptually, we contribute to the understanding of

⁶An exception is woodpeckers. However, woodpeckers are non-migratory and are therefore not captured by our instrument.

how species diversity supports ecosystem functioning. Empirically, we address the inherent identification challenge posed by the endogeneity between biodiversity and economic activity, specifically, how economic production contributes to biodiversity loss and vice versa. To overcome this, we leverage ecological shocks to South American winter habitats and the migratory pattern of birds to causally estimate the effect of biodiversity on ES provision and economic outcomes in North America.

Our findings demonstrate that shocks to birds' winter habitats in South America during the northern winter influence their return to North American summer habitats. These migratory disruptions predict variation in the provision of ES by insectivorous birds. Exploiting differences in bird migration across otherwise similar areas within U.S. states, we estimate that a 10% increase in bird-provided ES leads to a 1.1% increase in crop sales.

Over the past 50 years, North American bird populations have declined by approximately 30% on average, with likely greater declines in agricultural landscapes (Rosenberg et al., 2019). Suppose ES provision scales linearly with bird abundance, a plausible assumption given the homogeneity of degree one in the ES production function. In that case, this decline implies a 30% reduction in ES leading to 4% decline in agricultural sales based on our preferred estimate.⁷ Given total U.S. crop production valued at \$281 billion (USDA, 2024), this corresponds to annual losses of approximately \$11.1 billion.

Beyond agriculture, we find that birds also enhance forest productivity by reducing the likelihood of pest outbreaks. Specifically, a 10% increase in ES from birds is associated with a 1.15% decline in the probability of moth-dominated pest outbreaks. A 30% decline in bird populations thus implies a 4.2% increase in outbreak risk. While the economic costs of forest pest damage, including timber loss and property devaluation, are substantial (Aukema et al., 2011; Druckenmiller, 2020), precise quantification is hindered by species-specific heterogeneity and long-term forest dynamics.

Our results further contribute to the debate about the value of diversity and the substitutability among species. The impact of biodiversity on crop yields increases with the elasticity of substitution among bird species, suggesting high functional redundancy in agricultural landscapes. In contrast, forest ecosystems exhibit stronger complementarities: pest suppression benefits peak at intermediate levels of substitutability. These findings imply that conservation strategies should be tailored to ecosystem context. In agriculture, prioritizing species that are both efficient ES providers and cost-effective to conserve (i.e., low marginal conservation cost per unit of ES) may yield higher returns. In forests, maintaining species diversity itself may be essential due to limited substitutability.

Several limitations merit discussion. First, we focus on a small group of species and ES. Complementarities may be larger for different ES or across different groups of species, as illustrated by our examples of bark beetles, clerid beetles, and birds, and discussed more generally in Giglio et al.

⁷Calculated as $\left[\left(\frac{ES_1}{ES_0}\right)^{\beta_y} - 1\right] \times 100$ with $ES_1 = 0.7ES_0$.

(2024). Second, while this paper focuses on ES provision, there are important biodiversity values that were not captured by our study. Biodiversity provides important information that can be used, e.g., in technology development such as the development of new drugs or crop varieties (Weitzman, 1992). Focusing on these values prioritizes distinct species containing unique information (Weitzman, 1998) and not necessarily on those providing high levels of ES. Second, we have ignored the contribution of biodiversity to risk reduction (Quaas and Baumgärtner, 2008). In relation to ES, this could be the availability of redundant species that can take over important ecosystem roles of declining or extinct species (Lehman and Tilman, 2000).

Finally, the paper has broader policy implications. Our identification strategy, which exploits habitat shocks in South America, underscores the cross-border dependencies in biodiversity conservation. Environmental degradation in one region can propagate economic costs elsewhere through migratory species. This finding highlights the need for international coordination in conservation policy to ensure effective protection of biodiversity and the ecosystem services it supports.

Agriculture has changed dramatically since Henshaw's statement about the importance of birds for agricultural production. Bird populations have collapsed in the agricultural landscape while pesticide use has risen dramatically as a man-made substitute for biological pest control. Recent studies on the health impacts of pesticides caution that these substitutions came at a high cost (Dias et al., 2023; Taylor, 2021; Frank, 2024; Reynier and Rubin, 2025). While even Henshaw acknowledges the limitation of birds as pest control (Henshaw, 1907), they provide a further tool in the arsenal to control pest damage while avoiding the health consequences of pesticide use.

References

- Abadie, Alberto, Susan Athey, Guido W Imbens, and Jeffrey M Wooldridge**, “When should you adjust standard errors for clustering?,” *The Quarterly Journal of Economics*, 2023, 138 (1), 1–35.
- Alix-Garcia, Jennifer M, Katharine RE Sims, and Patricia Yañez-Pagans**, “Only one tree from each seed? Environmental effectiveness and poverty alleviation in Mexico’s payments for ecosystem services program,” *American Economic Journal: Economic Policy*, 2015, 7 (4), 1–40.
- Ambrosini, Roberto, Anna Maria Bolzern, Luca Canova, Silvia Arieni, Anders Pape Møller, and Nicola Saino**, “The distribution and colony size of barn swallows in relation to agricultural land use,” *Journal of Applied Ecology*, 2002, 39 (3), 524–534.
- Andam, Kwaw S, Paul J Ferraro, Alexander Pfaff, G Arturo Sanchez-Azofeifa, and Juan A Robalino**, “Measuring the effectiveness of protected area networks in reducing deforestation,” *Proceedings of the National Academy of Sciences*, 2008, 105 (42), 16089–16094.
- Assunção, Juliano, Robert McMillan, Joshua Murphy, and Eduardo Souza-Rodrigues**, “Optimal environmental targeting in the amazon rainforest,” *The Review of Economic Studies*, 2023, 90 (4), 1608–1641.
- Aukema, Juliann E, Brian Leung, Kent Kovacs, Corey Chivers, Kerry O Britton, Jeffrey Englin, Susan J Frankel, Robert G Haight, Thomas P Holmes, Andrew M Liebhold et al.**, “Economic impacts of non-native forest insects in the continental United States,” *PLoS One*, 2011, 6 (9), e24587.
- Bertram, Christine and Martin F Quaas**, “Biodiversity and optimal multi-species ecosystem management,” *Environmental and Resource Economics*, 2017, 67, 321–350.
- BirdLife**, “BirdLife International and Handbook of the Birds of the World,” *Bird species distribution maps of the world*, 2018.
- Borusyak, Kirill, Peter Hull, and Xavier Jaravel**, “A practical guide to shift-share instruments,” *Journal of Economic Perspectives*, 2025, 39 (1), 181–204.
- Brock, William A and Anastasios Xepapadeas**, “Valuing biodiversity from an economic perspective: a unified economic, ecological, and genetic approach,” *American Economic Review*, 2003, 93 (5), 1597–1614.

Burke, Marshall and David B Lobell, “Satellite-based assessment of yield variation and its determinants in smallholder African systems,” *Proceedings of the National Academy of Sciences*, 2017, 114 (9), 2189–2194.

