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Abstract

The US has banned several of its most widely used insecticides, and total use has more than halved since its mid-1970s peak. But it remains an open question whether bans and falling volume have made insecticide use safer in practice. I use the predictable 13- and 17-year cycles of periodical cicadas, which damage tree crops but not row crops, to show that their emergence increases insecticide use in apple-growing counties. At the mean apple intensity of the top decile of counties, next-year infant mortality rises by about 1%, and exposed cohorts score lower on elementary-school tests years later. I find no evidence that the adverse impact of insecticides has declined since the 1950s and 1960s, suggesting that the regulatory regime offers no assurance of lower health costs.

JEL Codes: I18, J13, Q15, Q53

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1 Introduction

Modern pesticides, along with other technological advances in agriculture, brought large productivity gains in the post-war period (Jorgenson and Gollop 1992; Wang et al. 2015). They are also toxic by design, and US regulation has often acted by canceling individual compounds after decades of heavy use. Since the Environmental Protection Agency (EPA) banned DDT in 1972, it has canceled dozens of pesticides on account of their risk to humans (Buffington and McDonald 2006). Seven of the ten most-used insecticides of 1968 are no longer registered (Figure 1), and total insecticide use has more than halved since its mid-1970s peak. Yet even for DDT, the EPA still holds that "a relationship between DDT exposure and reproductive effects in humans is suspected, based on studies in animals," and classifies it as a probable human carcinogen (EPA 2026).

While many pesticides are "suspected" of harming human health, non-random exposure makes causal effects difficult to establish. Recent quasi-experimental studies have established adverse effects on infant health (e.g., Dias et al. 2023; Frank 2024), but they rely on variation from the recent past. Understanding whether insecticide regulation and falling use have lowered health costs requires comparing estimates over time.

The unique phenomenon I rely on is the predictable mass emergence of periodical cicadas, which damage tree crops but not row crops. They appear on 13- and 17-year cycles.[1] In 1775, Thomas Jefferson described the "great locust years" at his Virginia home, Monticello (Jefferson 1944), and that brood still emerges on schedule, most recently in 2013. A different brood emerged in the Washington, DC area in the summer of 2021, to extensive media coverage, and will re-emerge in 2038. As a result, I can employ a triple-difference design that compares counties with and without endemic cicadas, in emergence and non-emergence years, by how intensively they grow tree crops (measured by apple production).

I first show that insecticide use rises in apple-intensive counties when cicadas emerge (by about 20% in the top decile). Infant mortality rises in the following year, consistent with exposure during gestation. At the mean apple intensity of the top decile of counties, an emergence is followed by 0.14 additional deaths per 1,000 births, about 1% of the mean infant mortality over the sample period. Exposed cohorts also score lower on elementary-school tests, suggestive evidence that the effects extend beyond infancy.

Because emergences recur every 13 or 17 years across the post-war chemical era, I can test whether the reduced-form infant mortality effect in apple-intensive counties, which reflects the health cost of cicada-driven spraying, has changed across regulatory periods. I find no evidence that the effect declined. The estimated effect in the modern era since 1997, following

[1] Given the nature of prime numbers, 2024 was a special year for cicada enthusiasts with the syncing of the 17-year Brood XIII and 13-year Brood XIX, an event that last occurred in 1803 (17×13 years ago).

the 1996 Food Quality Protection Act's stricter standards for infants and children, is about as large as in the early era through 1972, when DDT and related compounds dominated overall insecticide use and when pesticides were regulated primarily by the US Department of Agriculture (USDA), before the EPA took over. Because infant mortality fell sharply over the sample period, the same effect in levels is proportionally larger today, although comparisons across eras are imprecise.

The effect is identified from county-average exposure, diluted across whole county populations that mostly live far from any orchard, not from the heavier exposure of people living next to orchards. Tree crops cover a relatively small portion of US counties (at most about 5% of county land area in apples, and generally far less than 1%), especially compared to row crops such as soy and corn, which can account for the majority of total acreage in many counties. Orchards are among the most insecticide-intensive crops per acre, but they occupy little land, so apples account for about 4% of US insecticide use.

Together these facts suggest that moderate pesticide exposure affects human health and development, in line with the emerging medical literature on subclinical toxicity, which posits that pesticides can have population-level effects through low-level exposure over time in food, water, or air, in ways that never amount to direct poisoning and thus are not observed by medical providers (Landrigan 2018). Because apples are sold on regional and national markets, airborne spray drift and runoff into water are more plausible channels for a county-level effect than residues on the fruit itself.

A back-of-the-envelope calculation attributes roughly 200 infant deaths to cicada-driven insecticide spraying on apples, a mortality cost of $1.9 billion, or $27 million annually from 1950 to 2020, at the EPA's value of a statistical life of $9.5 million in 2020 dollars (EPA 2024b). The benefit of insecticides is the crop loss they prevent, which I do not observe, so I ask how large that loss would have to be. Across the top-decile apple counties where cicadas occur, the mortality cost of one emergence in each county equals more than three-quarters of the value of a year's apple harvest in those counties. This spraying covers its health cost only if it prevents a crop loss at least that large, summed over the emergence year and the years of tree damage that follow (Section 7).

This paper contributes to the literature on early-life environmental exposures and to a recent quasi-experimental literature on agricultural chemicals and health. Shocks early in pregnancy can have long-lasting effects (Barker 1995; Almond and Currie 2011), and environmental exposures, including heavy metals and air pollution, have been causally linked to adverse outcomes at birth and later in life.[2] For pesticides, causal evidence is more limited.

[2] There is a rich literature documenting the impact of pollution on birth outcomes (Chay and Greenstone 2003; Currie and Neidell 2005; Currie and Walker 2011; Clay et al. 2014; Rangel and Vogl 2019) and

Until recently, most estimates of their effect on infant health came from small non-randomized studies (Bell et al. 2001, Jurewicz et al. 2006; Tago et al. 2014) or from occupationally exposed groups unlikely to represent the broader population (Regidor et al. 2004; Garry et al. 2002).[3]

Quasi-experimental studies provide stronger causal evidence, but each covers a single setting in recent decades. They study adverse birth outcomes near reported applications in California (Larsen et al. 2017), fertilizer runoff timed by planting seasons in India (Brainerd and Menon 2014), a policy change in China (Lai 2017), aerial spraying and newborn health on banana plantations in Ecuador (Calzada et al. 2023), aerial spraying of coca in Colombia (Camacho and Mejía 2017), soy expansion and childhood leukemia in Brazil (Skidmore et al. 2023), and the pesticide use that followed an invasive fruit fly (Jones 2020). The expansion of glyphosate after genetically modified seed adoption has been linked to worse birth outcomes in Brazil (Dias et al. 2023) and the US (Reynier and Rubin 2025) and to higher neonatal mortality downstream of application in the Corn Belt (Chen 2026). The closest design is Frank (2024), which documents an increase in infant mortality as farmers substitute insecticides for bat predation after white-nose syndrome spread from 2006.

The cicada design differs from existing approaches in two main ways. First, there is a recurring treatment tied to the cicada's fixed biological calendar, so treatment cannot respond to local economic or health conditions. Since each county is treated several times, there is less concern about a single confounding shock coinciding with treatment. Further, the calendar year can be shifted as a placebo, and the analysis can be replicated separately for each brood. Herbicides can also be used as a placebo since they target weeds rather than insects. Row crops, which cicadas do not damage, and forests, which host cicadas without being sprayed for them, serve as land-use placebos. Second, the sample runs for seventy years from 1950 to 2020 (Figure 2), a period long enough to evaluate whether the health cost changed as the mix of insecticides and the regulatory environment changed—a question that a single recent shock cannot address.

Two papers use historical variation in apple orchards. Baran (2026) studies arsenical sprays before 1945, and Fletcher and Noghanibehambari (2024) apply the cicada design introduced in this paper to later-life male mortality for 1925–1940 birth cohorts. Neither measures the insecticide response directly nor studies infant mortality across the post-war era characterized by synthetic pesticides.

childhood and adult outcomes (Sanders 2012; Ebenstein 2012; Marcus 2022; Currie et al. 2014; Zheng et al. 2016; Isen et al. 2017; Deryugina et al. 2019; Colmer and Voorheis 2020; Persico et al. 2020).

[3] Ecological studies have linked nitrates and atrazine in surface water (Winchester et al. 2009) and wheat acreage, a proxy for herbicide exposure (Schreinemachers 2003), to birth defects, and a cohort study links prenatal chlorpyrifos exposure to brain anomalies in children (Rauh et al. 2012).

A central problem in the economics of regulation is that pesticides, like many chemicals that reach consumers, are authorized on pre-market evidence with little power to detect diffuse and delayed harms. For pesticides, that evidence is mainly toxicity tests in rodents and other animals. The European Union applies a more precautionary standard to the same kind of evidence (Donley 2019), and many insecticides still registered in the US are not approved there (Figure 1). Studies of drug regulation quantify the costs of both slow and harmful approvals (Philipson et al. 2008), and economists have argued for evaluating regulations after the fact (Greenstone 2009; Cropper et al. 2017), but few studies estimate the health cost of approved chemicals across successive regulatory periods. The cicada design provides such an estimate across seven decades of changing chemistry and regulation.

I find no evidence that the infant mortality cost fell as older insecticides were banned and replaced. This pattern is consistent with regulation inducing substitution toward replacement compounds that were registered under the same evidentiary approach, such that the updated set of insecticides in use is not necessarily safer as a whole. The conclusion examines why a regulatory regime that approves chemicals on animal tests and leaves them on the market for decades while evidence of irreversible harm accumulates is poorly equipped to reduce diffuse health costs.

Pesticide regulation is the subject of ongoing lawsuits and political debates in both the US and Europe. In April 2026, 64% of US adults (including majorities of both parties) said pesticides used in agriculture are not regulated enough (Kearney et al. 2026). Chlorpyrifos, banned in the EU in 2020 (European Commission 2020) but still widely used in the US, had its food tolerances revoked by the EPA in 2021 and reinstated by a court in 2023, and the EPA has since proposed to keep them for only 11 crops, apples among them (EPA 2024a). Legislation to ban all organophosphate and neonicotinoid insecticides was introduced in the US Senate in 2020, 2021, and 2023 but has not been enacted (US Senate 2023). In 2024 the European Commission withdrew its proposal to halve chemical pesticide use and risk by 2030 amid farmer protests (European Commission 2024).

The findings are also relevant in developing countries. Pesticide use per hectare of cropland in Brazil is more than four times the US level (Food and Agriculture Organization of the United Nations 2025), and use in sub-Saharan Africa is low but growing rapidly, often faster than regulatory enforcement (Snyder et al. 2015; Haggblade et al. 2022). India moved to phase out monocrotophos, an organophosphate with no US registration since 1991 (EPA 1999), as late as 2023 (Government of India 2023).[4] Insecticide spraying is likely to grow

[4] India's 2023 order followed a 2020 draft that proposed banning 27 pesticides. The final order banned three, phased out monocrotophos, and restricted seven others, including chlorpyrifos and carbofuran (Government of India 2020; 2023).

with climate change, which is projected to increase pest-driven crop losses as well as pesticide resistance (Deutsch et al. 2018; Ma et al. 2021; Yang et al. 2024).

The paper proceeds as follows. Section 2 provides background on trends in pesticide use and regulation and on the relationship between cicadas and insecticide use. Section 3 describes the data, and Section 4 the empirical approach. Section 5 presents the insecticide response, the effect on infant mortality, how that effect varies across regulatory eras, and effects on elementary-school test scores. Section 6 addresses sensitivity and threats to identification, and Section 7 concludes with a discussion of policy implications.

2 Background

2.1 Pesticide use and regulation

Pesticides have long played an important role in society, with use documented since ancient Egypt, Rome, and China (Costa 1987). In the absence of crop protection, animal pests would reduce yields by an estimated 18% on average globally (Oerke 2006), and agricultural intensification has reduced infant mortality through income gains (Bharadwaj et al. 2020). Any health cost of pesticides has to be weighed against these benefits. Farmers in the US spend $16.5 billion annually on pesticides (USDA NASS 2021). But insecticide use has been in decline since its peak in the mid-1970s, even as herbicide use grew with the adoption of genetically engineered crops (Appendix Figure A1). Average application rates on major field crops (pounds per acre) fell by more than half, and the share of persistent pesticides (half-life exceeding 60 days) dropped from 54% in 1968 to 25% in 2008 (Fernandez-Cornejo and Jans 1995; Fernandez-Cornejo et al. 2014).

