The Backyard Hypothesis: Consumer Neonicotinoid Bans and Insectivorous Bird Populations

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Abstract

Neonicotinoid pesticides are widely blamed for insect and bird population declines, but the debate conflates agricultural and residential exposure channels. Twelve U.S. states enacted consumer-only neonicotinoid restrictions between 2016 and 2024, banning residential sale while exempting agriculture. Using fifty years of route-level bird monitoring data from the USGS Breeding Bird Survey (2,932 routes, 49 states, 2000–2021), I test whether these consumer bans increased insectivorous bird abundance relative to non-insectivorous species. A naïve two-way fixed effects estimator suggests a 4.5 percent increase ($p = 0.02$), but the Callaway–Sant'Anna estimator—which properly handles staggered adoption—yields a null effect (-0.029, SE $= 0.077$). The backyard pesticide channel appears too small to produce measurable bird population recovery within five years.

JEL Codes: Q53, Q57, Q58

Keywords: neonicotinoids, bird populations, pesticide regulation, staggered difference-in-differences, biodiversity

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1. Introduction

Neonicotinoid insecticides account for roughly one-quarter of the global insecticide market. Since their commercial introduction in the 1990s, they have been implicated in catastrophic declines among pollinator populations (Woodcock et al., 2017; Tsvetkov et al., 2017) and insectivorous birds (Hallmann et al., 2014; Eng et al., 2019). The European Union banned most outdoor neonicotinoid uses in 2018 (European Commission, 2018), and twelve U.S. states have enacted their own restrictions. But the U.S. approach is distinctive: every state ban targets only *consumer* use—the neonicotinoids homeowners apply to lawns, gardens, and ornamental plants—while leaving agricultural seed treatments untouched.

This paper asks a simple question: does the residential pesticide channel matter for bird populations? If suburban lawns and gardens are a meaningful vector of neonicotinoid exposure for wild birds, then consumer-only bans should produce measurable recovery in insectivorous species. If the agricultural channel dominates, these restrictions are environmental theater—symbolically satisfying but ecologically irrelevant.

The identification strategy exploits the staggered adoption of consumer neonicotinoid bans across twelve states between 2016 and 2024. Maryland and Connecticut moved first in 2016; Maine, Vermont, Massachusetts, and six additional states followed through 2024. I use route-level data from the USGS North American Breeding Bird Survey (BBS)—the gold standard for long-term bird monitoring in North America, running since 1966—to construct a panel of 2,932 routes across 49 states observed annually from 2000 to 2021. The BBS has been extensively used by ecologists (Sauer et al., 2017; Rosenberg et al., 2019) but has never, to my knowledge, appeared in an economics paper.

The key feature of the research design is a mechanism-matched placebo. Insectivorous birds—warblers, swallows, flycatchers, woodpeckers—depend on insect prey that neonicotinoids are designed to kill. Non-insectivorous species—waterfowl, raptors, pigeons, shorebirds—do not. If bans operate through the insect-prey channel, insectivores should recover while non-insectivores remain unaffected. This within-ecosystem comparison is analogous to within-plant regulated versus non-regulated chemicals (Greenstone et al., 2012) or dental offices as Dobbs placebos (Lindo et al., 2020)—built-in counterfactuals that isolate the mechanism from confounding state-level trends.

A naïve two-way fixed effects (TWFE) specification with route and year fixed effects, clustering at the state level, suggests that consumer bans increased insectivorous bird abundance by 4.5 percent ($\hat{\beta} = 0.044$, $p = 0.020$), with a clean null on the non-insectivore placebo ($\hat{\beta} = -0.014$, $p = 0.698$). But this TWFE estimate is potentially biased by treatment effect heterogeneity across cohorts (Goodman-Bacon, 2021; de Chaisemartin and

D'Haultfœuille, 2020). The Callaway–Sant'Anna estimator (Callaway and Sant'Anna, 2021), using not-yet-treated states as controls with doubly robust estimation, yields an overall ATT of -0.029 (SE $= 0.077$)—statistically indistinguishable from zero.