Callaway, Brantly and Pedro HC Sant’Anna, “Difference-in-differences with multiple time periods,” *Journal of Econometrics*, 2021, 225 (2), 200–230.

Cardinale, Bradley J, J Emmett Duffy, Andrew Gonzalez, David U Hooper, Charles Perrings, Patrick Venail, Anita Narwani, Georgina M Mace, David Tilman, David A Wardle et al., “Biodiversity loss and its impact on humanity,” *Nature*, 2012, 486 (7401), 59–67.

CCI, ESA Land Cover, “Product user guide version 2.0,” *UCL-Geomatics: London, UK*, 2017, 685.

Chen, Jiafeng and Jonathan Roth, “Logs with zeros? Some problems and solutions,” *The Quarterly Journal of Economics*, 2024, 139 (2), 891–936.

Conley, TG, “GMM estimation with cross sectional dependence,” *Journal of Econometrics*, 1999, 92 (1), 1–45.

Cooper, Nathan Wands, Scott W Yanco, Clark S Rushing, T Scott Sillett, and Peter P Marra, “Winter and Summer Conditions Induce Carry-Over Effect on Survival of Migratory Birds,” Available at SSRN 4818765, 2024.

Dasgupta, P, *The Economics of Biodiversity: the Dasgupta Review.*, HM Treasury, 2021.

Dee, Laura E, Paul J Ferraro, Christopher N Severen, Kaitlin A Kimmel, Elizabeth T Borer, Jarrett EK Byrnes, Adam Thomas Clark, Yann Hautier, Andrew Hector, Xavier Raynaud et al., “Clarifying the effect of biodiversity on productivity in natural ecosystems with longitudinal data and methods for causal inference,” *Nature Communications*, 2023, 14 (1), 2607.

Dias, Mateus, Rudi Rocha, and Rodrigo R Soares, “Down the river: glyphosate use in agriculture and birth outcomes of surrounding populations,” *Review of Economic Studies*, 2023, 90 (6), 2943–2981.

Dossman, Bryant C, Colin E Studds, Shannon L LaDeau, T Scott Sillett, and Peter P Marra, “The role of tropical rainfall in driving range dynamics for a long-distance migratory bird,” *Proceedings of the National Academy of Sciences*, 2023, 120 (52), e2301055120.

Druckenmiller, Hannah, “Estimating an Economic and Social Value for Healthy Forests: Evidence from Tree Mortality in the American West,” Technical Report, Working Paper 2020.

Durre, Imke, Anthony Arguez, Carl J Schreck III, Michael F Squires, and Russell S Vose, “Daily high-resolution temperature and precipitation fields for the contiguous United States from 1951 to present,” *Journal of Atmospheric and Oceanic Technology*, 2022, 39 (12), 1837–1855.

Engist, Dennis, Laura Melissa Guzman, Ashley Larsen, Trevor Church, and Frederik Noack, “The impact of genetically modified crops on bird diversity,” *Nature Sustainability*, 2024, 7 (9), 1149–1159.

Falco, Salvatore Di and Jean-Paul Chavas, “On crop biodiversity, risk exposure, and food security in the highlands of Ethiopia,” *American Journal of Agricultural Economics*, 2009, 91 (3), 599–611.

FAO, *FAO’s Plant Production and Protection Division*, FAO Rome, Italy, 2022.

Fenichel, Eli P, Monica F Dean, and Oswald J Schmitz, “The path to scientifically sound biodiversity valuation in the context of the Global Biodiversity Framework,” *Proceedings of the National Academy of Sciences*, 2024, 121 (34), e2319077121.

Fernandez-Cornejo, Jorge, Richard F Nehring, Craig Osteen, Seth Wechsler, Andrew Martin, and Alex Vialou, “Pesticide use in US agriculture: 21 selected crops, 1960-2008,” *USDA-ERS Economic Information Bulletin*, 2014, (124).

Ferraro, Paul J, Craig McIntosh, and Monica Ospina, “The effectiveness of the US endangered species act: An econometric analysis using matching methods,” *Journal of Environmental Economics and Management*, 2007, 54 (3), 245–261.

Frank, Eyal and Anant Sudarshan, “The social costs of keystone species collapse: Evidence from the decline of vultures in india,” *American Economic Review*, 2024, 114 (10), 3007–3040.

Frank, Eyal G, “The economic impacts of ecosystem disruptions: Costs from substituting biological pest control,” *Science*, 2024, 385 (6713), eadg0344.

García, Daniel, Marcos Miñarro, and Rodrigo Martínez-Sastre, “Enhancing ecosystem services in apple orchards: Nest boxes increase pest control by insectivorous birds,” *Journal of Applied Ecology*, 2021, 58 (3), 465–475.

Garibaldi, Lucas A, Lisa A Schulte, Diego N Nabaes Jodar, Dulce S Gomez Carella, and Claire Kremen, “Time to integrate pollinator science into soybean production,” *Trends in Ecology & Evolution*, 2021, 36 (7), 573–575.

Giglio, Stefano, Theresa Kuchler, Johannes Stroebel, and Olivier Wang, “The Economics of Biodiversity Loss,” Technical Report, National Bureau of Economic Research 2024.

Goodman-Bacon, Andrew, “Difference-in-differences with variation in treatment timing,” *Journal of Econometrics*, 2021, 225 (2), 254–277.

Henshaw, Henry Wetherbee, *Value of Swallows as Insect Destroyers* number 56, US Government Printing Office, 1907.

Hill, Mark O, “Diversity and evenness: a unifying notation and its consequences,” *Ecology*, 1973, 54 (2), 427–432.

Hsiang, Solomon and Nitin Sekar, “Does legalization reduce black market activity? Evidence from a global ivory experiment and elephant poaching data,” Technical Report, National Bureau of Economic Research 2016.

Iverson, Autumn R, Diana L Humble, Renée L Cormier, Thomas P Hahn, Theadora A Block, Daizaburo Shizuka, Bruce E Lyon, Alexis S Chaine, Emily J Hudson, and Elisha M Hull, “Winter GPS tagging reveals home ranges during the breeding season for a boreal-nesting migrant songbird, the Golden-crowned Sparrow,” *Plos one*, 2024, 19 (6), e0305369.

Jayachandran, Seema, Joost De Laat, Eric F Lambin, Charlotte Y Stanton, Robin Audy, and Nancy E Thomas, “Cash for carbon: A randomized trial of payments for ecosystem services to reduce deforestation,” *Science*, 2017, 357 (6348), 267–273.

Kremen, Claire, “Managing ecosystem services: what do we need to know about their ecology?,” *Ecology Letters*, 2005, 8 (5), 468–479.

Langpap, Christian, Joe Kerkvliet, and Jason F Shogren, “The economics of the US Endangered Species Act: A review of recent developments,” *Review of Environmental Economics and Policy*, 2018.

Lehman, Clarence L and David Tilman, “Biodiversity, stability, and productivity in competitive communities,” *The American Naturalist*, 2000, 156 (5), 534–552.