Yet these ostensible improvements over time mask compositional shifts. Organochlorines such as DDT were slowly replaced by organophosphates and carbamates, which target the nervous system in ways similar to chemical nerve agents. These were in turn replaced by pyrethroids and neonicotinoids (Fernandez-Cornejo et al. 2014). Further details on pesticide trends are provided in Appendix Figures A1 and A2.

Regulation occurred in tandem with these changes in pesticide use. Public concern about health and environmental impacts grew throughout the 1960s, culminating in the ban of DDT in 1972 (EPA 1975; Carson 1962). Panel (a) of Figure 1 shows that seven of the ten most-used insecticides on major US crops in 1968 (Fernandez-Cornejo et al. 2014) have since been banned or canceled, leaving carbaryl, dicrotophos, and malathion in use. Carbaryl is notably the only one to rank among the five most used in both 1968 and 2008. In light of growing health and environmental concerns, US and European regulators diverged in 2007:

the EPA re-registered carbaryl for agricultural use while restricting residential and pet uses, whereas the EU banned it outright (Commission Decision 2007/355/EC).

The two most widely used insecticides in 2008, chlorpyrifos and aldicarb, remain registered, but both have been contested in court (EPA 2024a). Several of the remaining widely used insecticides in the US are banned in the EU. For example, acephate and phosmet remain registered in the US despite being banned in the EU. However, the EPA proposed in April 2024 to cancel almost all registered uses of acephate except tree injections, and in 2023 agreed with phosmet's registrant on early measures to reduce risks to workers. Panel (b) shows when the primary insecticides of the last seventy years were first registered for use in the US and when they were phased out. Two things to note: first, many of the insecticides were developed in the immediate post-war 1950s, and second, those since canceled were typically in use for several decades first.

The US is unique in its regulatory regime in that many pesticides are "voluntarily" deregistered by their corporate owners, which is different from the regulator-driven approach in most countries. In some cases, cancellation occurs under pressure from the EPA in response to mounting evidence on health impacts or legal challenges, and in other cases, it can be an economic decision in the face of a product's declining sales, improved replacement pesticide availability, and patent lapses (Carroll 2016; Donley 2019). Nevertheless, all the canceled insecticides listed in Figure 1 have well-documented health and environmental risks in the scientific literature, and none is approved in the EU.[5]

2.2 Cicadas and insecticides

Periodical cicadas (*Magicicada* spp.) occur throughout the eastern half of the US.[6] Bob Dylan described the distinctive mating song of the cicada (colloquially called a locust) in "Day of the Locusts," about receiving an honorary degree from Princeton in June 1970:

> *And the locusts sang, yeah, it give me a chill*

[5] In the US, the pesticide industry has to demonstrate to the EPA that a product "will not generally cause unreasonable adverse effects on the environment" defined as "any unreasonable risk to man or the environment, taking into account the economic, social, and environmental costs and benefits of the use of any pesticide" under the Federal Insecticide, Fungicide, and Rodenticide Act, 7 U.S.C. §136 et seq. and "reasonable certainty of no harm" for pesticide residues on food under the Food Quality Protection Act Public Law 104-170 (1996), which placed particular emphasis on risks to infants and children, including an additional tenfold margin of safety unless reliable data support a different margin. For comparison, the EU places the burden of proof on companies to show that pesticides are safe (regulations 1107/2009 and 396/2005), requiring "that industry demonstrates that substances or products produced or placed on the market do not have any harmful effect on human or animal health or any unacceptable effects on the environment." (Donley 2019).

[6] Many annual (non-periodical) cicada species exist globally, but their populations do not tend to vary greatly from year to year.

Oh, the locusts sang such a sweet melody
Oh, the locusts sang their high whining trill
Yeah, the locusts sang and they were singing for me

These cicadas belonged to Brood X, the same cohort that visited the Mid-Atlantic three 17-year cycles later in 2021. Altogether, there are fifteen extant broods, three of which are on 13-year cycles and twelve of which are on 17-year cycles. Because adults rarely fly more than 50 meters from where they emerge, each brood returns to the same place at the cycle's end. Figure 3 maps each brood's range and cycle, with its emergence year in 2020–2036. Note that some counties receive multiple broods.

Cicadas spend most of their lives underground feeding on tree roots before synchronously emerging in late spring, with reported densities of 1.5 million per acre (Dybas and Davis 1962). Adults lay eggs in small tree branches (oviposition), causing particular harm to young trees, while feeding nymphs reduce tree growth by up to 30% (Karban 1980). Hamilton (1961) reported a complete loss among unprotected young apple and pear trees in the Hudson Valley following a 1945 cicada event, and Karban (1982) experimentally confirmed that removing cicada nymphs significantly increases wood accumulation in apple trees.

Pest risk is particularly high for perennial crops like apples, where insects can destroy long-lived assets and reduce yields both during and after an infestation (e.g., Cerda et al. 2017). Commercial tree growers use insecticides to mitigate cicada damage, a practice documented in both agronomic studies (Hamilton 1961)[7] and extension resources for fruit growers (Krawczyk 2026). Application is intensive during emergence: a 2021 Purdue University extension newsletter recommends spraying young trees as often as every 3 to 4 days during the month of adult activity (Long 2021).

3 Data

Cicada data. The US Forest Service provides shapefiles with county-level presence-absence data on periodical cicadas by brood with emergences projected through 2030 (Liebhold et al. 2013). Given the temporal and spatial consistency of cicada emergence, I extend the time series further into the past using each brood's 13- or 17-year cycle, assuming that cicada emergence occurred in the same counties. Distribution maps are derived from pioneering work done over a century ago (Marlatt 1898; 1907). These maps may overestimate current brood boundaries due to habitat loss and the misassignment of straggler cicadas (Marshall

[7] A 1960 USDA guide on periodical cicadas recommends applying carbaryl (Sevin) at 2 lb of 50% wettable powder per 100 gallons of water to control cicadas during emergence (USDA 1960).

2001), and have been updated periodically (Simon 1988; Cooley et al. 2009; Cooley et al. 2016; Cooley et al. 2021). While there are examples of accelerations in cycles and changes in the range of broods (Lloyd and Dybas 1966; Williams and Simon 1995), cicada behavior and brood distribution have been remarkably consistent for the most part (Marshall 2001).

For sensitivity analysis, I use a more recent map (2021) of eight cicada broods in four Mid-Atlantic states, which records where broods are absent as well as where they emerge, based on field surveys during emergences as well as museum and literature records (Cicadas.info 2021).

Agricultural and land use data. The land use dataset comes from the USDA's National Agricultural Statistics Service (NASS) online tool and from the historical US Census of Agriculture, as compiled by Haines et al. (2018) and distributed by the Inter-university Consortium for Political and Social Research (ICPSR). Various measures of apple intensity are collected at the county-year level (i.e., number of acres and production in bushels).[8] I use apple production as the measure of a county's exposure because apples are the historically dominant tree crop in the US. There is also ample agronomic and ecological literature on the effect of cicadas on apple trees, as described earlier.

My main measure averages county apple production in bushels over the eleven censuses from 1950 to 1997, the last census to report production in bushels. I hold the measure fixed because cicada damage can itself lower yields in the following years. An annual measure is also not available: the census is taken only every five years, and from the 1970s onward it records apple production in fewer than half of US counties. Orchards are long-lived, so county apple intensity changes slowly, and I also measure it in single census years early (1964) and late (1997) in the sample period. Apple production is well-distributed geographically among the cicada-endemic eastern US states, with top producers in the Northeast (NY, MA, CT), Central-Midwest (PA, MI, OH), and the South (VA, NC). Figure 3 shows the states included in my analysis along with cicada presence and quintile of apple production intensity. For a row-crop placebo, I also use NASS annual county estimates of corn and soybean harvested acreage.

The USDA's Cropland Data Layer (CDL), a remotely sensed, high-resolution (30m) measure of land use (USDA NASS 2008–17), is used as an alternative to administratively derived USDA census data (Appendix Figure A3). CDL data were spatially processed using Google Earth Engine. I use the 2008–2017 layers, beginning with the first year the CDL

[8] County-year data values of '(D)', which NASS uses to denote confidentiality, were coded as not available, and values of '(Z)', which denote being too small to estimate, were coded as zero. Given that only positive values are included in NASS output, excluded county-years are assumed to have a value of zero. All measures of agricultural intensity are standardized by county land area.

covers the entire US, and count a pixel as apple orchard or forest if it is classified as such in any of those years.

Pesticide data. The United States Geological Survey (USGS)'s National Water-Quality Assessment Project provides county-level pesticide use data from 1992 to 2018 (USGS 2019).[9] Information was compiled from surveys of farm operations in USDA Crop Reporting Districts and annual crop acreage reports. I measure insecticide use from the USGS EPest-high estimates as the sum of all compounds classified as insecticides, excluding sulfur, petroleum oil, and petroleum distillate. The EPA reports these three separately from conventional pesticides in its *Pesticide Industry Sales and Usage* series, and Douglas et al. (2020) similarly exclude oils. Herbicide and fungicide use are built the same way. Each aggregate is expressed in kilograms per square kilometer of county land area.[10] On this measure, apples account for about 4% of US insecticide use.[11]

Birth, death, and infant health data. Infant mortality comes from two sources. For 1942 to 1958, I use county-level resident infant mortality from ICPSR's County-Level Natality and Mortality Data (Bailey et al. 2018), which assigns deaths to the mother's county of residence. For 1959 to 2021, I construct infant mortality as deaths under age one per 1,000 births in the same county and year. Deaths come from National Center for Health Statistics (NCHS) mortality records (NCHS 2019), using NBER's files through 1988 and restricted files with county identifiers from 1989, when ICPSR begins masking infant deaths in counties with fewer than 100,000 residents. I double 1972 death counts because NCHS processed only half of that year's records. Births come from ICPSR through 2007 and from the denominator file of the NCHS Linked Birth-Infant Death Data thereafter. Appendix A describes the files, their coverage, and the adjustments in detail.

NCHS Natality Data files contain information on all US births from 1968 to 1988 (publicly available) and 1989 to 2016 (restricted use). I use these files for maternal characteristics, which enter the composition checks in Appendix Table A9. Note that these natality files are comparable to the denominator file in the NCHS Linked Birth-Infant Death Data, but

[9] USGS has finalized county-level pesticide data from 1992–2017 and preliminary data for 2018 and 2019. I use 1992–2018 since the 2018 data are preliminary but complete in coverage, whereas 2019 is excluded because of a reduced set of reported compounds, limiting comparability to earlier years.

[10] Compounds are classified by function (insecticide, herbicide, fungicide) using a crosswalk from the replication materials of Frank (2024), assembled from EPA registration documents and secondary pesticide databases. Categories are not mutually exclusive, so a compound registered for more than one use enters more than one aggregate. Ten entries are updated against current EPA registration records and the Pesticide Properties Database and compound names are updated with the USGS constituent list. In the resulting crosswalk of 513 compounds, 155 carry an insecticide flag, and 23 carry more than one.

[11] In 2007, the seven states in the NASS apple chemical use survey, which held 79% of US apple acreage, applied about 3 million pounds of insecticide to apples (USDA NASS 2008). This is 3.5% of US insecticide use that year, or 4.4% if the other states used insecticide on apples at the same rate per acre. As in the USGS measure, petroleum oils are excluded and lime sulfur, which NASS lists as a fungicide, is included.

cover a longer time period. Infant mortality in cicada-endemic counties is similar to that in the other sample counties, in both level and trend. Averaged over 1950–2021, county infant mortality is 14.4 deaths per 1,000 births in cicada-endemic counties and 14.8 in the others, and from 1950–1959 to 2012–2021 it fell by 76% and 78%, respectively. Appendix Figure A4 maps county average infant mortality and its change over time.

Education data. For educational achievement, I use standardized annual county-level test scores from the Stanford Education Data Archive (SEDA) 5.0 (Reardon et al. 2024). SEDA harmonized state and federal National Assessment of Educational Progress (NAEP) test results to create a spatially and temporally consistent dataset available for the eleven years from 2009 to 2019. Despite the challenges in comparing state-level test results, Kuhfeld et al. (2019) find high correlations between the SEDA data and NWEA's MAP Growth, which is another nationally administered test given to a subset of the population. SEDA reports county mean scores in units of the national student-level standard deviation for each grade and subject. I use SEDA third-grade scores, the earliest grade available, matched to each cicada exposure cohort defined by the test year minus ten.

Economic data. County farm income (1969 to 2020) and county GDP come from the US Department of Commerce, Bureau of Economic Analysis (BEA).

Summary statistics of the primary variables are included in Appendix Table A1.