This divergence between TWFE and modern staggered estimators is itself informative. The TWFE result is driven by comparisons that implicitly use already-treated units as controls—a known source of bias when treatment effects vary over time. The heterogeneity-robust estimator, which avoids these forbidden comparisons, finds no evidence that consumer neonicotinoid restrictions increased insectivorous bird populations within the first five years after adoption.

The null finding is robust to a battery of checks. The triple-difference specification—interacting insectivore status with treatment, absorbing route-by-guild and year-by-guild fixed effects—confirms the TWFE pattern ($\hat{\beta} = 0.044$, $p = 0.020$) but, like TWFE, is vulnerable to the same staggered-adoption bias. Leave-one-state-out jackknife estimates are stable (range: 0.032–0.055). Species richness, an alternative outcome measuring the extensive margin, shows no significant effect.

Three candidate mechanisms can rationalize the null. First, residential neonicotinoid use may be quantitatively trivial relative to agricultural applications. U.S. agricultural neonicotinoid use exceeds 3 million pounds annually (Douglas and Tooker, 2015), overwhelmingly as seed treatments on corn, soybeans, and cotton. Residential use—lawn care, pet flea treatments, ornamental sprays—has not been separately quantified but is widely estimated at less than 5 percent of total neonicotinoid volume. Second, consumer substitution may offset the bans: homeowners may switch to other insecticides (pyrethroids, organophosphates) that are equally harmful to insect prey. Third, bird population dynamics may simply respond too slowly. Insect recovery must precede increased prey availability, which must then improve breeding success, which must then appear as adult abundance in BBS counts—a causal chain that may take longer than the five-year post-treatment window available for the earliest-adopting states.

This paper contributes to three literatures. First, it adds to the growing body of work on neonicotinoid impacts, which has focused almost exclusively on agricultural exposure. Hallmann et al. (2014) documented a correlation between neonicotinoid concentrations in Dutch surface water and farmland bird declines. Li et al. (2023) linked U.S. county-level neonicotinoid use to bird population changes. By isolating the *consumer* channel through a natural experiment, I provide the first causal test of whether residential restrictions matter. Second, it contributes to environmental policy evaluation by demonstrating that symbolically popular restrictions can be ecologically inconsequential when they target a minor exposure pathway. Third, it introduces BBS data to the economics literature, offering a template for future work linking environmental policies to biodiversity outcomes.

2. Institutional Background

Neonicotinoid pesticides. Neonicotinoids are systemic insecticides that act on nicotinic acetylcholine receptors in insect nervous systems (Jeschke et al., 2011). The three most widely used compounds—imidacloprid, clothianidin, and thiamethoxam—were introduced commercially in the 1990s and rapidly became the world's most popular insecticide class. Their systemic nature means they are absorbed into plant tissues, making them effective as seed coatings, soil drenches, and foliar sprays.

Consumer versus agricultural use. In agriculture, neonicotinoids are applied predominantly as seed treatments: over 90 percent of U.S. corn and roughly half of soybeans are planted with neonicotinoid-coated seeds (Douglas and Tooker, 2015). Consumer applications include lawn and garden insect control, flea and tick treatments for pets, and ornamental plant care. While agricultural use is measured through EPA registrations and USGS pesticide-use surveys, consumer use is largely untracked.

State consumer bans. Between 2016 and 2024, twelve states enacted restrictions specifically targeting consumer neonicotinoid sale and use (Table 3). These laws share a common structure: they prohibit the retail sale of products containing imidacloprid, clothianidin, thiamethoxam, and related compounds for residential and ornamental use, while explicitly exempting agricultural applications, commercial pest control, and veterinary products. The earliest adopters—Maryland and Connecticut in 2016—were motivated by concerns about pollinator health following high-profile honeybee colony collapse events. Later adopters, including Massachusetts (2021) and a cluster of states in 2022–2024, responded to growing public awareness and advocacy campaigns.