Li, Yijia, Ruiqing Miao, and Madhu Khanna, “Neonicotinoids and decline in bird biodiversity in the United States,” *Nature Sustainability*, 2020, 3 (12), 1027–1035.

Liang, Yuanning, Ivan J Rudik, and Eric Zou, “The environmental effects of economic production: Evidence from ecological observations,” Technical Report, National Bureau of Economic Research 2021.

Lichtenberg, Erik and David Zilberman, “The econometrics of damage control: why specification matters,” *American Journal of Agricultural Economics*, 1986, 68 (2), 261–273.

Liu, Xianfeng, Gaopeng Sun, Zheng Fu, Philippe Ciais, Xiaoming Feng, Jing Li, and Bojie Fu, “Compound droughts slow down the greening of the Earth,” *Global Change Biology*, 2023, 29 (11), 3072–3084.

Loreau, Michel and Andy Hector, “Partitioning selection and complementarity in biodiversity experiments,” *Nature*, 2001, 412 (6842), 72–76.

Lovett, Gary M, Marissa Weiss, Andrew M Liebhold, Thomas P Holmes, Brian Leung, Kathy Fallon Lambert, David A Orwig, Faith T Campbell, Jonathan Rosenthal, Deborah G McCullough et al., “Nonnative forest insects and pathogens in the United States: Impacts and policy options,” *Ecological Applications*, 2016, 26 (5), 1437–1455.

Luck, Gary W, Richard Harrington, Paula A Harrison, Claire Kremen, Pam M Berry, Rob Bugter, Terence P Dawson, Francesco De Bello, Sandra Díaz, Christian K Feld et al., “Quantifying the contribution of organisms to the provision of ecosystem services,” *Bio-science*, 2009, 59 (3), 223–235.

Lueck, Dean and Jeffrey A Michael, “Preemptive habitat destruction under the Endangered Species Act,” *The Journal of Law and Economics*, 2003, 46 (1), 27–60.

Marini, Lorenzo, Matthew P Ayres, and Hervé Jactel, “Impact of stand and landscape management on forest pest damage,” *Annual Review of Entomology*, 2022, 67 (1), 181–199.

Marra, Peter P, Keith A Hobson, and Richard T Holmes, “Linking winter and summer events in a migratory bird by using stable-carbon isotopes,” *Science*, 1998, 282 (5395), 1884–1886.

Martin, Emily A, Björn Reineking, Bumsuk Seo, and Ingolf Steffan-Dewenter, “Natural enemy interactions constrain pest control in complex agricultural landscapes,” *Proceedings of the National Academy of Sciences*, 2013, 110 (14), 5534–5539.

May, Robert and Angela R McLean, *Theoretical Ecology: Principles and Applications*, OUP Oxford, 2007.

May, Robert M, “Biological populations with nonoverlapping generations: stable points, stable cycles, and chaos,” *Science*, 1974, 186 (4164), 645–647.

Mayne, Samuel J, David I King, Jeremy C Andersen, and Joseph S Elkinton, “Crop-specific effectiveness of birds as agents of pest control,” *Agriculture, Ecosystems & Environment*, 2023, 348, 108395.

MEA, *Millennium ecosystem assessment: Ecosystems and human well-being*, Vol. 5, Island press Washington, DC, 2005.

Moll, Jason D and Joel S Brown, “Competition and coexistence with multiple life-history stages,” *The American Naturalist*, 2008, 171 (6), 839–843.

Murphy, Stephen J, André M Bellve, Reymond J Miyajima, Natalie A Sebunia, Molly M Lynch, Walter Jetz, and Marta A Jarzyna, “SAViTraits 1.0: Seasonally varying dietary attributes for birds,” *Global Ecology and Biogeography*, 2023, 32 (10), 1690–1698.

Noack, Frederik, Dennis Engist, Josephine Gantois, Vasundhara Gaur, Batoule F Hyjazie, Ashley Larsen, Leithen K M’Gonigle, Anouch Missirian, Matin Qaim, Risa D Sargent et al., “Environmental impacts of genetically modified crops,” *Science*, 2024, 385 (6712), eado9340.

Nyffeler, Martin, agan H ekerciolu, and Christopher J Whelan, “Insectivorous birds consume an estimated 400–500 million tons of prey annually,” *The Science of Nature*, 2018, 105, 1–13.

Oerke, E-C, “Crop losses to pests,” *The Journal of Agricultural Science*, 2006, 144 (1), 31–43.

Otto, Sarah P and Troy Day, “A biologist’s guide to mathematical modeling in ecology and evolution,” in “A Biologist’s guide to mathematical modeling in ecology and evolution,” Princeton University Press, 2011.

Pinzon, JE, EW Pak, CJ Tucker, US Bhatt, GV Frost, and MJ Macander, “Global Vegetation Greenness (NDVI) from AVHRR GIMMS-3G+, 1981–2022, ORNL DAAC, Oak Ridge, Tennessee, USA,” 2023.

Polasky, Stephen and Andrew R Solow, “On the value of a collection of species,” *Journal of Environmental Economics and Management*, 1995, 29 (3), 298–303.

_ , Christopher Costello, and Andrew Solow, “The economics of biodiversity,” *Handbook of Environmental Economics*, 2005, 3, 1517–1560.

Quaas, Martin F and Stefan Baumgrtner, “Natural vs. financial insurance in the management of public-good ecosystems,” *Ecological Economics*, 2008, 65 (2), 397–406.

Raynor, Jennifer L, Corbett A Grainger, and Dominic P Parker, “Wolves make roadways safer, generating large economic returns to predator conservation,” *Proceedings of the National Academy of Sciences*, 2021, 118 (22), e2023251118.