4 Empirical Approach

Cicada emergences are publicly anticipated. Google searches for 'cicada' in Virginia spike in the emergence years of the state's main broods, both in the apple-growing region of Charlottesville, where Thomas Jefferson recorded the emergences at Monticello, and in Richmond and the Washington, DC area, which have few orchards (Appendix Figure A5). Public attention follows cicada timing in each location regardless of land use. Prior to this study, no research or media reports documented any health risks associated with a cicada emergence via increased pesticide use or other channels, and most coverage emphasizes that cicadas are harmless to humans. I therefore have no reason to believe that people take protective measures in response to cicadas.

The design compares apple-intensive counties in cicada emergence years with the same counties in other years, along with non-apple-intensive counties experiencing the same emergence. The identifying assumption is that, other than an increase in insecticide use, nothing related to the emergence calendar affects human outcomes differentially in apple- versus non-apple-intensive counties. This assumption is tested with various sensitivity checks and placebo tests in Section 6.

Insecticide exposure and its potential impact on health should be related to the life cycle of the cicada, the risk to tree crops, and the timing of human exposure. Appendix Figure A6 provides a conceptual framework. If accurate, one would expect three patterns. First, insecticide use should increase in the year of cicada emergence. Second, infant mortality should rise in the year following emergence. Third, yield impacts on tree crops should begin in the year before emergence as nymphs increase their root feeding and continue for several years. I test each of these propositions in the analyses that follow.

The empirical approach involves two steps: first testing whether there is an increase in insecticide use in treated counties in cicada emergence years, and second, whether there is a follow-on impact on infant health and longer-term outcomes. The regressor is a cicada presence-absence dummy, $cicada_{it}$, taking the value of 1 if there is a cicada emergence in county i in year t, and 0 otherwise. Cicada emergence for each brood is based on its endemic location and cycle time, as visualized in Figure 3. The cicada dummy is interacted with a fixed measure of apple intensity, $apple_i$, in county i. The sample is restricted to all the counties in the 33 states in the eastern half of the US that span the range of periodical cicadas, including non-cicada endemic counties in these states as well.[12]

Functionally, the empirical approach can be thought of as a triple-difference with spatial variation in cicada broods (i.e., is the cicada brood endemic to a given county?), temporal variation (i.e., is it a cicada emergence year?), and spatial variation in land use (i.e., what is the county's level of apple production?). But since the cicada presence-absence variable collapses the first two forms of variation into one, the specification can be written as a double-difference:

$$y_{it} = \beta_1 cicada_{it} + \beta_2 cicada_{it} \times apple_i + \alpha_i + \gamma_t + \mu_{s(i)}t + \epsilon_{it} \tag{1}$$

where y_{it} is the outcome of interest, α_i is a county-level fixed effect, and γ_t is a year fixed effect. County fixed effects absorb time-invariant county characteristics that could affect outcomes. Year fixed effects absorb national trends and annual shocks, such as changes in commodity prices and recessions. State-specific linear trends $\mu_{s(i)}t$ absorb gradual changes, such as those driven by state policy or the larger decline in infant mortality across the South (Appendix Figure A4).[13] Note that this model does not separately estimate the effect of apple

[12] The main analysis uses a balanced panel of 2,443 counties that are observed in every year. This excludes Massachusetts, which reported town-level vital statistics during the 1950s, and some Virginia independent cities that were incorporated at later dates. With a full unbalanced sample of 2,489 counties in 34 states, which includes these units for their available years, the results are essentially unchanged.

[13] The baseline uses state linear trends rather than state-year fixed effects or year-by-apple interactions because some cicada brood-years cover much of certain states (e.g., Brood X and Indiana) or major eastern orchard regions in the same year, as shown in Figure 3. Appendix Table A5 reports infant mortality estimates with state-year fixed effects, county linear trends, and year-by-apple interactions.

intensity, in itself, on outcomes because this measure is fixed over time and thus absorbed by the county fixed effect. The coefficient of interest, therefore, is β_2, which estimates the change in the outcome of interest in apple-intensive counties driven by cicada emergence.

In a first step, I specify the outcome variable y_{it} to be the log of insecticide use intensity, measured in kilograms of insecticide per km^2 in county i in state s in year t. In a second step, I model y_{it} as the infant mortality rate (infant deaths per 1,000 live births) in county i in the following year, $t+1$, $imr_{i,t+1}$. In addition to infant mortality, I test for impacts on educational outcomes.

This reduced-form analysis with cicada emergence as the treatment is preferable to using cicadas as an instrument for insecticide use for three reasons. First, USGS county-level pesticide estimates are not county measurements. They apply use rates surveyed across USDA Crop Reporting Districts, which average about ten counties, to each county's crop acreage. A county's measured insecticide use in a given year therefore partly reflects insecticide use in other counties of its district, which may or may not have had an emergence. An IV estimate would divide the reduced form by this mismeasured response, so its scale per kilogram is not easily interpretable.

Second, county-level pesticide data begin only in 1992, and a brood returns to a county only every 13 or 17 years, so the years in which both insecticide and mortality are observed contain few emergences per county. The reduced form does not require insecticide data, and so can be estimated from the start of the intensive chemical era in US agriculture after World War II, spanning the use of organochlorines, organophosphates, carbamates and the compounds in use today. The timeline of the datasets and national trends can be visualized in Figure 2.

Third, a dose response per kilogram of a 'general' insecticide does not have much meaning given that I use an aggregate measure of insecticide use: the composition of insecticides by chemical type changed drastically over time in response to regulation and technological development, as shown in Appendix Figure A2, often over the course of a 17-year cicada treatment life cycle. Such challenges in linking specific insecticide compounds to health outcomes are further exacerbated by the significant heterogeneity in pest management practices among apple producers across US geographies.

The coefficient β_2 therefore has a specific interpretation. It measures the change in next-year infant mortality when a cicada emergence prompts insecticide spraying in an apple-intensive county, averaged over all births in that county. It is thus not an individual-level effect. It averages over all births in the county, many of them far from any orchard, and cannot distinguish a large effect on a few nearby births from a smaller effect spread across the county. It also does not count pregnancies lost before birth, something inconsistently

recorded in vital statistics. Finally, the estimate is net of any defensive behavior. The spraying response to a cicada emergence is not publicized and the exposure is not visible, so parents have little reason to act differently in emergence years, and the estimate should be close to the health effect of the exposure itself rather than a behaviorally mitigated one.

5 Results

5.1 Insecticide response to cicada emergence

I first examine whether cicada emergence raises insecticide use in apple-intensive counties. Table 1 regresses log insecticide use on cicada emergence and its interaction with apple production, using county-level USGS pesticide data from 1992 to 2018. Columns (1)–(3) interact emergence with apple intensity, and columns (4)–(6) with an indicator for the top decile of apple intensity. Each set is estimated with state-specific linear trends, without trends, and with county-specific linear trends.

The interaction is positive in all six columns, statistically significant at the 10% level or better in each, and changes little across the trend specifications. Cicada emergence on its own does not raise insecticide use. The uninteracted estimate is small, negative, and indistinguishable from zero in every column. During an emergence, insecticide use in these apple-intensive counties rises by about 20% relative to other counties with an emergence.[14]

As discussed in Section 4, there is measurement error in the pesticide data. USGS applies survey rates from Crop Reporting Districts (ten counties on average) to county crop acreage, so the estimates do not record differences in application rates between counties of the same district. Since cicada broods intersect many of these districts, this will blur the contrast between counties with and without an emergence. Thus, I interpret results from this section as evidence for what is a known agronomic practice among orchard growers, confirming the direction of the insecticide response and providing an estimate of its rough magnitude.

That said, the detailed nature of the USGS pesticide data allows me to perform a number of robustness and placebo tests. Appendix Table A2 repeats the specifications in Table 1 using alternative aggregations of the pesticide measure based on the included chemicals. Panel A shows that adding sulfur and oils, which the EPA reports separately from conventional pesticides, to the insecticide aggregate leaves the interaction positive and statistically significant in every column. Without the three soil fumigants that the crosswalk flags as insecticides (methyl bromide, chloropicrin, and methyl isothiocyanate), the effect remains

[14] For ease of interpretation, I quote the magnitude under the top-decile indicator in column (4).

positive, though smaller and less precise, especially under the top-decile indicator. Carbaryl alone, the compound USDA recommended for periodical cicada control in 1960 (USDA 1960), also rises, although the estimate is imprecise in some specifications.

Herbicide use provides a falsification test. In orchards, herbicides are generally applied to control weeds on the orchard floor, in a process separate from foliar (leaf) spraying of pesticides (Washington State University 2018). So cicada-driven pest pressure would not be expected to materially change herbicide use. Panel B shows that herbicide use does not respond under the continuous apple intensity measure. Under two of the top-decile specifications, the estimate is marginally significant, though smaller than the insecticide estimates. The coefficient sign reverses between the continuous and top-decile specifications, further suggesting instability.

Interestingly, fungicide use also rises (Panel C). This likely reflects orchard spray programs, in which insecticides and fungicides are routinely tank-mixed in the same cover spray applied from late spring through summer (Ritchie and Walgenbach 2022). Because insecticides and fungicides are generally applied together, the health effects identified below reflect the overall orchard spray response to an emergence. A direct fungicide channel may relate to the mechanical wounds on branches from cicada oviposition. Wounds caused by insect feeding and egg-laying are a known infection pathway for fungus and bacteria, including fire blight in pear (Agrios 2005; Van der Zwet et al. 1997). Nevertheless, insecticides are the input that growers direct at the cicada, but I cannot rule out a contribution from the fungicides applied with them, some of which have documented developmental toxicity (Runkle et al. 2017; European Food Safety Authority 2020).

Appendix Figure A7 examines the timing of the response with an event study and with a placebo that shifts the whole cicada emergence calendar. The insecticide response generally falls in the emergence year, consistent with the timing framework in Appendix Figure A6, and none of the shifted calendars produce a response. Under the continuous apple measure, the event study is too imprecise to date the response, and under the top-decile indicator one pre-emergence coefficient is also positive. Given the measurement concerns noted above and the shorter sample period with pesticide data, these event studies should be treated as suggestive evidence of the agronomically documented insecticide response.

5.2 Cicadas and infant mortality

Given the insecticide response documented above, if insecticides are indeed dangerous, one would expect to see adverse health outcomes in apple-intensive counties following cicada emergence. I test this prediction by estimating Equation 1 with next-year infant mortality as

the outcome. Because this reduced form does not require insecticide data, it can be estimated over a much longer period than the 1992–2018 USGS pesticide data (Figure 2). The sample covers cicada emergences between 1949 and 2020, so that infant mortality, measured in the following year, spans 1950 to 2021 (for simplicity, I refer to the sample period as 1950–2020 throughout the paper). This window covers the post-war period when farmers used synthetic chemical pesticides *en masse*.

The main results are reported in Table 2, which regresses next-year infant mortality on cicada emergence. Model (1) shows no significant impact of cicada emergence, in itself, on infant mortality. Models (2)–(4) interact cicada emergence with continuous measures of county apple production in bushels: the average across 1950 to 1997 (the censuses in the sample period that report bushels), 1997 alone, and 1964 alone, with the latter two capturing apple intensity late and early in the period. All standard errors are clustered at the state level, which is the administrative level at which birth records are collected and aggregated.[15]

The interaction term is positive and statistically significant across all three apple measures, consistent with the long life of orchards. About two-thirds of counties in the top decile of apple production in 1964 remained there in 1997, and the correlation in county apple intensity between the two years is 0.84. I use the time-averaged measure from Model (2) of Table 2 as the preferred specification for the rest of the paper.

Model (2) yields a coefficient of 0.559 on the cicada-apple interaction, with apple intensity measured in thousands of bushels per km^2. The cross-county standard deviation of this measure is 0.185, so an emergence in a county one standard deviation more apple-intensive raises next-year infant mortality by $0.559 \times 0.185 \approx 0.10$ deaths per 1,000 births. Among the 244 counties in the top decile of apple intensity, mean intensity is 0.243, implying $0.559 \times 0.243 \approx 0.14$ additional deaths per 1,000.[16] An effect of 0.14 deaths per 1,000 is about 1% of mean infant mortality over the sample period. This effect is smaller than, but generally in line with, Frank (2024)'s modern day estimate of 0.54 deaths per 1,000 attributable to increased insecticide use after white-nose syndrome reduced bat populations.