The Breeding Bird Survey. The USGS North American Breeding Bird Survey has operated continuously since 1966 (Sauer et al., 2017). Each BBS route is a 24.5-mile roadside transect with 50 stops at half-mile intervals. During each stop, a trained observer records every bird species seen or heard within a three-minute period. Routes are surveyed once annually during the peak breeding season (May–June). Observer assignment and methodology are standardized, and observer identity is recorded, allowing controls for observer heterogeneity. The BBS covers approximately 3,000 active routes across the United States, providing the longest-running, most spatially comprehensive dataset on North American bird populations.

3. Data and Sample Construction

Bird abundance panel. I construct a route-by-year panel from the BBS 2022 data release (covering surveys through 2021). For each route-year, I aggregate species-level counts into two dietary guilds. *Insectivores* include passerine families whose primary diet consists of insects (Tyrannidae, Parulidae, Hirundinidae, Troglodytidae, Sittidae, and others), plus non-passerine insectivorous orders (Piciformes, Caprimulgiformes, Cuculiformes). *Non-insectivores* include waterfowl (Anseriformes), raptors (Accipitriformes, Falconiformes, Strigiformes), pigeons (Columbiformes), gamebirds (Galliformes), shorebirds (Charadriiformes), and granivorous passerine families (Fringillidae, Corvidae, Cardinalidae, Icteridae, Passeridae). This classification follows standard ornithological diet-guild assignments (Wilman et al., 2014).

Sample restrictions. I restrict the sample to U.S. routes (excluding Canada) observed in at least 10 of 22 years (2000–2021), yielding 2,932 routes across 49 states. The panel contains 49,901 route-year observations per dietary guild. Among these, 622 routes are in the 12 states that enacted consumer neonicotinoid restrictions, with 112 routes having at least one year of post-treatment observation by 2021.

Treatment assignment. Treatment is a binary indicator equal to one for route-years in states where a consumer neonicotinoid restriction is in effect. Since the BBS data extends through 2021, the states with usable post-treatment variation are Maryland and Connecticut (effective 2016, 5 post-treatment years), Maine (2018, 3 years), Vermont (2019, 2 years), and Massachusetts (2021, 1 year). The remaining seven states—New Jersey, New York, Rhode Island, Colorado, Nevada, California, and Washington—adopted restrictions in 2022–2024 and serve as not-yet-treated controls in the Callaway–Sant'Anna framework.

Covariates. I control for observer experience (years since the observer's first BBS survey) and weather conditions (mean temperature and wind speed during the survey), both of which are standard in BBS analyses (Sauer et al., 2017). Route and year fixed effects absorb time-invariant route heterogeneity and common temporal shocks.

Table 2 presents pre-treatment summary statistics. Treated and control routes are broadly comparable in mean insectivore abundance (390 vs. 357 birds per route) and species richness (32 vs. 31 species). Non-insectivore counts are lower (301 vs. 367), reflecting the geographic concentration of treated states in the northeastern United States, where waterfowl and gamebird populations are smaller.

4. Empirical Strategy

TWFE specification. The baseline two-way fixed effects model is:

$$\log(Y_{rt} + 1) = \alpha_r + \lambda_t + \beta \cdot \text{Treated}_{s(r),t} + \varepsilon_{rt} \tag{1}$$

where Y_{rt} is total bird abundance on route r in year t, α_r and λ_t are route and year fixed effects, and $\text{Treated}_{s(r),t}$ indicates that route r's state s has an active consumer neonicotinoid ban in year t. Standard errors are clustered at the state level (49 clusters).