- Reeve, John D**, “Predation and bark beetle dynamics,” *Oecologia*, 1997, 112, 48–54.
- Reynier, Emmett and Edward Rubin**, “Glyphosate exposure and GM seed rollout unequally reduced perinatal health,” *Proceedings of the National Academy of Sciences*, 2025, 122 (3), e2413013121.
- Rigal, Stanislas, Vasilis Dakos, Hany Alonso, Ainārs Auniņš, Zoltán Benkő, Lluís Brotons, Tomasz Chodkiewicz, Przemysław Chylarecki, Elisabetta De Carli, Juan Carlos Del Moral et al.**, “Farmland practices are driving bird population decline across Europe,” *Proceedings of the National Academy of Sciences*, 2023, 120 (21), e2216573120.
- Roberts, Michael J, Wolfram Schlenker, and Jonathan Eyer**, “Agronomic weather measures in econometric models of crop yield with implications for climate change,” *American Journal of Agricultural Economics*, 2013, 95 (2), 236–243.
- Rosenberg, Kenneth V, Adriaan M Dokter, Peter J Blancher, John R Sauer, Adam C Smith, Paul A Smith, Jessica C Stanton, Arvind Panjabi, Laura Helft, Michael Parr et al.**, “Decline of the North American avifauna,” *Science*, 2019, 366 (6461), 120–124.
- Savary, Serge, Laetitia Willocquet, Sarah Jane Pethybridge, Paul Esker, Neil McRoberts, and Andy Nelson**, “The global burden of pathogens and pests on major food crops,” *Nature Ecology & Evolution*, 2019, 3 (3), 430–439.
- Schnabel, Florian, Xiaojuan Liu, Matthias Kunz, Kathryn E Barry, Franca J Bongers, Helge Bruelheide, Andreas Fichtner, Werner Härdtle, Shan Li, Claas-Thido Pfaff et al.**, “Species richness stabilizes productivity via asynchrony and drought-tolerance diversity in a large-scale tree biodiversity experiment,” *Science Advances*, 2021, 7 (51), eabk1643.
- Schultz, Michael, Jan GPW Clevers, Sarah Carter, Jan Verbesselt, Valerio Avitabile, Hien Vu Quang, and Martin Herold**, “Performance of vegetation indices from Landsat time series in deforestation monitoring,” *International journal of applied earth observation and geoinformation*, 2016, 52, 318–327.
- Sekercioglu, Cagan H**, “Increasing awareness of avian ecological function,” *Trends in Ecology & Evolution*, 2006, 21 (8), 464–471.
- Springborn, Michael R, Joakim A Weill, Karen R Lips, Roberto Ibáñez, and Aniruddha Ghosh**, “Amphibian collapses increased malaria incidence in Central America,” *Environmental Research Letters*, 2022, 17 (10), 104012.

Stanton, RL, CA Morrissey, and RG Clark, “Analysis of trends and agricultural drivers of farmland bird declines in North America: A review,” *Agriculture, Ecosystems & Environment*, 2018, 254, 244–254.

Taylor, Charles A, “Cicadian rhythm: Insecticides, infant health and long-term outcomes,” *Center for Environmental Economics and Policy Working Paper*, 2021, (9), 1–45.

— and **Hannah Druckenmiller**, “Wetlands, flooding, and the clean water act,” *American Economic Review*, 2022, 112 (4), 1334–1363.

TEEB, RECOMMENDATIONS OF, *Mainstreaming the economics of nature*, TEEB Geneva, Switzerland, 2010.

Terza, Joseph V, Anirban Basu, and Paul J Rathouz, “Two-stage residual inclusion estimation: addressing endogeneity in health econometric modeling,” *Journal of health economics*, 2008, 27 (3), 531–543.

Tilman, David, Forest Isbell, and Jane M Cowles, “Biodiversity and ecosystem functioning,” *Annual Review of Ecology, Evolution, and Systematics*, 2014, 45 (1), 471–493.

—, **Michael Clark, David R Williams, Kaitlin Kimmel, Stephen Polasky, and Craig Packer**, “Future threats to biodiversity and pathways to their prevention,” *Nature*, 2017, 546 (7656), 73–81.

—, **Peter B Reich, and Johannes MH Knops**, “Biodiversity and ecosystem stability in a decade-long grassland experiment,” *Nature*, 2006, 441 (7093), 629–632.

USDA, “2022 Census of Agriculture Highlights: Farm Economics,” Technical Report, U.S. Department of Agriculture 2024. Accessed: 2025-06-12.

Watson, Robert, Ivar Baste, Anne Larigauderie, Paul Leadley, Unai Pascual, Brigitte Baptiste, Sebsebe Demissew, Luthando Dziba, Günay Erpul, Asghar Fazel et al., *Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*, IPBES Secretariat: Bonn, Germany, 2019.

Weitzman, Martin L, “On diversity,” *The quarterly journal of economics*, 1992, 107 (2), 363–405.

—, “The Noah’s ark problem,” *Econometrica*, 1998, pp. 1279–1298.

—, “Economic profitability versus ecological entropy,” *The Quarterly Journal of Economics*, 2000, 115 (1), 237–263.

Whelan, Christopher J, Çağan H Şekercioğlu, and Daniel G Wenny, “Why birds matter: from economic ornithology to ecosystem services,” *Journal of Ornithology*, 2015, 156, 227–238.

Wooldridge, Jeffrey M, *Econometric Analysis of Cross Section and Panel Data*, MIT press, 2010.

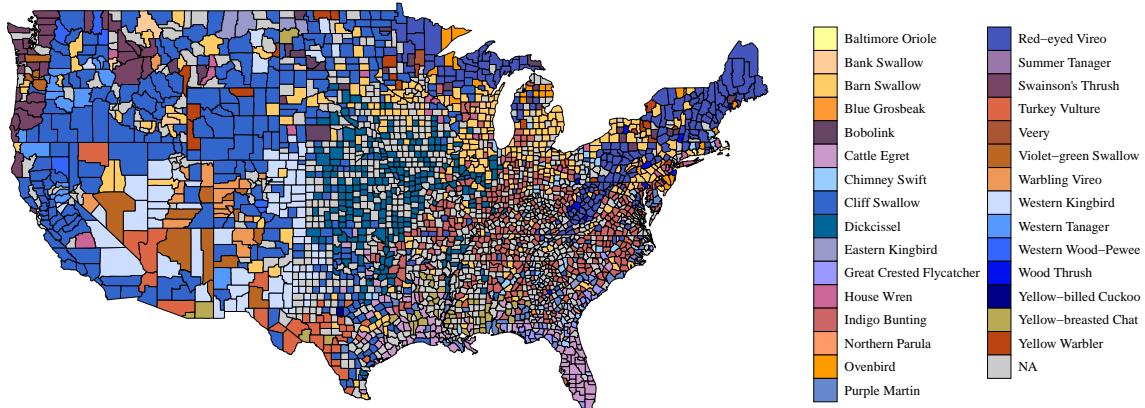
— , “Control function methods in applied econometrics,” *Journal of Human Resources*, 2015, 50 (2), 420–445.

Ziolkowski, D, M Lutmerding, Willow B English, and Marie-Anne R Hudson, “2024 Release - North American Breeding Bird Survey Dataset (1966 - 2023),” 2024.

A Appendix

A.1 Distribution of Dominant Long-Distance Migrant Species

Figure 8: Distribution of Dominant Long-Distance Migrant Species



Notes: Shading indicates the species with the highest average abundance, calculated across all route segments and years from 1983 to 2022 (the study period), within each county.

A.2 Properties of ES Function

Call total abundance \bar{x} and assume $x_i = \frac{\bar{x}}{N}$. Then

$$\begin{aligned} ES_{jt} &= \bar{x}^\iota \left(\sum_{i=1}^N x_{ijt}^{\rho_1} \right)^{\frac{1}{\rho_2}} \\ &= \bar{x}^\iota \left(N \left(\frac{\bar{x}}{N} \right)^{\rho_1} \right)^{\frac{1}{\rho_2}} \\ &= \bar{x}^\iota \left(N^{1-\rho_1} \bar{x}^{\rho_1} \right)^{\frac{1}{\rho_2}} \end{aligned}$$

Effective Species Numbers: For $\rho_2 = 1 - \rho_1$, and $\iota = -\frac{\rho_1}{\rho_2}$ the expression reduces to

$$ES_{jt} = N$$

which is independent of \bar{x} but increases in N .