Panel (a) of Figure 4 plots the cicada-apple interaction coefficients from the preferred Model (2) of Table 2 as an event study with leads and lags of cicada emergence:

$$imr_{i,t+1} = \sum_{k=-4,\, k\neq -1}^{4} \left(\delta_k D_{iet}^k + \beta_k D_{iet}^k \times apple_i\right) + \alpha_i + \gamma_t + \mu_{s(i)} t + \epsilon_{iet} \tag{2}$$

[15] Section 6.1 and Appendix Table A4 report alternative standard errors (spatial, county-clustered, and state-jackknife) and randomization inference that shifts each brood's emergence schedule, given that cicada emergences cross state borders and vary across only 15 broods.

[16] A regression using a top-decile apple intensity indicator, reported in Appendix Table A6, gives a similar point estimate of 0.18, though it is not statistically significant.

where D_{iet}^{k} equals one when year t is k years from emergence e in county i, and the year before emergence ($k = -1$) is the omitted reference. Each emergence with complete outcome data contributes a nine-year window, and the windows are stacked, so a county-year enters once for each window it falls in. Fixed effects are common across windows, and a window may contain a second brood's emergence. Leads and lags are limited to four years to reduce distortion from the fact that many counties receive more than one cicada brood, as seen in Figure 3. Because the outcome is next-year infant mortality, the coefficient at $k = 0$ measures mortality in the year following an emergence.

The coefficient of interest is similar in size to the static estimate in Model (2) of Table 2. While somewhat noisy, the coefficients before and after $k = 0$ are jointly indistinguishable from zero, so there is no evidence of differential pre-trends. Panel (b) of Figure 4 shifts each brood's emergence calendar by one to three years in either direction and re-estimates Model (2). Only the actual calendar produces an effect distinguishable from zero.

5.3 Brood-level analysis

To test whether the main result replicates across regions and treatment years, I estimate the specification separately for each of the five largest 17-year broods. Each brood's sample includes both counties in its endemic range and neighboring counties within 100 km as spatial controls. The five broods cover different regions of the US, each with its own climate, socio-economic characteristics, agricultural composition, and pest management practices. I exclude the two main 13-year broods, which are concentrated in warmer southern areas with little apple production (Figure 3).

Model (1) of Table 3 repeats the pooled estimate from Model (2) of Table 2 for comparison. The cicada-apple interaction is positive and statistically significant for four of the five broods, with coefficients close to the pooled estimate.[17] The exception is Brood XIII in the upper Midwest (Illinois, Indiana, and Wisconsin), whose coefficient is positive but imprecisely estimated. Its range has little apple production: total apple intensity across its 80 counties is less than one-eighth that of the 217 counties of Brood X.

Appendix Figure A8 shows event studies for each brood, using the stacked design of Figure 4. Despite the smaller samples, Broods II, V, X, and XIV each show a positive coefficient at $k = 0$ without clear pre-trends, most precisely for Broods X and XIV. Brood XIII, whose range contains the fewest top-decile apple counties of the five, shows no increase at emergence and noisier coefficients before it. Overall, the brood-level results suggest that

[17] Brood ranges overlap, so each brood's sample also includes emergences of the other broods within it, and the five samples are not disjoint.

the main finding is not driven by a single brood, region, or set of treatment years.

5.4 The effect across regulatory eras

This research design spans seven decades of regulatory change from 1950 to 2020. For the first two decades, pesticides were regulated by USDA until December 1970, when the newly created EPA took over (EPA 1975). Organophosphates and carbamates began displacing the organochlorines in the late 1960s. The 1972 cancellation of DDT and the follow-on cancellations of other organochlorines like aldrin, dieldrin, and chlordane made the shift in the composition of pesticides by chemical class permanent. Since the 1996 Food Quality Protection Act, several organophosphates have been restricted or canceled as pyrethroids and neonicotinoids gained share. The compounds that apple growers used against cicadas largely followed this evolution (Hamilton 1961; USDA 1960; Hogmire et al. 1990; Krawczyk 2026; Athey 2024). I therefore ask whether the cicada-driven infant mortality effect changed along with the regulatory regime and the primary class of insecticides in use.

To test this, I split the panel by cicada emergence year into three 24-year eras which can be visualized in Figure 2: an early era of primarily USDA regulation through the 1972 DDT cancellation, a transition era from 1973 to 1996 with EPA as the main regulator, and a modern era from 1997, coinciding with the Food Quality Protection Act signed in August 1996. Table 4 lets the treatment slope differ across these eras while holding the county, year, and state-trend structure common across the panel.

Column (1) shows the baseline estimate for comparison. Column (2) shows that the modern-era slope for 1997 to 2020 is positive, statistically significant, and close in magnitude to both the baseline and early-era estimates. The early era ($\leq$1972) and transition era (1973–1996) estimates are positive but imprecise. The transition era coefficient is about half the magnitude of the other two era coefficients, though none are statistically different from each other. With the 13- and 17-year cicada cycles and few emergences in each era, the design has limited power.

But because infant mortality fell by about three-quarters since 1950, a similar effect size in levels across eras represents a much larger effect relative to average infant mortality rates in later years. At the mean apple intensity of top-decile counties, the pooled effect of 0.14 deaths per 1,000 is about 0.6% of those counties' mean infant mortality in the early era and over 2% in the modern era. Estimated directly in proportional terms, with a Poisson model in column (3) or with log infant deaths in column (4), the era slopes show no decline in the modern era either—and if anything the relative effect is larger in the modern era. As an alternative modeling approach to using an era interaction, Appendix Figure A9 lets the

effect vary smoothly by year using a spline and tells a similar story: larger infant mortality effects in the early and modern periods, and smaller, statistically insignificant effects in the middle transition era.

While these analyses do not evaluate bans of specific chemical compounds, they show that the infant mortality cost of insecticide spraying is not confined to the early DDT-dominated regulatory era: the cost is present after 1997 and about as large in levels as over the full period, and larger relative to the current baseline mortality rates.

5.5 Test scores

I now examine longer-term effects on educational achievement among cohorts exposed to a cicada emergence in utero, using test scores at the county-year level from the Stanford Education Data Archive (Reardon et al. 2024). These standardized tests are given in the spring, so for third graders, when most students are nine, the cohort tested in year T was mostly born in $T-9$ and would be in utero during a cicada emergence in $T-10$. I therefore match each cohort to cicada events that occurred in its county in the test year minus ten. Third grade is the earliest grade with annual county-level scores, which makes it the outcome closest in time to the exposure and the least affected by later confounders involving migration, school-level effects, or other exposures.

I estimate Equation 1 with test scores (in student-level standard deviations) as the outcome across the 2009–2019 test cohorts, which correspond to cicada emergences from 1999 to 2009. Appendix Table A3 reports results for overall achievement (math and English average) and math and English individually, interacting cicada emergence with either continuous apple intensity or the top-decile indicator and including county and year fixed effects and state-specific linear trends. Cicada emergence on its own has no detectable effect in any column. The cicada-apple interaction is negative across all six specifications and statistically significant for overall scores and math under both apple measures, while estimates for English are smaller and marginally significant only under the top-decile indicator. Appendix Figure A10 plots the corresponding math event study for the four cohorts before and after each emergence: earlier cohorts show no pre-trend, the exposed cohort scores lower, and subsequent cohorts return toward zero.

These estimates are less precise than the infant mortality results and are likely biased toward zero. Children who move counties before third grade are assigned the wrong exposure. Cohorts born just before a cicada emergence (which serve as a comparison group) were infants during the insecticide shock and may themselves be affected. Selective survival works in the same direction: if prenatal exposure increases mortality among high-risk fetuses and infants

who would otherwise have scored lower, the surviving cohort is positively selected. However, given the estimated infant mortality effect of 0.14 deaths per 1,000 births in top apple counties, such selection effects would be too small to meaningfully shift county-level average test scores.

Overall, this negative impact of insecticides on test scores is consistent with evidence on the cognitive effects of prenatal and childhood exposure to environmental harms, including radiation (Almond et al. 2009), toxic waste (Persico et al. 2020), and lead (Aizer et al. 2018), and with medical evidence linking prenatal insecticide exposure to long-term cognitive harm (Rauh et al. 2012).

6 Robustness and Threats to Identification

This section examines the sensitivity of the main results to inference, specification, and treatment measurement. I also consider alternative explanations including farm income, maternal composition, fertility, and behavior. To be a confounder, such factors would have to align with the timing of cicada emergence and have a differential effect on infant mortality in apple versus non-apple intensive counties.

6.1 Inference and specification sensitivity

I first test whether the infant mortality results are driven by a single state or county. In Section 5.3, I showed that the main results hold when tested regionally at the individual cicada brood level. Appendix Figure A11 Panel (a) shows leave-one-out estimates after individually dropping each of the ten largest apple-producing states and ten largest apple-producing counties. All estimates remain positive and statistically significant.

As a design-based check, I perform randomization inference with 10,000 placebos that shift each brood's emergence schedule by a random number of years while preserving the spatial footprint and periodicity of the cicada treatment. The observed state-clustered t statistic lies in the upper tail of the placebo distribution (Panel (b) of Appendix Figure A11) and is significant at the 5% level in a two-sided test. Comparing coefficients rather than t statistics, the estimate is significant at the 10% level. Appendix Table A4 also reports county-clustered, state jackknife, and Conley spatial standard errors.

Appendix Table A5 shows that the results are also stable under alternative sample constructions, estimation approaches, and fixed effect structures. Weighting by number of births yields a coefficient roughly half the baseline that remains marginally significant. However, this implicitly weights high-population urban counties that are less likely to be exposed to

both intensive apple production and cicada emergences. Count-based estimators, a Poisson model with births as an offset and a log infant deaths specification, produce consistent positive effects (Panel A).

Replacing state-specific linear trends with no state trends, state-by-decade or state-by-year fixed effects, or county-specific linear trends leaves the coefficient similar or slightly larger, and adding year-by-apple interactions leaves the estimate positive and statistically significant, at roughly three-quarters of the baseline (Table A5 Panel B). State-by-year fixed effects absorb all discrete state-level interventions that could affect birth outcomes, county-specific trends allow each county to have its own linear trajectory, and year-by-apple interactions absorb any national shock that differentially affects apple-intensive counties in a given year.

6.2 Alternative treatment measures and placebos

Next, I allow for different measures of apple intensity. Appendix Table A6 Panel A adds a time-varying apple bushel measure and planted acreage to the bushel measures of Table 2. The interaction is positive and statistically significant for all five measures, whether or not they are divided by county land area. Under a log transform, the estimates are not statistically significant. This is somewhat expected as county-wide exposure depends on the overall intensity of apple orchards, whereas taking logs drops zero-apple counties and treats proportional differences among minimal producers the same as among major apple-producing counties. Top-decile indicators give positive estimates of similar magnitude, though collapsing production into a binary indicator reduces precision.

The USDA Cropland Data Layer provides a spatially explicit alternative land use measure derived from surveys and remote sensing starting in 2008. I compute the share of county area classified as apple orchard (Appendix Figure A3). The USDA census measure (the paper's baseline) and the CDL measures rank counties similarly, with a correlation of 0.7 between log census acreage and log CDL apple area. The CDL apple interaction in column (2) of Panel B is positive and marginally significant. Coefficient magnitudes should be read with caution since CDL pixel summation can misestimate county cropland area (Lark et al. 2017).

Two placebos examine land uses where a cicada emergence should not raise insecticide use. Like orchards, forests host cicadas, but I could find no evidence in the literature or anecdotally that owners of forests, whether natural or plantation, use insecticides in response to cicadas. Mature forest trees tolerate twig damage, and the economics and scale of forestry operations do not justify chemical control, unlike high-value fruit trees. Major row crops like corn and soybeans, on the other hand, are not hosts to cicadas, as cicadas feed on

woody root xylem and lay eggs in branch tips of trees—not herbaceous plants. Neither the forest interaction in column (3) nor the row-crop interactions in columns (4) and (5) is distinguishable from zero. Controlling for row-crop intensity leaves the apple estimate unchanged.

Another concern is the accuracy of the century-old cicada maps (Section 3). Inaccuracies could reflect both error in the placement of broods and major land use changes that have shifted the distribution of cicadas, largely through urbanization (i.e., cicadas cannot emerge from soils covered in concrete). The 2021 map described in Section 3 (Cicadas.info 2021) provides an alternative source of treatment data for four Mid-Atlantic states (MD, PA, VA, WV). The two maps differ materially. The new map disagrees with the original map's recorded absences 1.9% of the time and with its recorded presences 29% of the time. Appendix Table A7 shows regression results alongside the baseline estimates from the main model. On roughly a tenth of the sample, both Mid-Atlantic columns give a positive and statistically significant interaction of similar magnitude to the full-sample estimate, whether cicada treatment is defined by the original map or by the 2021 map.