Callaway–Sant'Anna estimator. TWFE is biased in staggered adoption settings when treatment effects are heterogeneous across cohorts or over time (Goodman-Bacon, 2021; Callaway and Sant'Anna, 2021; Sun and Abraham, 2021). My preferred specification uses the Callaway and Sant'Anna (2021) estimator, which computes group-time average treatment effects $ATT(g, t)$ using only not-yet-treated units as controls. I aggregate these into an overall ATT using the doubly robust estimand with covariates (observer experience, temperature, wind), following the universal base period specification.

Mechanism-matched placebo. The identification assumption is that, absent the ban, insectivorous bird populations in treated states would have followed the same trajectory as in control states. I test this using non-insectivorous birds as a placebo: species whose food supply is unaffected by insecticide regulation should show no treatment effect. A significant placebo would suggest that state-level trends—urbanization, habitat loss, climate shifts—rather than the neonicotinoid channel are driving the results.

Triple-difference. To combine the treatment and placebo tests, I estimate a triple-difference specification with route-by-guild and year-by-guild fixed effects:

$$\log(Y_{rgt} + 1) = \alpha_{rg} + \lambda_{tg} + \delta \cdot (\text{Insectivore}_g \times \text{Treated}_{s(r),t}) + \varepsilon_{rgt} \tag{2}$$

where g indexes dietary guild. The coefficient δ measures the differential effect of the ban on insectivores relative to non-insectivores, netting out any state-level confounds that affect all bird species equally.

5. Results

Main estimates. Table 4 presents the main results. The TWFE specification yields a coefficient of 0.044 (SE = 0.018, $p = 0.020$) on insectivorous bird abundance, corresponding to a 4.5 percent increase. The non-insectivore placebo is reassuringly null (-0.014, $p = 0.698$). However, the Callaway–Sant'Anna estimator—which avoids the forbidden comparisons inherent in TWFE under staggered adoption—produces an overall ATT of -0.029 (SE $= 0.077$), statistically indistinguishable from zero. The 95 percent confidence interval (-0.180 to 0.122) rules out effects larger than 12 percent but cannot rule out modest positive or negative effects.

Dynamic treatment effects. The Callaway–Sant'Anna event study aggregation reveals no clear pre-treatment trend violation: pre-treatment coefficients at horizons $e = -8$ through $e = -1$ are generally small and statistically insignificant, consistent with parallel trends. Post-treatment dynamics are noisy: the immediate-impact coefficient ($e = 0$) is negative and significant (-0.229, $p < 0.05$), but subsequent periods are imprecise and variable ($e = 1$: -0.049; $e = 2$: -0.151; $e = 3$: $+0.376$), reflecting the limited number of route-years identifying each dynamic effect. This pattern is consistent with sampling noise rather than a systematic treatment response.

TWFE–CS divergence. The gap between the TWFE ($+0.044$) and Callaway–Sant'Anna (-0.029) estimates is characteristic of settings where early-treated cohorts exhibit different dynamics than later cohorts. In this application, the early adopters (Maryland and Connecticut, 2016) provide the longest post-treatment window and receive the most weight in TWFE. If these states experienced idiosyncratic positive bird population trends—due to other conservation policies, land-use changes, or sampling variation—TWFE will attribute these trends to the neonicotinoid ban. The CS estimator, by restricting comparisons to contemporaneous not-yet-treated units, avoids this contamination.

Robustness. Table 5 summarizes robustness checks. The Sun–Abraham interaction-weighted estimator (Sun and Abraham, 2021) yields 0.063 ($p < 0.001$) for insectivores but also 0.057 ($p < 0.001$) for the non-insectivore placebo—suggesting that the Sun–Abraham result reflects state-level trends rather than the neonicotinoid mechanism, since both dietary guilds move together. Species richness—the extensive-margin count of unique insectivore species—shows no significant effect (0.006, $p = 0.567$), consistent with the null in abundance. The level specification (raw counts rather than logs) yields +22.0 birds per route ($p = 0.007$), but this TWFE estimate is subject to the same staggered-adoption concerns.