Ecosystem Service Production: For $\iota = 0$ and $\rho_1 = \rho_2 = \frac{\sigma-1}{\sigma}$ the expression becomes

$$\begin{aligned} ES_{jt} &= N^{\frac{1-\rho_1}{\rho_2}} \bar{x}^{\frac{\rho_1}{\rho_2}} \\ ES_{jt} &= N^{\frac{1-\rho_1}{\rho_2}} \bar{x} \\ ES_{jt} &= N^{\frac{1}{\sigma-1}} \bar{x} \end{aligned}$$

which increases in \bar{x} and increases in N for $1 > \rho > 0$ or $\sigma > 1$, which we assume in the following.

A.3 Impact of Population Increases on ES Provision

Take the derivative of equation (4) with respect to x_{ijt} :

$$\begin{aligned} \frac{\partial ES(\mathbf{x}_{jt+1})}{\partial x_{ijt}} &= \frac{1}{\rho} \left(\sum_{l=1}^N x_{ljt+1}^\rho \right)^{\frac{1}{\rho}-1} \left[\rho x_{ijt+1}^{\rho-1} \left(\frac{x_{ijt+1}}{x_{ijt}} - x_{ijt+1} \alpha_i \right) - \alpha_i \rho \sum_{k=1}^{N \setminus i} x_{kjt+1}^{\rho-1} x_{kj+1} \right] \\ &= \frac{1}{\rho} \left(\sum_{l=1}^N x_{ljt+1}^\rho \right)^{\frac{1}{\rho}-1} \rho \left[\frac{x_{ijt+1}^\rho}{x_{ijt}} - \alpha_i \sum_{l=1}^N x_{ljt+1}^\rho \right] \end{aligned}$$

where $x_{ijt+1} := x_{ijt} e^{1 - \sum_{l=1}^N \alpha_l x_{ljt} + f(w_{it}^{hs}) + g(w_{it}^{hw})}$. The marginal effect of an additional species i in time t on the provision of ES is positive if

$$\frac{x_{ijt+1}^\rho}{x_{ijt}} - \alpha_i \sum_{l=1}^N x_{ljt+1}^\rho > 0.$$

A.4 Farmer's Solution

Introducing the damage function and the steady state pest stock into the farmer's problem yields:

$$\begin{aligned} \max_{M_{jt}} p\psi_{jt} e^{-\kappa(1-\gamma \log(ES_{jt}) - \mu \log(M_{jt}))} - cM_{jt} \\ \iff \\ \max_{M_{jt}} p\psi_{jt} e^{-\kappa} ES_{jt}^{\gamma\kappa} M_{jt}^{\mu\kappa} - cM_{jt} \end{aligned}$$

The first order condition is

$$p\psi_{jt}\mu\kappa e^{-\kappa}ES_{jt}^{\gamma\kappa}M_{jt}^{\mu\kappa-1} - c = 0$$

with solution

$$M_{jt} = \phi_{jt} ES_{jt}^{\frac{\gamma\kappa}{1-\mu\kappa}}$$

where we define $\phi_{jt} := \left(\frac{c}{p\psi_{jt}\mu\kappa e^{-\kappa}}\right)^{\frac{1}{\mu\kappa-1}}$. The sufficiency or second-order condition is

$$p\psi_{jt}\mu\kappa(\mu\kappa-1)e^{-\kappa}ES_{jt}^{\gamma\kappa}M_{jt}^{\mu\kappa-2} < 0$$

which is satisfied for $\mu\kappa < 1$. The yield equation is defined by

$$Y_{jt} := \psi_{jt}d(Z_{jt}^*) = \psi_{jt}e^{-\kappa}ES_{jt}^{\gamma\kappa}M^{\mu\kappa}$$

Using the optimal solution for M in the yield equation yields

$$Y_{jt} = \psi_{jt}e^{-\kappa}ES_{jt}^{\gamma\kappa}\left(\phi_{jt}ES_{jt}^{\frac{\gamma\kappa}{1-\mu\kappa}}\right)^{\mu\kappa}$$

Simplifying the expression yields

$$Y_{jt} = \phi'_{jt}ES_{jt}^{\frac{\gamma\kappa}{1-\mu\kappa}}$$

where $\phi'_{jt} := \psi_{jt}\phi_{jt}^{\mu\kappa}e^{-\kappa}$.

A.5 Impact of ES Provision on Farm Profits

The proof is based on the envelope theorem. In the following, we omit subscripts to simplify notation:

$$\begin{aligned} \frac{d\pi}{dES} &= p\psi \left(\frac{\partial d(Z^*)}{\partial Z^*} \frac{\partial Z^*}{\partial ES} + \frac{\partial d(Z^*)}{\partial Z^*} \frac{\partial Z^*}{\partial M} \frac{\partial M}{\partial ES} \right) - \frac{\partial c(M)}{\partial M} \frac{\partial M}{\partial ES} \\ &= p\psi \frac{\partial d(Z^*)}{\partial Z^*} \frac{\partial Z^*}{\partial ES} + \frac{\partial M}{\partial ES} \left[p\psi_{jt} \frac{\partial d(Z^*)}{\partial Z^*} \frac{\partial Z^*}{\partial M} - \frac{\partial c(M)}{\partial M} \right] \end{aligned}$$

The term $\frac{\partial M}{\partial ES} := \frac{\partial M}{\partial Z^*} \frac{\partial Z^*}{\partial ES}$ is the response of pesticide use to changes in pest populations caused by changes in ES. The term in square brackets is zero for an interior solution to the farmer's problem.

A.6 Decomposition of the ES function

Under the assumption that $g(w_{it}^{hw}) := \log(w_{it}^{hw})$ and $f(w_{jt+1}^{hs}) := \log(w_{jt+1}^{hs})$ (see Section 2.4), equation (4) can be written as

$$\begin{aligned} ES_{jt} &= \left[\sum_{i \in N} \left(x_{ijt-1} e^{1 - \sum_{l=1}^N \alpha_l x_{ljt-1} + \log(w_{jt}^{hs}) + \log(w_{it-1}^{hw})} \right)^\rho \right]^{\frac{1}{\rho}} \\ &= w_{jt}^{hs} e^{(1 - \sum_{l=1}^N \alpha_l x_{ljt-1})} \left[\sum_{i=1}^N (x_{ijt-1} w_{it-1}^{hw})^\rho \right]^{\frac{1}{\rho}} \\ &= w_{jt}^{hs} e^{(1 - \sum_{l=1}^N \alpha_l x_{ljt-1})} \left[\sum_{i=1}^N \left(\frac{x_{ijt-1} \bar{x}_{jt-1}}{\bar{x}_{jt-1}} w_{it-1}^{hw} \right)^\rho \right]^{\frac{1}{\rho}} \\ &= w_{jt}^{hs} e^{(1 - \sum_{l=1}^N \alpha_l x_{ljt-1})} \bar{x}_{jt-1} \left[\sum_{i=1}^N (s_{ijt-1} w_{it-1}^{hw})^\rho \right]^{\frac{1}{\rho}} \end{aligned}$$

where $s_{ijt-1} := \frac{x_{ijt-1}}{\bar{x}_{jt-1}}$ and $\bar{x}_{jt-1} := \sum_{i=1}^N x_{ijt-1}$ is the total number of individual across species. Taking logs yields equation (10).