6.3 Yields and income

Agricultural yields offer an alternative explanation. If cicadas sharply reduce apple production, an emergence could affect infant health through lost income rather than through insecticide spraying. The agricultural census is collected roughly every five years and cannot capture annual shocks, but USDA tracks annual apple production for 168 counties in Virginia, South Carolina, Kansas, Pennsylvania, and New Jersey from 1972 to 2012. I estimate an event study of county apple production around each emergence. Effects are measured relative to two years before emergence, since nymph feeding below ground may reduce yields beginning in the year before.

Apple production dips in the year before and the year of emergence by roughly 6 to 8% in logs, and returns to its earlier level afterward (Appendix Figure A12). The estimates are imprecise, and with annual data for only five states, only the decline in the year before emergence is marginally significant. The timing matches the third prediction in Appendix Figure A6, that crop losses should begin before cicada emergence as nymphs increase tree root feeding, which in turn reduces tree growth (Karban 1982). Given the observed yield reduction in the year before cicada emergence, if lower income harmed pregnancies in utero and raised infant mortality the following year, that effect would manifest in the year the cicadas emerged. Yet Figure 4 shows no such concurrent increase in infant mortality, only one in the year after emergence.

Lower yields are also unlikely to explain the result because orchards account for a small share of a county's economic output. Even in Adams County, PA, the largest apple producer among cicada-endemic counties, the value of *all* fruit production in 2017 was $62 million, or 1.5% of its $4 billion GDP (US Bureau of Economic Analysis 2026; USDA NASS 2019). A direct test regresses BEA county farm income, 1969 to 2020, on cicada emergence and its interaction with apple intensity (Appendix Table A8). Farm income is slightly lower in cicada emergence years across all counties, but the interaction with apple intensity is positive and not statistically significant, so there is no sign that emergences depress farm income more in the apple-intensive counties that drive the infant mortality result.

6.4 Maternal composition, fertility, and behavior

Another concern is that the mothers who give birth in apple-intensive counties the year after a cicada emergence differ from other mothers in ways that affect infant health. Panel A of Appendix Table A9 compares these mothers to ones who gave birth in non-emergence years and finds no meaningful difference in average education, racial and ethnic makeup, or age. Panel B adds these maternal characteristics as controls and the infant mortality estimate is essentially unchanged within each sample window. These columns use shorter samples, because Hispanic origin is recorded only from 1978, and not in every state until 1991, and education only through 2008.

Panel C regresses the next-year birth rate, per 1,000 people and per 1,000 women aged 15–44, on cicada emergence and its interaction with apple intensity. The birth rate is slightly higher in the year after an emergence in all counties,[18] but this common shift does not enter the comparison that identifies the mortality effect, and the interaction with apple intensity is negative and not statistically significant. In principle, fewer births could raise the infant mortality rate mechanically through a smaller denominator. Even at the point estimate in column (2), however, the top-decile mean apple intensity of 0.243 implies $-0.132 \times 0.243 \approx -0.03$ births per 1,000 people, about 0.2% of the mean birth rate, compared with an increase in infant mortality of about 1% of its mean.

A further concern is that cicadas could alter behavior, such as risk-taking, in ways that affect infants outside the insecticide channel. Appendix Table A10 decomposes infant mortality by cause of death for 1968 to 2020. In this shorter window the all-cause estimate is smaller and less precise than in the full sample. Deaths from external causes, such as accidents and violence, do not rise with apple intensity after an emergence, while the es-

[18] Interestingly, birth rates rise by 1% of their mean in the year after a cicada emergence, regardless of apple intensity. Whether cicadas affect fertility, and through what channel, is a question for future work.

timate for all other causes is positive and similar in size to the all-cause estimate. The estimates for specific causes are imprecise, and only cardiovascular deaths, about 2% of the total, show a statistically significant increase. Whether prenatal insecticide exposure impairs cardiovascular development, and through what biological mechanism, merits further study.

7 Conclusion

US insecticide use has more than halved since the mid-1970s, and seven of the ten most-used insecticides of 1968 have since been banned or canceled (Figure 1). This paper asks whether the health cost of insecticides fell as the chemistry and its regulation changed. To test this, I use a natural experiment involving a unique insect: periodical cicadas emerge in fixed locations on 13- and 17-year cycles and damage tree crops. Such emergences raise insecticide use in apple-intensive counties. At the mean apple intensity of the top decile of counties, next-year infant mortality rises by 0.14 deaths per 1,000 births, about 1% of mean infant mortality over the sample period. The effect appears in most of the major broods estimated separately, and exposed cohorts score lower on elementary-school tests.

I find no evidence that this public health cost has declined. The estimate for emergences since the 1996 Food Quality Protection Act is statistically significant and about as large in levels as in the early era of the 1950s and 1960s (Table 4). Because infant mortality has fallen sharply, the same effect in levels is proportionally larger today, although comparisons across eras are imprecise. Falling insecticide use is no assurance that the health cost of insecticides has fallen.

That the infant mortality effect appears at the county level suggests that the cost of pesticide use extends beyond the orchards themselves. Apples cover about 0.1% of US cropland (USDA NASS 2024). In 2002, the average top-decile apple county in the sample had about 730 acres of apples, compared with more than 200,000 acres of corn and soybeans in the average county in Illinois or Iowa. Given this small footprint, relatively few people live close to orchards. While the exposure pathways are beyond the scope of this study, plausible channels are spray drift and runoff into drinking water, and drift can carry insecticide well beyond treated orchards (Damalas and Eleftherohorinos 2011). Residues on the fruit itself are an unlikely channel for the county-level effect, because apples are sold regionally or nationally rather than in the county where they are grown.

Applying the estimated effect to each top-decile county's apple intensity and births in the year after an emergence gives roughly 200 infant deaths from 1950 to 2020.[19] At the EPA's

[19] The 200 deaths come from 164 top-decile counties with an endemic brood, each with about six emergences across 1950 to 2020 and an average of 2,400 live births in the year after each one, or 2.3 million births in

value of a statistical life of $9.5 million in 2020 dollars, these deaths amount to a mortality cost of $1.9 billion, or $27 million per year. The calculation covers only the additional insecticide use that follows emergences in 164 apple-intensive counties of the eastern US. Apples account for about 4% of US insecticide use, so if other uses carried a similar risk per pound, the national health cost would be much larger. However, I do not estimate this, because the crops, chemical compounds, and nearby populations differ too much for a simple scaling. The value of this paper's cicada design is in its ability to provide causal evidence on pesticide health costs across a wide geography, a long time period, and a broad array of chemical types, the dimensions over which credible aggregate estimates are otherwise unavailable.

The private benefit of insecticide use is the avoided crop loss, which I do not observe, so I ask instead how large that loss would have to be. The 200 deaths figure above comes from about six cicada emergences in each county, so a single emergence in each of the 164 top-decile counties with an endemic brood implies about 34 infant deaths, or a VSL-equivalent $320 million, against about $410 million of annual apple production in those counties.[20] For insecticides to cover the health cost, they would have to prevent a crop loss of more than three-quarters of one year's harvest in these counties, including damage in both the emergence year and the years following. This threshold counts only infant deaths and omits non-lethal effects on surviving children, on which the test-score results in Section 5.5 offer suggestive evidence.

Such costs can arise even from long-approved pesticides: carbaryl, which USDA recommended for cicadas in 1960 (USDA 1960), is still used in the US though not approved in the EU. The herbicide paraquat, registered in the US since 1964, likewise remains widely used despite decades of evidence linking exposure to Parkinson's disease (Tanner et al. 2011; Paul et al. 2024) and bans in the EU, China, and Brazil (Donley 2019). For decades, the EPA assumed minimal airborne vapor exposure based on manufacturer data, until 2024, when the manufacturer disclosed a much higher vapor pressure of paraquat and greater potential for post-application volatilization, to which the EPA responded by planning a request for additional data (EPA 2025). The episode illustrates the challenges that the EPA faces in verifying basic exposure parameters under its own rules, let alone detecting diffuse health

total. For each emergence, I multiply the Model (2) coefficient in Table 2 by the county's apple intensity and its births, divide by 1,000, and sum. The EPA's central value of a statistical life is $10.7 million in 2022 dollars (EPA 2024b, Appendix B), which I convert to 2020 dollars.

[20] Apple value is the NASS state value of utilized production (USDA NASS 2014–21), averaged over 2011–2020 in 2020 dollars and allocated to counties by their share of each state's 1997 census apple bushels. Calculating the single-emergence mortality cost from 1997 apple intensity (Model (3) of Table 2) and 2011–2020 births, or from the births-weighted specification in Appendix Table A5, gives about two-fifths of that annual apple value.

effects that pesticide registration is not designed to capture. In the absence of federal action, states have responded on their own: in 2026, Vermont enacted a phase-out of paraquat (Vermont General Assembly 2026), and manufacturers withdrew their California registrations rather than submit state-mandated safety data (CDPR 2026).

Where regulation has been slow to restrict a chemical, the courts have offered a partial ex post remedy, most visibly in glyphosate herbicide litigation, where settlements have exceeded $10 billion (Cohen 2020). However, this channel has recently narrowed. In *Monsanto Co. v. Durnell* (2026), the Supreme Court held that the law governing EPA's treatment of pesticides, the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), preempts state-level failure-to-warn claims that would require a warning beyond the EPA-approved label. Diffuse harms like those measured here were always difficult to litigate (Shavell 1984), and the ruling makes them harder still. Yet bipartisan public concern persists (Kearney et al. 2026), and the 2025 "Make America Healthy Again" assessment named pesticides among suspected contributors to childhood chronic disease (Make America Healthy Again Commission 2025).

A central question for regulators is how much evidence to require before a chemical is approved for sale, and how much before an approved chemical is restricted or banned. The Food and Drug Administration and the EPA take different approaches. Both review a product before sale based on evidence the manufacturer supplies, but they differ in what that evidence must demonstrate. For the FDA, a new drug must be shown safe and effective in human clinical trials that often span several years. While human trials are not used for pesticides, European regulators place a similar burden on manufacturers to demonstrate the absence of harmful health effects from toxicological data, without weighing economic benefits (Donley 2019).

For the EPA, a pesticide is registered mainly on toxicity tests in rodents and other animals, together with exposure studies, none of which are designed to detect a diffuse harm like the one captured in this paper. Under FIFRA, this process mainly screens out acute or high-dose toxicity in animals and registers a chemical unless its risks are "unreasonable" relative to its economic benefits. The 1996 Food Quality Protection Act added a tenfold margin of safety for infants and children exposed to pesticides, but the margin is applied to risks estimated from those same animal studies. Once registered, a pesticide stays on the market, often for decades (Panel (b) of Figure 1), while evidence of harm accumulates and the EPA builds the case for cancellation.

Defining an evidentiary standard involves a choice between two errors, restricting a safe chemical or allowing a harmful one. Peltzman (1973) examined this trade-off in the context of drug approval. A regulatory regime that registers pesticides primarily on animal tests and withdraws a compound only after evidence of harm accumulates over decades accepts

more of the second type of error relative to the first. The more precautionary standard of the EU (and the FDA) leans the other way. The welfare effects are not obvious and depend on factors including the nature of the compound, its economic value, and the availability of substitutes. Henry et al. (2022) show that lenient approval can be optimal when harm is learned after adoption and the product can be withdrawn. That rationale weakens when the harm cannot be reversed, as with infant deaths, and when the evidence accumulates slowly. Irreversibility then gives quasi-option value to limiting use until that evidence arrives (Arrow and Fisher 1974; Gollier et al. 2000). EPA's own cancellation decisions reveal how it weighs health risks against agricultural benefits (Cropper et al. 1992), and this paper provides causal evidence on the health side of that trade-off.

When a pesticide is withdrawn, growers substitute toward replacements registered under the same evidentiary approach, so a ban does not necessarily lower the overall health cost of insecticide use.[21] Over the course of the seventy-year sample period, successive classes of insecticides replaced one another, yet the infant mortality effect did not fall in tandem, suggesting that newer chemicals have not been meaningfully safer in practice. Given the combination of its reliance on manufacturer-supplied animal toxicity studies under FIFRA, multi-decade cancellation timelines, and the replacement of canceled compounds with ones approved through the same process, the EPA is poorly equipped to detect or protect against the diffuse health costs of pesticides.

These choices especially matter in countries where pesticide use is growing faster than regulatory capacity. Under the 2023 UN Global Framework on Chemicals, governments committed to taking effective measures to phase out highly hazardous pesticides in agriculture where the risks have not been managed and safer, affordable alternatives are available (UNEP 2023), a commitment that depends on knowing whether replacement compounds are in fact safer.