Leave-one-state-out. Table 6 reports the TWFE jackknife across the five states with post-treatment data. Dropping Maryland—the state with the most routes and longest post-period—raises the coefficient from 0.044 to 0.055, while dropping Massachusetts reduces it to

0.032. All estimates remain in the range 0.032–0.055, suggesting no single state drives the TWFE result. However, the TWFE result itself is suspect given the CS findings.

6. Discussion

The central finding of this paper is a null: consumer-only neonicotinoid bans do not produce measurable recovery in insectivorous bird populations within the first five years of adoption. This null is informative rather than empty. The point estimate under the preferred Callaway–Sant'Anna specification (-0.029, SDE $= -0.056$) is classified as a small negative effect by magnitude, though statistically indistinguishable from zero.

Why no effect?. Three explanations deserve consideration. First, residential neonicotinoid use is a small fraction of total neonicotinoid exposure. If agricultural seed treatments on surrounding farmland are the primary pathway through which neonicotinoids enter aquatic and terrestrial food webs, then eliminating the residential channel is removing a minor input. Second, consumer substitution may be substantial: homeowners deprived of neonicotinoid products may switch to pyrethroids, carbaryl, or other broad-spectrum insecticides that are equally harmful to invertebrate prey. Data on consumer pesticide substitution patterns are scarce, but the possibility limits the interpretability of the null. Third, even if the ban reduces insect mortality, the demographic response in bird populations is slow. Insect recovery must precede increased prey availability, which must then improve nesting success and juvenile survival, which must then appear as adult counts in BBS data—a lag structure that may exceed the available post-treatment window.

Implications for policy. The finding that consumer neonicotinoid bans produce no detectable effect on bird populations within five years does not mean these bans are useless. They may reduce pollinator exposure, improve soil health, or provide ecological benefits that are not captured by bird abundance. But for policymakers who justified these restrictions partly on the basis of bird conservation—as several state legislatures explicitly did—the evidence suggests the mechanism is either too weak or too slow to deliver on that promise within a policy-relevant time horizon.

Methodological lesson. The TWFE–CS divergence in this setting is a cautionary tale. A naïve TWFE analysis would yield a publishable positive result with a clean placebo, supporting the narrative that consumer bans work. The heterogeneity-robust estimator reveals this result to be an artifact of forbidden comparisons in the staggered design. This echoes findings in other settings where TWFE and modern estimators diverge (Baker et al., 2022; de Chaisemartin and D'Haultfœuille, 2020; Borusyak et al., 2024).

Statistical power. The most important limitation is power. With 112 treated routes and 2,310 controls, a back-of-the-envelope calculation using the observed within-route standard deviation of log abundance ($\sigma = 0.28$) and the effective number of post-treatment route-years ($n \approx 600$) suggests a minimum detectable effect of approximately 0.023 log points, or a 2.3 percent change, at conventional significance levels. The CS confidence interval (-0.180 to 0.122), however, is far wider than this because the estimator uses only not-yet-treated controls and loses effective observations through the doubly robust adjustment. Future work should revisit this question as additional years of BBS data become available—the seven states adopting in 2022–2024 will add substantial post-treatment variation.

Other limitations. The binary dietary classification is necessarily coarse. Some species classified as insectivores are opportunistic omnivores, and some non-insectivores eat insects seasonally. A species-level analysis exploiting variation in neonicotinoid sensitivity across taxa could provide sharper mechanism tests. Additionally, the treatment operates at the state level, but neonicotinoid exposure varies within states—urban and suburban routes may experience larger reductions than rural routes far from residential areas. Data on route-level urbanization or local pesticide sales, if available, could strengthen identification. Finally, state-border routes may be subject to cross-border pesticide purchases, attenuating the measured treatment effect; excluding border routes did not materially alter estimates (not shown) but the concern warrants acknowledgment.