A.7 Long-Distant Migratory Birds

The following bird species are included in the set of long-distance migrants that defines our instrument. We define these long-distance migrants as among the 100 most common species within the BBS data and with a winter habitat at least 90 % outside of the United States. The species include: Acadian Flycatcher, American Redstart, Baltimore Oriole, Bank Swallow, Barn Swallow, Blue Grosbeak, Bobolink, Cattle Egret, Chestnut-sided Warbler, Chimney Swift, Cliff Swallow, Dickcissel, Eastern Kingbird, Eastern Wood-Pewee, Great Crested Flycatcher, House Wren, Indigo Bunting, Least Flycatcher, Northern Parula, Orchard Oriole, Ovenbird, Purple Martin, Red-eyed Vireo, Rose-breasted Grosbeak, Scarlet Tanager, Summer Tanager, Swainson's Thrush, Turkey Vulture, Veery, Violet-green Swallow, Warbling Vireo, Western Kingbird, Western Tanager, Western Wood-Pewee, Wood Thrush, Yellow-billed Cuckoo, Yellow-breasted Chat, Yellow Warbler.

A.8 Summary Statistics

Table 8: Breeding Bird Abundance and Winter Habitat Shocks of Long-Distance Migrant Birds

	N	Mean	SD	Min	Q25	Median	Q75	Max
All Insectivorous Birds								
Abundance	314,587	129	88	0	80	113	155	6,275
Richness	314,587	24	9	0	17	24	31	63
ES (Elasticity = 2)	314,587	2,537	1,969	0	1,183	2,150	3,403	41,232
ES (Elasticity = 10)	314,587	173	113	0	106	154	212	6,401
NDVI Index (Elasticity = 2)	314,587	10,005	9,565	0	2,299	7,225	15,193	76,400
NDVI Index (Elasticity = 10)	314,587	1,758	1,202	0	805	1,630	2,547	8,089
Share Migratory Species (%)	314,587	21	14	0	10	20	30	100
Insectivorous Farmland Birds								
Abundance	312,946	82	64	0	41	72	108	5,235
Richness	312,946	13	6	0	8	13	18	34
ES (Elasticity = 2)	312,946	954	862	0	296	752	1,374	11,236
ES (Elasticity = 10)	312,946	105	80	0	51	92	140	5,350
Share Migratory Species (%)	312,946	17	15	0	5	14	24	100
Insectivorous Forest Birds								
Abundance	294,913	50	35	0	22	47	71	3,227
Richness	294,913	14	8	0	7	14	20	48
ES (Elasticity = 2)	294,913	786	798	0	138	580	1,185	14,067
ES (Elasticity = 10)	294,913	66	48	0	27	62	95	3,297
Share Migratory Species (%)	294,913	31	19	0	18	31	43	100
Crop Production								
Yield Apples [pounds/acre]	6,404	5,582	5,241	1	1,092	4,372	8,656	41,001
Yield Corn [bushels/acre]	28,828	99	43	2	68	94	125	316
Yield Soybean [bushels/acre]	22,242	32	11	3	24	31	40	100
Yield Wheat [bushels/acre]	27,175	41	17	2	29	37	49	175
Sales [Million USD]	36,221	96	2,977	0	12	31	75	400,000
Insecticide Use								
Insecticides [kilograms]	79,034	20,823	136,617	0	1,306	4,303	10,723	4,764,205
Forest Pests								
Bark Beetle Detection	234,078	0.12	0.33	0	0	0	0	1
Moth Detection	234,078	0.06	0.24	0	0	0	0	1

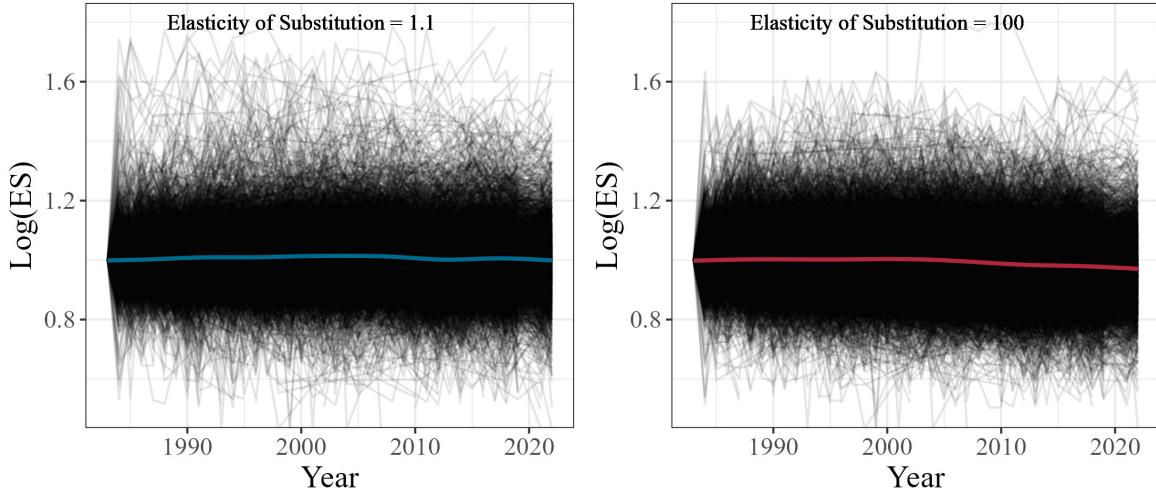
Notes: The table reports the number of observations (N), the mean, the standard deviation (SD), the minimum (Min), the 25th percentile (Q25), the Median, the 75th percentile (Q75), and the maximum value (Max). The biodiversity data are summarized separately for all insectivorous birds, insectivorous farmland birds, and insectivorous forest birds.

A.9 Ecosystem Service Dynamics

For the figure, we only use route segments with observations at the beginning of our study period and normalize values to that period. The year-to-year fluctuations of ES measured on individual route segments (thin black lines) are more pronounced than the long-term changes over time (red and blue

bold lines). Reasons for the lack of a strong overall trend may include the selection of the route segment or our aggregation method (ES).