Pesticides bring large benefits to agriculture, but the findings raise concerns about insecticides and the regulatory regime governing their use. The diffuse harms that pre-market testing is least able to detect are also the hardest to measure after approval. This paper shows how ecological phenomena like periodical cicadas can generate the quasi-random variation needed to measure them, and that humans remain beholden to the ancient cicadian rhythm.

[21] A graduated, toxicity-weighted application tax is a more targeted alternative (Zilberman et al. 1991). Unlike a ban, it can limit use while evidence accumulates, although setting it would require compound-specific damage estimates that are difficult to attain.

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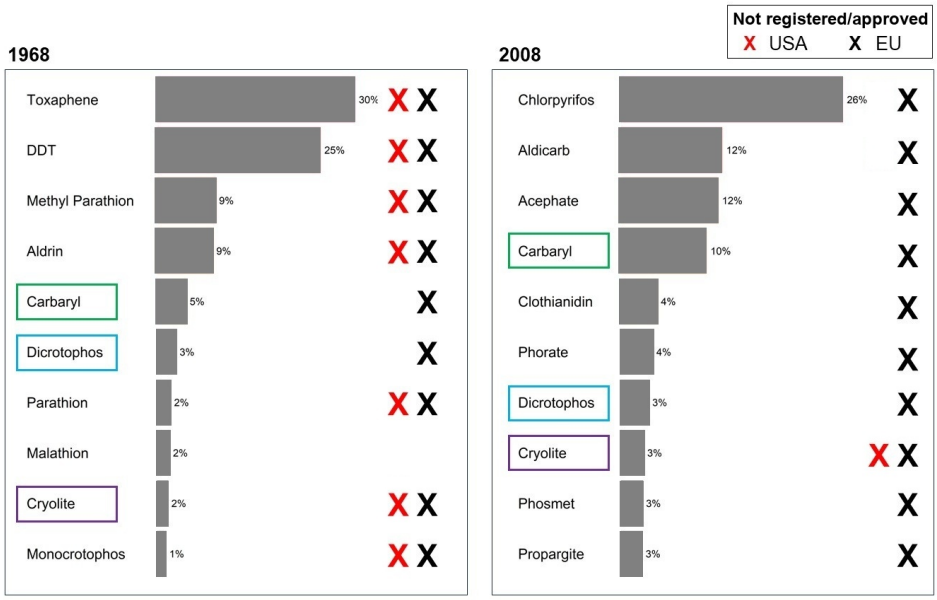
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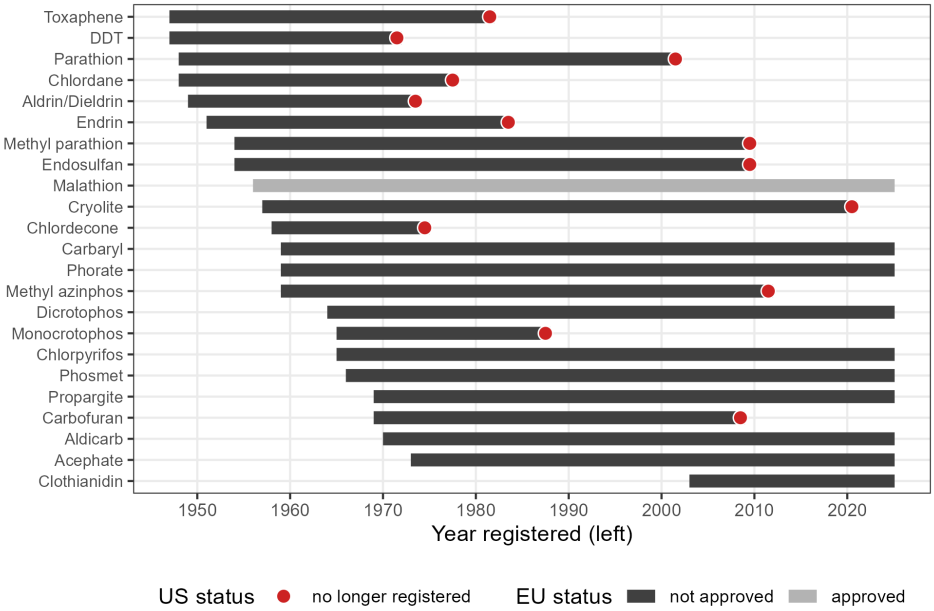
Figures

Figure 1: Insecticide Use and Regulation in the United States

(a) Top 10 insecticides in the US by year

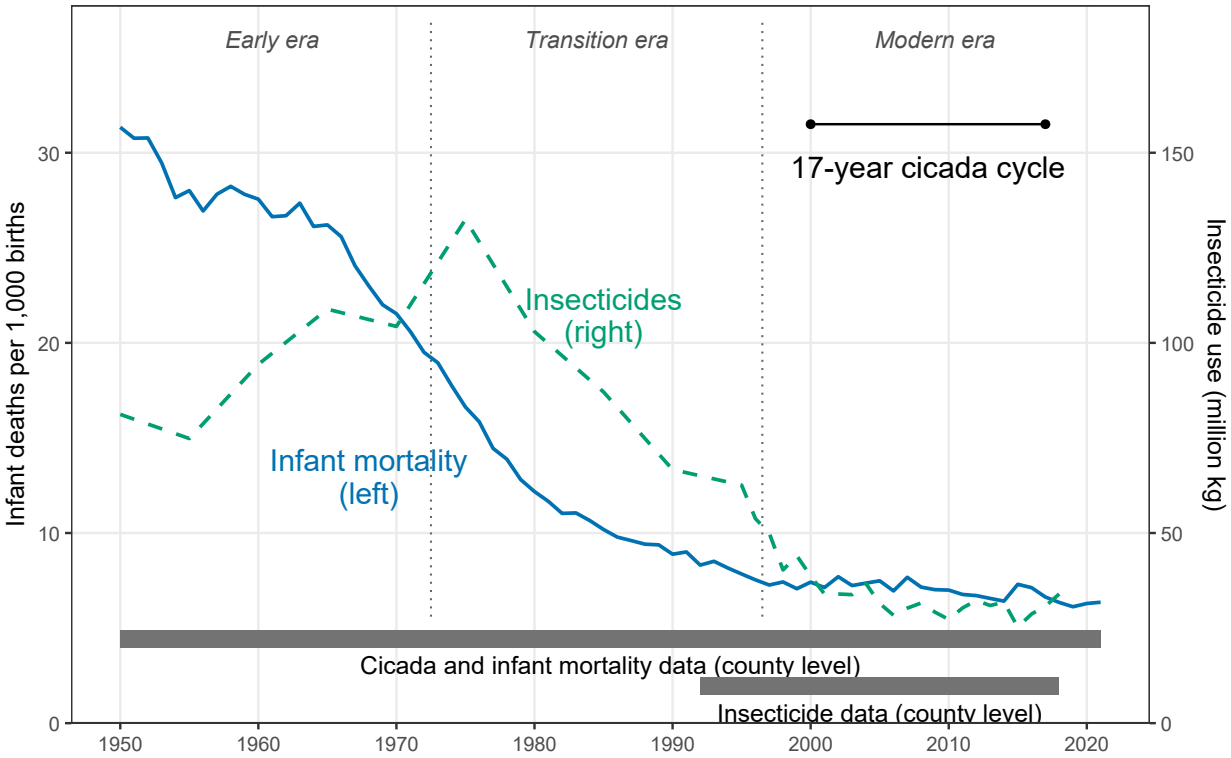


(b) Top insecticides in the US over time



Notes: Panel (a): insecticide use by active ingredient on 21 major crops in 1968 and 2008. A red X marks an insecticide no longer registered in the US and a black X one not approved in the EU. Colored boxes mark insecticides in the top ten in both years. Panel (b): year of first US registration and, with a red dot, the cancellation of major agricultural uses. Data from the EPA Office of Pesticide Programs, USDA (Fernandez-Cornejo et al. 2014), and Donley (2019).

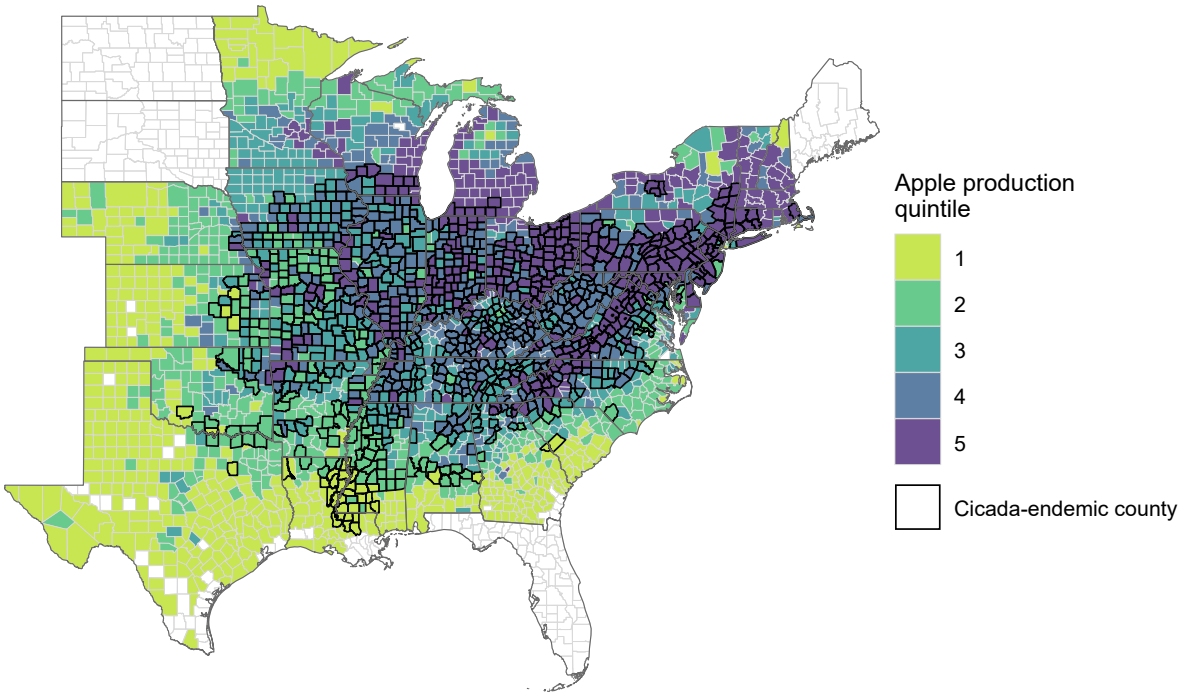
Figure 2: National-Level Trends in Infant Mortality and Insecticide Use



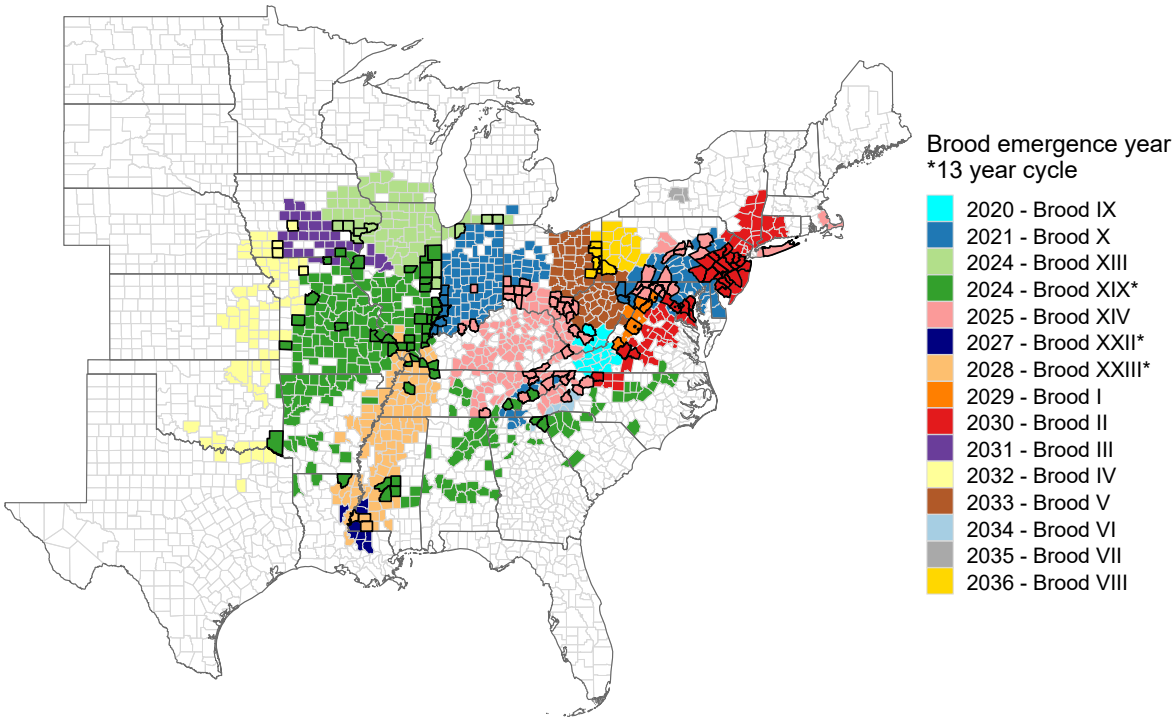
Notes: Annual infant mortality data (NCHS 2019; Bailey et al. 2018). Insecticide use data prior to 1995 come from the EPA in five-year intervals (Aspelin 2003), combined with recent aggregated county-level data from USGS (USGS 2019). Dotted vertical lines separate the three regulatory eras of Section 5.4: through 1972, 1973–1996, and 1997–2020.