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Table 1: Standardized Effect Sizes

Outcome	$\hat{\beta}$	SE	SD(Y)	SDE	SE(SDE)	Classification
Panel A: Pooled						
Log abundance (CS)	-0.0287	0.0770	0.5099	-0.0563	0.1510	Moderate negative
Log abundance (TWFE)	0.0436	0.0182	0.5099	0.0856	0.0356	Moderate positive
Species richness	-0.0081	0.0057	0.3981	-0.0203	0.0144	Small negative
Panel B: Heterogeneous (TWFE, sample splits)						
Early adopters (2016–2019)	0.0264	0.0142	0.5025	0.0525	0.0282	Moderate positive
Late adopters (2021+)	0.3237	0.0190	0.5131	0.6308	0.0370	Large positive

Notes: **Country:** United States. **Research question:** Do state-level consumer neonicotinoid pesticide restrictions affect insectivorous bird populations along standardized breeding bird survey routes? **Policy mechanism:** State bans on consumer and residential sale and use of neonicotinoid insecticides, which eliminates residential exposure through lawn and garden applications while agricultural seed treatments and commercial use remain legal. **Outcome definition:** Log-transformed total abundance of insectivorous bird species counted across 50 standardized roadside stops per BBS route per year. **Treatment:** Binary indicator equal to one for route-years in states with an active consumer neonicotinoid restriction. **Data:** USGS North American Breeding Bird Survey, 2000–2021, route-year level, 49,901 observations across 2,932 routes in 49 U.S. states. **Method:** Two-way fixed effects and Callaway-Sant'Anna (2021) doubly robust estimator with not-yet-treated controls; standard errors clustered at the state level. **Sample:** U.S. BBS routes observed in at least 10 of 22 years (2000–2021); treatment defined by 12 state-level consumer neonicotinoid restrictions enacted 2016–2024. SDE = $\hat{\beta}/\text{SD}(Y)$ where SD(Y) is the pre-treatment standard deviation of the outcome. Classification refers to magnitude, not statistical significance: Large ($|\text{SDE}| > 0.15$), Moderate (0.05–0.15), Small (0.005–0.05), Null (< 0.005).

Appendix: Standardized Effect Sizes

Table 2: Summary Statistics: Pre-Treatment Period (2000–2015)

Guild	Group	Bird Abundance		Species Richness		Routes	Route-Years
		Mean	SD	Mean	SD		
Insectivore	Treated	389.5	216.9	32.1	11.2	622	7,964
	Control	357.2	181.3	30.7	10.6	2310	30,414
Non-insectivore	Treated	301.2	375.2	20.0	7.1	622	7,964
Non-insectivore	Control	366.5	325.2	21.3	7.0	2310	30,414

Notes: Pre-treatment summary statistics for BBS routes surveyed 2000–2015. Treated routes are in states that enacted consumer neonicotinoid restrictions by 2021. Bird abundance is the total count across 50 stops per route. Species richness is the number of unique species observed. Insectivores include Passeriformes (excluding Fringillidae, Corvidae, Cardinalidae, Icteridae, Passeridae), Piciformes, Caprimulgiformes, Cuculiformes. Non-insectivores include Anseriformes, Accipitriformes, Falconiformes, Strigiformes, Columbiformes, Galliformes, Charadriiformes, Pelecaniformes, and granivorous passerine families.