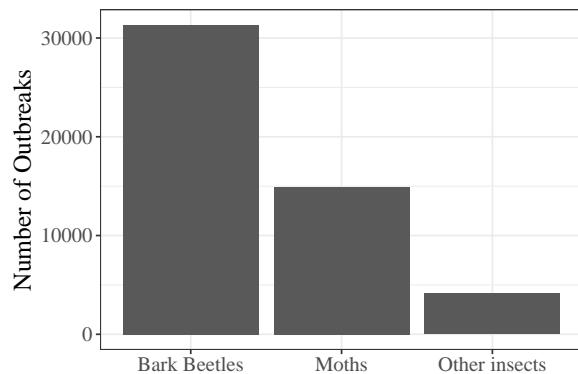
Figure 9: Ecosystem Service Dynamics



Notes: ES dynamics for low (1.1) elasticities of substitution (left panel) and high (100) elasticities of substitution (right panel). All values are expressed relative to the first period. Only route segments with observations in the first period of our study (1983) are included. The data include all insectivorous birds.

A.10 Forest Pests

Figure 10: Forest Pests



Notes: Number of pest outbreaks in 5km bands on both sides of the BBS route segments.

A.11 Main OLS Results

Table 9: OLS Results: Ecosystem Services and Agricultural Outcomes

Outcome Elasticity	Log(Sales)				Log(Yield)			
	2		10		2		10	
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Log(ES)	0.008*	0.006	0.001	-0.002	0.054***	0.055***	0.007	0.007
	(0.005)	(0.006)	(0.004)	(0.005)	(0.014)	(0.019)	(0.010)	(0.012)
Route-Segment FE	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Year FE	Yes	Yes			Yes	Yes		
Year-State FE			Yes	Yes			Yes	Yes
Observations	62,917	62,917	62,917	62,917	241,638	241,638	241,638	241,638

Notes: Standard errors are clustered at the county and route level. *, **, and *** indicate significance at the 10%, 5%, and 1% level. All specifications control for the share of migratory birds, $Ihs\left(\sum_{i=1}^N s_{ijt-1}^\rho\right)$, and lhs transformed growing degree days, killing degree days and precipitation in the current year.

A.12 Main IV Results with Conley Errors

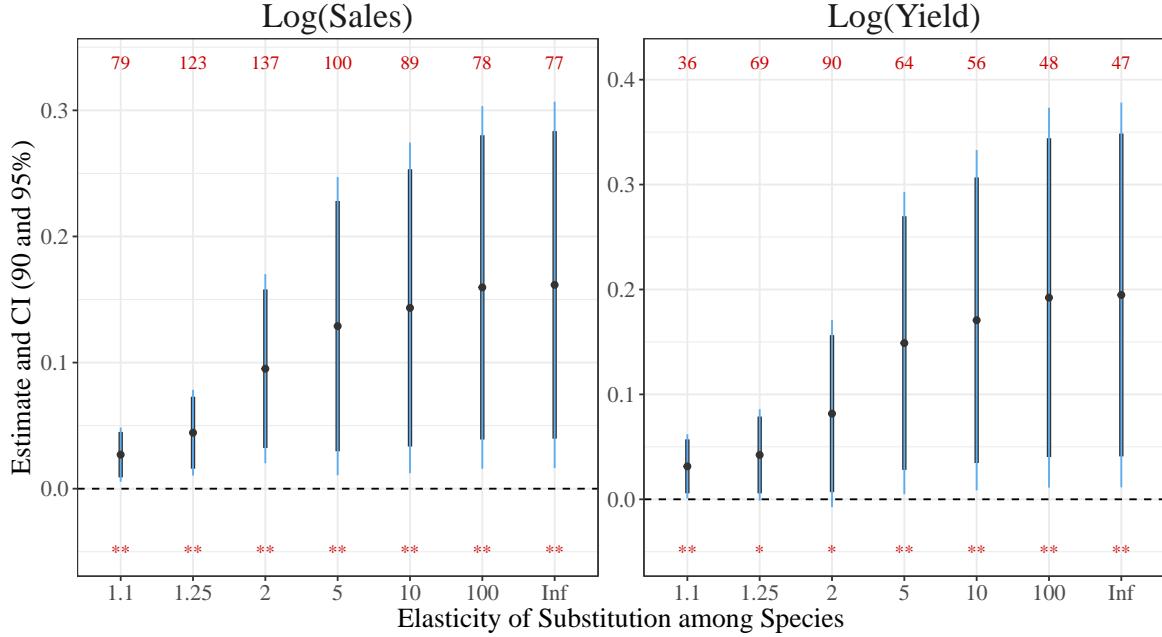
Here we present our main results using standard errors robust to spatial correlation (Conley, 1999). We use a distance cutoff of 50km because of the observation route length (40km) and the average county radius of about 30km. Additionally, spillovers could come from bird mobility. However, birds are localized during their breeding season, moving only a few 100 meters to kilometers from their nesting sites (e.g. Ambrosini et al. (2002); Iverson et al. (2024)).

Table 10: IV Results with Conley Errors: Ecosystem Services and Agricultural Outcomes

Outcome Elasticity	Log(Sales)				Log(Yield)			
	2		10		2		10	
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Log(ES)	0.095**	0.143**	0.070*	0.113*	0.047	0.171**	0.058	0.149*
	(0.038)	(0.067)	(0.037)	(0.066)	(0.029)	(0.083)	(0.045)	(0.085)
Route-Segment FE	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Year FE	Yes	Yes						
Year-State FE			Yes	Yes			Yes	Yes
Crop-Year FE					Yes	Yes		
Crop FE							Yes	Yes
Observations	62,917	62,917	62,917	62,917	241,638	241,638	241,638	241,638
Kleibergen-Paap	136.5	89.0	113.1	71.6	201.2	55.8	71.4	41.4

Notes: Conley standard errors are robust to spatial correlation within a 50km radius. *, **, and *** indicate significance at the 10%, 5%, and 1% level. All specifications control for the share of migratory birds, $Ihs\left(\sum_{i=1}^N s_{ijt-1}^\rho\right)$, and lhs transformed growing degree days, killing degree days and precipitation in the current year. ‘Kleibergen-Paap’ is the first stage Kleibergen-Paap Wald Statistic.

Figure 11: Agricultural Outcomes and the Elasticity of Substitution among Species.



Notes: Point estimates (black dots), 90% confidence intervals (black lines), 95% confidence intervals (blue lines), p-values (red asterisks at the bottom), and first-stage Kleibergen-Paap Wald statistic (red numbers on top) for our main regression results based on equation (13) and different underlying elasticities of substitution among species (x-axis). An infinite elasticity of substitution is equivalent to the total abundance of individuals across all species. All specifications control for the share of migratory birds, $Ihs\left(\sum_{i=1}^N s_{ijt-1}^\rho\right)$, and lhs transformed growing degree days, killing degree days, and precipitation in the current year. Additionally, all specifications include route-segment and year fixed effects in the Sales regression and route-segment and crop-by-year fixed effects in the Yield regressions. Conley standard errors are robust to spatial correlation within a 50km radius. *, **, and *** indicate significance at the 10%, 5%, and 1% level.