Figure 3: Periodical Cicada Broods and Apple Production

(a) Counties by cicada presence and apple intensity



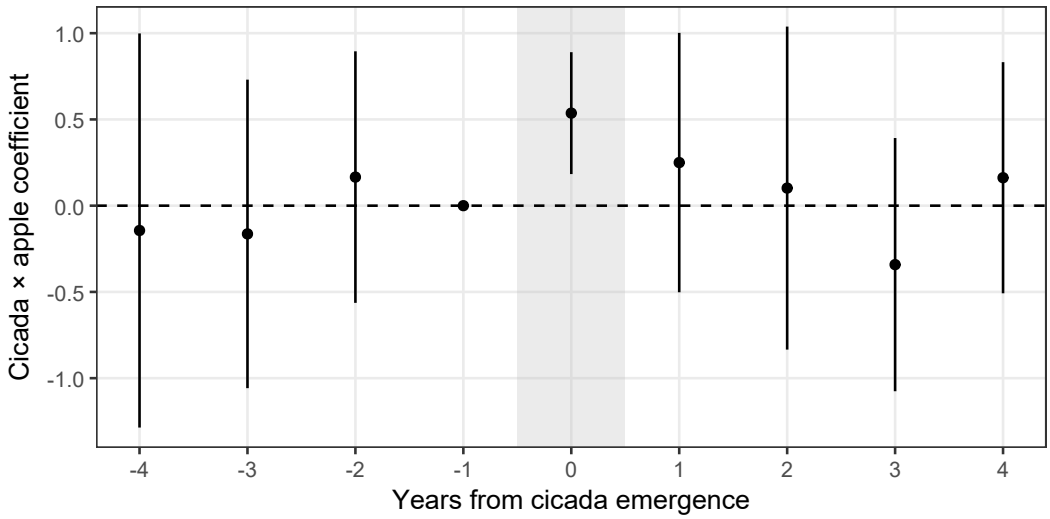
(b) Counties by periodical cicada brood



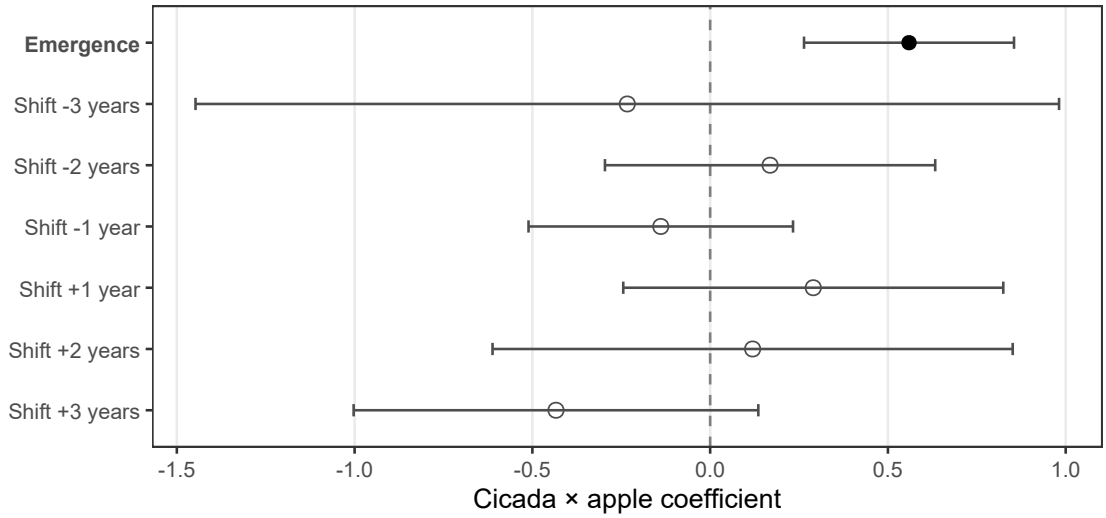
Notes: Panel (a) shows USDA census apple production intensity, averaged across census years. ME, ND, SD, and FL are excluded because they have no cicada broods. In panel (b), outlined counties contain multiple broods. Cicada brood data from the USDA Forest Service Northern Research Station (Liebhold et al. 2013).

Figure 4: Timing of the Infant Mortality Response to Cicada Emergence

(a) Event study



(b) Shifted calendar



Notes: Cicada-apple interaction on next-year infant mortality, in deaths per 1,000 births per thousand bushels per km^2 of apple intensity, based on Model (2) of Table 2 for average apple production. Panel (a) is an event study with event-time indicators from four years before to four years after emergence. Coefficients are relative to $k = -1$, the year before emergence, and the shaded band marks $k = 0$. The sample is limited to cicada-endemic counties and stacked over complete nine-year windows around each cicada emergence such that a county-year enters once for each window. The null hypotheses that the coefficients for $k = -4$ to -2, and for $k = +1$ to $+4$, are jointly equal to zero cannot be rejected ($p = 0.94$ and $p = 0.39$). Panel (b) shifts the emergence calendar from three years prior (-3) to three years after ($+3$) actual emergence across the balanced panel of 2,443 counties over 1950–2020 ($N = 175{,}896$). The top estimate (solid marker) is the actual emergence coefficient from Model (2) of Table 2. The open markers report the same interaction from six separate regressions in which every brood's emergence calendar is shifted by the stated number of years. County and year fixed effects and state-specific linear trends are included. Standard errors are clustered at the state level. Bars show 95% confidence intervals.

Tables

Table 1: Cicada Emergence and Insecticide Use

	Log insecticide use					
	Apple			Top decile of apple		
	(1)	(2)	(3)	(4)	(5)	(6)
Cicada	−0.038	−0.057	−0.051	−0.064	−0.087	−0.082
	(0.045)	(0.049)	(0.049)	(0.046)	(0.051)	(0.050)
Cicada × apple	0.142*	0.156**	0.166**	0.182**	0.206**	0.215**
	(0.071)	(0.074)	(0.078)	(0.080)	(0.085)	(0.081)
R^2	0.869	0.858	0.903	0.869	0.858	0.903
Observations	64,870	64,870	64,870	64,870	64,870	64,870
Year FE	Yes	Yes	Yes	Yes	Yes	Yes
County FE	Yes	Yes	Yes	Yes	Yes	Yes
State linear trend	Yes	No	No	Yes	No	No
County linear trend	No	No	Yes	No	No	Yes

Notes: Dependent variable is the log of county-level insecticide use (USGS EPest-high) in kg per km^2, 1992–2018. Insecticide use sums every compound the classification crosswalk flags as an insecticide, including kaolin clay, lime sulfur and the soil fumigants methyl bromide, chloropicrin and methyl isothiocyanate, and excludes sulfur, petroleum oil and petroleum distillate, EPA's sulfur and oil category. County-years with no EPest record or no recorded insecticide use (1.7% of the balanced 1992–2018 panel) are dropped. Cicada is 1 if a cicada emergence occurs in the county-year. Each regression includes cicada and its interaction with apple, in columns (1)–(3), or with a dummy for the top decile of apple, in columns (4)–(6). Apple is county apple production in thousands of bushels per km^2, averaged across census years 1950–1997. Standard errors are clustered at the state level. $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$

Table 2: Cicada Emergence and Infant Mortality, 1950–2020

	Next-year infant mortality rate			
	(1)	(2)	(3)	(4)
Cicada	0.055	0.029	0.041	0.028
	(0.121)	(0.122)	(0.122)	(0.122)
Cicada × apple		0.559***		
		(0.151)		
Cicada × apple, 1997			0.434**	
			(0.168)	
Cicada × apple, 1964				0.609***
				(0.125)
R^2	0.495	0.495	0.495	0.495
Observations	175,896	175,896	175,896	175,896
Mean of dependent variable	14.66	14.66	14.66	14.66
SD of apple		0.185	0.215	0.168
Year FE	Yes	Yes	Yes	Yes
County FE	Yes	Yes	Yes	Yes
State linear trend	Yes	Yes	Yes	Yes

Notes: Dependent variable is next-year infant mortality rate (deaths per 1,000 live births). Cicada is 1 if a cicada emergence occurs in that county-year. Apple is county apple production in thousands of bushels per km^2, measured as the average across census years 1950–1997 (2), 1997 alone (3), and 1964 alone (4). SD of apple is the cross-county standard deviation of each measure. Years 1950–2020. County and year fixed effects and state-specific linear trends are included. Standard errors are clustered at the state level. $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$

Table 3: Cicada Emergence and Infant Mortality, by Brood

	Next-year infant mortality rate					
	All broods	Brood II	Brood V	Brood X	Brood XIII	Brood XIV
	(1)	(2)	(3)	(4)	(5)	(6)
Cicada	0.029	0.366	0.275**	0.302**	−0.175	0.173
	(0.122)	(0.216)	(0.106)	(0.117)	(0.133)	(0.128)
Cicada × apple	0.559***	0.567***	0.526***	0.389**	0.237	0.496***
	(0.151)	(0.136)	(0.130)	(0.151)	(0.205)	(0.159)
R^2	0.495	0.386	0.291	0.601	0.577	0.446
Observations	175,896	20,376	14,328	46,440	15,768	43,992
Year FE	Yes	Yes	Yes	Yes	Yes	Yes
County FE	Yes	Yes	Yes	Yes	Yes	Yes
State linear trend	Yes	Yes	Yes	Yes	Yes	Yes

Notes: Column (1) pools all broods and reproduces Model (2) of Table 2, and columns (2)–(6) restrict to each brood's counties plus neighboring control counties within approximately 100 km, deduplicated to unique county-years. Dependent variable is next-year infant mortality rate (deaths per 1,000 live births). Cicada is 1 if a cicada emergence occurs in the county-year. Apple is county apple production in thousands of bushels per km^2, averaged across census years. County and year fixed effects and state-specific linear trends are included. Standard errors are clustered at the state level, with 33 clusters in column (1) and between 7 and 19 clusters in columns (2)–(6). $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$

Table 4: Cicada Emergence and Infant Mortality, by Regulatory Era

	Infant mortality rate (levels)		Infant deaths (proportional)	
	Pooled	By era	Poisson	Log deaths
	(1)	(2)	(3)	(4)
	OLS	OLS	Poisson	OLS
Cicada	0.029			
	(0.122)			
Cicada × apple	0.559***			
	(0.151)			
× Early era (≤1972)		0.689	0.029	0.043***
		(0.426)	(0.023)	(0.013)
× Transition era (1973–1996)		0.347	−0.009	0.016
		(0.472)	(0.037)	(0.039)
× Modern era (1997–2020)		0.644**	0.060*	0.102***
		(0.256)	(0.036)	(0.030)
Modern = early era, p		0.936	0.468	0.097
Equal slopes across eras, p		0.801	0.352	0.021
Observations	175,896	175,896	175,896	152,877
Year FE	Yes	Yes	Yes	Yes
County FE	Yes	Yes	Yes	Yes
State linear trend	Yes	Yes	Yes	Yes

Notes: Column (1) is the pooled specification, Model (2) of Table 2, and column (2) interacts the cicada-apple term with three regulatory eras within a single model. The dependent variable in columns (1) and (2) is the next-year infant mortality rate (deaths per 1,000 live births). Eras are assigned by emergence year. Columns (3) and (4) estimate the same era model in proportional terms. Column (3) is a Poisson model of next-year infant deaths with log births as an offset (its coefficient fixed at one), so it models the mortality rate and its coefficients approximate the proportional change in that rate per unit of apple intensity. It weights county-years by expected deaths, so larger counties count for more than in the other columns, which weight county-years equally. Column (4) regresses log deaths on the same terms, controls for log births, and drops county-years with no infant deaths: 3% in the early era, 14% in the transition era, and 22% in the modern era. Mean next-year infant mortality in top-decile apple counties is 22.9, 9.9, and 6.1 deaths per 1,000 births in the three eras. The by-era models also include era-specific cicada main effects (not shown). The last two rows report Wald tests that the modern-era and early-era cicada-apple slopes are equal and that all three are equal. County and year fixed effects and state-specific linear trends are included. Years 1950–2020. Standard errors are clustered at the state level. $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$