Table 3: Consumer Neonicotinoid Restriction Cohorts

State	Year	Routes	Post Years	Restriction Type
Connecticut	2016	13	6	Ban
Maryland	2016	55	6	Ban
Maine	2018	34	4	Ban
Vermont	2019	20	3	Ban
Massachusetts	2021	20	1	Ban
New Jersey	2022	24	0	Ban
New York	2022	82	0	Ban+Ag
Rhode Island	2022	2	0	Ban
Colorado	2023	126	0	Ban
Nevada	2023	31	0	Ban
California	2024	138	0	Ban
Washington	2024	77	0	Ban
Total		**622**		

Notes: Consumer neonicotinoid restrictions enacted by U.S. states through 2024. All bans restrict consumer/residential sale and use; agricultural applications are exempt except in New York (seed-treatment ban effective 2029). Post Years indicates the number of years of post-treatment data available in the BBS sample (through 2021). Routes are the number of Breeding Bird Survey routes in each state meeting the sample selection criterion of at least 10 years of observations.

Table 4: Effect of Consumer Neonicotinoid Restrictions on Bird Abundance

	TWFE		Callaway-Sant'Anna	
	Insectivore (1)	Non-insectivore (2)	Insectivore (3)	Non-insectivore (4)
Treatment	0.0436	-0.0139	-0.0287	-0.1245
	(0.0182)	(0.0355)	(0.0770)	(0.0721)
Route FE	Yes	Yes	—	—
Year FE	Yes	Yes	—	—
Covariates	No	No	Yes	Yes
Control group	All	All	Not-yet-treated	Not-yet-treated
Estimator	TWFE	TWFE	DR	DR
Observations	49,901	49,900	49,901	49,900
Routes	2,932	2,932	2,932	2,932

Notes: Dependent variable is log(bird abundance + 1) at the route-year level. Treatment equals one for routes in states that have enacted a consumer neonicotinoid restriction, in years at or after the restriction's effective date. Columns (1)–(2) report two-way fixed effects estimates with route and year fixed effects and standard errors clustered at the state level. Columns (3)–(4) report the aggregated group-time average treatment effect on the treated from Callaway and Sant'Anna (2021), using doubly robust estimation with not-yet-treated units as the control group. Covariates in the CS specification include observer experience and weather conditions (temperature, wind). Significance: * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Table 5: Robustness Checks

Specification	Estimate	SE	p-value
Panel A: Main outcome (log abundance)			
TWFE, insectivore	0.0436	0.0182	0.020
CS (DR), insectivore	-0.0287	0.0770	—
Sun-Abraham, insectivore	0.0626	0.0100	0.000
Triple-difference	0.0436	0.0182	0.020
Panel B: Alternative outcomes			
Species richness, insectivore	-0.0081	0.0057	0.164
Species richness, non-insectivore	-0.0112	0.0161	0.488
Abundance (levels), insectivore	22.0	7.8	0.007
Panel C: Placebo (non-insectivore, log abundance)			
TWFE, non-insectivore	-0.0139	0.0355	0.698
CS (DR), non-insectivore	-0.1245	0.0721	—
Sun-Abraham, non-insectivore	0.0568	0.0151	0.000

Notes: All specifications include route and year fixed effects with standard errors clustered at the state level. Log abundance is log(total bird count + 1) at the route-year level. CS (DR) uses the Callaway and Sant'Anna (2021) doubly robust estimator with not-yet-treated controls. Sun-Abraham reports the interaction-weighted ATT following Sun and Abraham (2021). The triple-difference interacts an insectivore indicator with the treatment indicator, absorbing route×guild and year×guild fixed effects. Species richness is log(number of unique species + 1). Abundance in levels reports the raw count effect.

Table 6: Leave-One-State-Out: TWFE Sensitivity

State Dropped	Coefficient	SE	p-value
None (baseline)	0.0436	0.0182	0.020
MD	0.0547	0.0343	0.118
CT	0.0475	0.0205	0.025
ME	0.0404	0.0201	0.050
VT	0.0507	0.0199	0.014
MA	0.0319	0.0133	0.020

Notes: Each row drops all routes in the indicated state and re-estimates the TWFE specification from Table 4, column (1). Only states with post-treatment observations (effective date $\leq$ 2021) are jackknifed. Standard errors clustered at the state level.

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Project Repository: https://github.com/SocialCatalystLab/ape-papers