A.13 Crop Specific OLS Results

Table 11: OLS Results: Ecosystem Services and Crop Yields

	Log(Yield)							
	Corn		Wheat		Soybean		Apples	
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Log(ES)	-0.003 (0.007)	0.005 (0.006)	0.022*** (0.006)	0.010* (0.005)	0.009 (0.008)	-0.0005 (0.006)	-0.025 (0.051)	-0.061 (0.038)
Route-Segment FE	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Year FE	Yes		Yes		Yes		Yes	
Year-State FE		Yes		Yes		Yes		Yes
Observations	45,123	45,123	43,957	43,957	31,979	31,979	8,109	8,109

Notes: Standard errors are clustered at the county and route level. *, **, and *** indicate significance at the 10%, 5%, and 1% level. All specifications control for the share of migratory birds, $Ihs\left(\sum_{i=1}^N s_{ijt-1}^\rho\right)$, and lhs transformed growing degree days, killing degree days and precipitation in the current year.

A.14 OLS Results on Ecosystem Services and Insecticide Use

Table 12: OLS Results: Ecosystem Services and Insecticide Use

Elasticity	Log(Insecticide Use)			
	2		10	
	(1)	(2)	(3)	(4)
Log(ES)	-0.007 (0.008)	-0.008 (0.010)	-0.007 (0.007)	-0.010 (0.009)
Route-Segment FE	Yes	Yes	Yes	Yes
Year FE	Yes	Yes		
Year-State FE			Yes	Yes
Observations	222,270	222,270	222,270	222,270

Notes: Standard errors are clustered at the county and route level. *, **, and *** indicate significance at the 10%, 5%, and 1% level. All specifications control for the share of migratory birds Ihs ($\sum_{i=1}^N s_{ijt-1}^\rho$) and ihs transformed growing degree days, killing degree days and precipitation in the current year.

A.15 OLS Results on Forest Pests and 2SLS Results for Different Band-widths

Table 13: OLS Results: Ecosystem Services and Forest Pests

Pest Elasticity	Bark Beetle				Moth			
	2		10		2		10	
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Log(ES)	-0.0008 (0.002)	-0.003 (0.004)	2.64×10^{-6} (0.002)	-0.0002 (0.003)	0.009*** (0.002)	0.013*** (0.004)	0.008*** (0.002)	0.011*** (0.003)
Route-Segment FE	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Year FE	Yes	Yes			Yes	Yes		
Year-State FE			Yes	Yes			Yes	Yes
Observations	144,588	144,588	144,588	144,588	144,588	144,588	144,588	144,588
Dependent variable mean	0.123	0.123	0.123	0.123	0.061	0.061	0.061	0.061

Notes: Standard errors are clustered at the county and route level. *, **, and *** indicate significance at the 10%, 5%, and 1% level. All specifications control for the share of migratory birds, Ihs ($\sum_{i=1}^N s_{ijt-1}^\rho$), and ihs transformed growing degree days, killing degree days and precipitation in the current year.

A.16 2SLS Results for Different Bandwidths

Table 14: 2SLS Results: Ecosystem Services and Forest Pests (10 km Bandwidth)

Pest Elasticity	Bark Beetle				Moth			
	2		10		2		10	
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Log(ES)	0.038 (0.064)	0.050 (0.081)	0.035 (0.064)	0.059 (0.082)	-0.075** (0.036)	-0.119** (0.051)	-0.103** (0.042)	-0.137** (0.055)
Route-Segment FE	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Year FE	Yes	Yes			Yes	Yes		
Year-State FE			Yes	Yes			Yes	Yes
Observations	143,452	143,452	143,452	143,452	143,452	143,452	143,452	143,452
Dependent variable mean	0.164	0.164	0.164	0.164	0.095	0.095	0.095	0.095
Kleibergen-Paap	22.9	34.5	22.7	32.7	22.9	34.5	22.7	32.7

Notes: Standard errors are clustered at the county and route level. *, **, and *** indicate significance at the 10%, 5%, and 1% level. All specifications control for the share of migratory birds Ihs ($\sum_{i=1}^N s_{ijt-1}^\rho$) and ihs transformed growing degree days, killing degree days and precipitation in the current year.
'Kleibergen-Paap' is the first stage Kleibergen-Paap Wald Statistic.

Table 15: 2SLS Results: Ecosystem Services and Forest Pests (1 km Bandwidth)

Pest Elasticity	Bark Beetle				Moth			
	2		10		2		10	
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Log(ES)	-0.009 (0.043)	-0.007 (0.052)	-0.013 (0.045)	-0.005 (0.056)	-0.025 (0.019)	-0.027 (0.025)	-0.042** (0.021)	-0.035 (0.025)
Route-Segment FE	Yes	Yes						
Year FE	Yes	Yes			Yes	Yes		
Year-State FE			Yes	Yes			Yes	Yes
Observations	143,452	143,452	143,452	143,452	143,452	143,452	143,452	143,452
Dependent variable mean	0.062	0.062	0.062	0.062	0.030	0.030	0.030	0.030
Kleibergen-Paap	22.9	34.5	22.7	32.7	22.9	34.5	22.7	32.7

Notes: Standard errors are clustered at the county and route level. *, **, and *** indicate significance at the 10%, 5%, and 1% level. All specifications control for the share of migratory birds ihs ($\sum_{i=1}^N s_{ijt-1}^\rho$) and ihs transformed growing degree days, killing degree days and precipitation in the current year.
'Kleibergen-Paap' is the first stage Kleibergen-Paap Wald Statistic.

A.17 Two-Stage Residual Inclusion Logit Model Results: Ecosystem Services and Forest Pests (5 km Bandwidth)

This section reports the results using a logit model in a two-stage residual inclusion estimation (Terza et al., 2008; Wooldridge, 2015). The approach uses the residuals of the first stage as controls in the second stage. The number of observations is reduced compared to the OLS or IV case because

observations from groups that have no variation in the outcome variable are dropped from the analysis in non-linear estimations with fixed effects.

Table 16: Two-Stage Residual Inclusion Logit Model Results: Ecosystem Services and Pest Outbreaks

Pest Elasticity	Bark Beetle				Moth			
	2 (1)	10 (2)	2 (3)	10 (4)	2 (5)	10 (6)	2 (7)	10 (8)
Log(ES)	0.958 (1.15)	1.68 (1.45)	0.819 (1.28)	1.72 (1.66)	-3.59* (1.86)	-4.97** (2.23)	-5.91*** (2.03)	-7.64*** (2.53)
Route-Segment FE	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Year FE	Yes	Yes			Yes	Yes		
Year-State FE			Yes	Yes			Yes	Yes
Observations	49,296	49,296	37,406	37,406	48,333	48,333	40,399	40,399
Dependent variable mean	0.318	0.318	0.420	0.420	0.182	0.182	0.217	0.217

Notes: Standard errors are clustered at the county and route level. *, **, and *** indicate significance at the 10%, 5%, and 1% level. All specifications control for the share of migratory birds Ihs ($\sum_{i=1}^N s_{ijt-1}^\rho$) and ihs transformed growing degree days, killing degree days and precipitation in the current year.

‘Kleibergen-Paap’ is the first stage Kleibergen-Paap Wald Statistics.