Online Appendix

Cicadian Rhythm: Insecticides and Infant Health over Seven Decades

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List of Figures

A1 Aggregate Pesticide Use and Characteristics, 1960–2010 . . . . . A7
A2 Insecticide Use and Composition over Time A8
A3 Apple Intensity: Census versus Cropland Data Layer A9
A4 County Infant Mortality, 1950–2021 . . . . . A10
A5 Public Salience of Cicada Emergences: Google Trends Search Interest A11
A6 Timing Framework A12
A7 Timing of the Insecticide Response to Cicada Emergence A14
A8 Brood-Specific Event Study of Infant Mortality A15
A9 Infant Mortality Estimate over Time (Restricted Cubic Spline) A16
A10 Cicada Emergence and Third-Grade Math Scores: Event Study A18
A11 Inference: Leave-One-Out and Randomization A19
A12 Cicada Emergence and Apple Production: Event Study A24

List of Tables

A1 County-Level Summary Statistics, 1950–2020 . . . . . A3
A2 Pesticide Response by Class and Compound A13
A3 Cicada Emergence and Elementary School Test Scores A17
A4 Inference for the Infant Mortality Estimate A20
A5 Specification Sensitivity for Infant Mortality A21
A6 Alternative Treatment Measures and Land-Use Placebos A22
A7 Alternative Cicada Brood Map, Mid-Atlantic States A23
A8 Cicada Emergence and Farm Income A25

A9 Maternal Composition and Fertility A26
A10 Cicada Emergence and Infant Mortality, by Cause of Death, 1968–2020 . . . A27

A. Data and measurement

Table A1: County-Level Summary Statistics, 1950–2020

	Mean	SD	Min	Max	Observations
Cicada emergence	0.03	0.17	0	1	175,896
Area (km^2)	1,621	1,076	5	16,181	175,896
Births per year	1,073	3,253	1	127,338	175,896
Infant deaths per year	15.8	60.0	0	3,421	175,896
Next-year infant mortality rate	14.66	13.36	0	1,000	175,896
Apple bushels, 1964	30.8	203.9	0	5,336.7	175,896
Apple bushels, 1997	32.0	305.8	0	8,388.5	175,896
Apple acres, 2002	85	656	0	19,590	175,896
Apple bushels/km^2, 1964	0.02	0.17	0	3.41	175,896
Apple bushels/km^2, 1997	0.02	0.22	0	5.36	175,896
Apple bushels/km^2, average	0.03	0.18	0	4.16	175,896
Insecticides (kg)	11,333	34,307	0	3,006,761	65,155
Insecticides (kg/km^2)	7.84	23.99	0	2,122.19	65,155

Notes: County-year observations for the balanced sample of eastern US counties used in the analysis, 1950–2020. Apple bushels are in thousands, and per-area apple measures are normalized by county land area. Insecticide use excludes sulfur, petroleum oil, and petroleum distillate. USGS EPest-high estimates for 1992–2018 are reported in kilograms per county and kilograms per square kilometer. The infant mortality rate is in deaths per 1,000 live births.

Infant mortality construction

Infant mortality data come directly from the US National Center for Health Statistics (NCHS) (NCHS 2019) as well as a number of associated sources like Inter-university Consortium for Political and Social Research (ICPSR).

For early time periods, I use ICPSR's County-Level Natality and Mortality Data, 1915–2007 (Bailey et al. 2018). The ICPSR data are averaged annually and do not allow for within-year or demographic disaggregation aside from race. I use ICPSR's preferred 'fixed' variables whenever available. ICPSR's resident infant death data begin in 1942 and are based on the mother's county of residence rather than the county of birth. I incorporate data from NCHS vital statistics after 1988 when ICPSR masks infant deaths in counties with populations of less than 100,000 (its birth counts remain available for all counties through 2007), which is challenging given that many high apple production counties of interest are rural and low population.

Infant deaths come from the NCHS Mortality Multiple Cause-of-Death Data files. From 1959 to 1988, I use the CSV files constructed by NBER from the raw NCHS data (National Bureau of Economic Research 2026). From 1989 to 2021, I use restricted NCHS mortality data, which contain geographic identifiers at the county level for all US counties.

To construct a measure of infant mortality, I filter the mortality records to deaths under age 1 to get a count of infant deaths. I then link this to birth count data from either ICPSR (through 2007) or the denominator file of births in the NCHS Linked Birth-Infant Death Data (after 2007). Note that in 1972, NCHS processed 50% of death records, so I double the number of deaths in each county to compute infant mortality. With this information, I compute county-level infant mortality for each year from 1959 to 2021 by dividing the number of deaths by the number of births and multiplying by 1,000. The core measure of infant mortality is extended back further in time by combining it with county-level infant mortality data from ICPSR, which is available from 1942 to 1958.

Pesticide trends

Among the main pesticide types, herbicide use increased rapidly through the early 1980s, and then stabilized before increasing again in the 2000s following the mass adoption of genetically engineered, herbicide-tolerant crops and no-till agriculture. Insecticide use, on the other hand, which exceeded herbicide use for much of the 1960s, has been in decline since the mid-1970s (Figure A1).

In terms of quality characteristics, average pesticide application rates (i.e., pounds per acre over a year) declined by more than half since the late 1960s, as shown in Figure A1, implying that pesticide efficiency increased starting in the 1970s. Potential impacts on humans and the environment, however, are a function of pesticide toxicity and persistence in addition to the quantity applied. To this end, average toxicity[22] declined rapidly in the 1970s driven by the bans of DDT and aldrin (primarily used on corn), combined with increasing use of relatively less toxic insecticides like carbaryl, chlorpyrifos, and pyrethroids (Fernandez-Cornejo et al. 2014).

Average pesticide persistence fell from 54% in 1968 to 25% in 2008, defined as the share of pesticides with a half-life greater than 60 days (Fernandez-Cornejo and Jans 1995; Fernandez-Cornejo et al. 2014). The 1970s saw a notable decrease in persistence following the removal of DDT and aldrin from the market, followed by an increase in the early 1990s coinciding with longer-lasting metolachlor and pendimethalin. Average persistence then declined starting in the late 1990s following the large-scale adoption of glyphosate herbicide (Fernandez-Cornejo et al. 2014). Figure A1 shows that despite the increase in pesticide efficiency and change in pesticide composition, pesticide prices stayed relatively in line with average crop prices over the decades, with some periods of lower relative prices in the 1970s and 1980s and higher prices in the late 1990s and early 2000s. But relative to fuel and labor input costs, pesticides have become far less expensive over time.

To visualize the effect of regulation and technological change over time, Figure A2 plots long-term trends in insecticide use by active ingredient type, merging historical data at the national level (Aspelin 2003) with the more recent United States Geological Survey (USGS) county-level data used in this paper (USGS 2019). Similar to Figure A1, aggregate insecticide use peaks in 1975, then declines, but in terms of chemical type, there is significant variation. Insecticide use is dominated by inorganics like arsenicals until about 1950, at which point the synthetic chemical age begins with organochlorines like DDT driving insecticide growth. In the late 1960s, organophosphates begin to replace organochlorines (DDT was banned in

[22] Toxicity refers to the chemical's risk to humans and the environment, and is estimated based on the inverse of the safe drinking water threshold (Kellogg et al. 2002) in terms of constituent concentration in parts per billion.

1972). Organophosphates, in turn, are replaced by other types of insecticides starting in the early 1990s, coinciding with increases in pyrethroid and neonicotinoid insecticide use as carbamates are phased out.

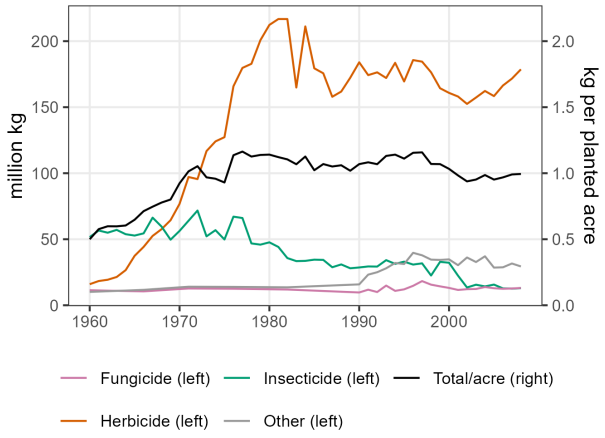
Such patterns reflect the increasing proportion of organochlorines, then organophosphates, and then carbamates that were subject to EPA restrictions over time.[23]

Historical crop-specific estimates of insecticide use are often imputed for a given geography based on overall pesticide consumption and crop-specific acreages. Looking at five major US crops, panel (a) of Figure A2 plots insecticide use trends with total quantity on the left and intensity (kg per acre) on the right. Cotton accounts for much of historical insecticide use, and its decline can be attributed to several factors: the successful USDA Boll Weevil Eradication Program that began in the early 1970s and reduced insecticide use in cotton by 40 to 80% (Smith 1998), and the introduction of Bt cotton in the mid-1990s, which was genetically engineered for pest resistance.

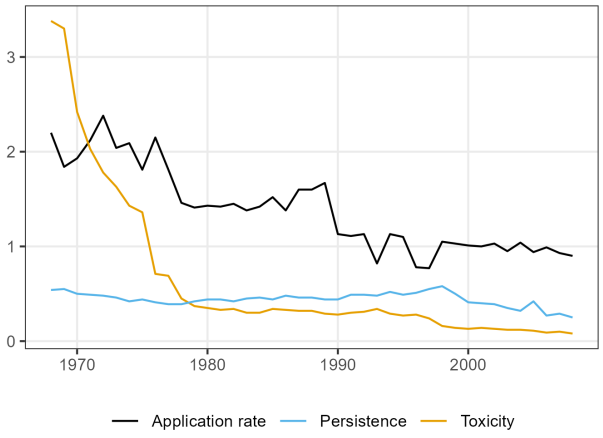
[23] USGS data allows for some disaggregation by chemical type and land use at the state level. Figure A2 sums insecticide use in the 'Orchards and grapes' category, which includes apple production, across the eastern states in this study. Patterns are broadly consistent with the national trends since 1993, shown in Figure A1 in which insecticide use is declining overall. However, organophosphates still accounted for the majority of orchard insecticide use through 2018.

Figure A1: Aggregate Pesticide Use and Characteristics, 1960–2010

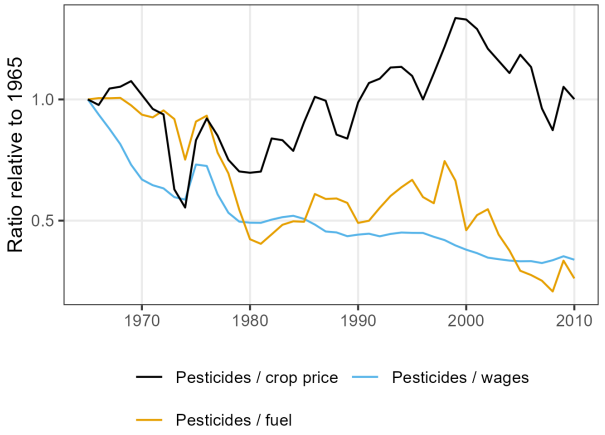
(a) Aggregate use and intensity



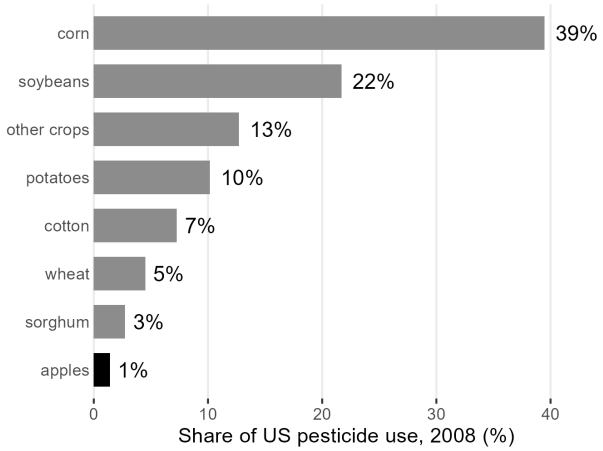
(b) Characteristic trends



(c) Price trends



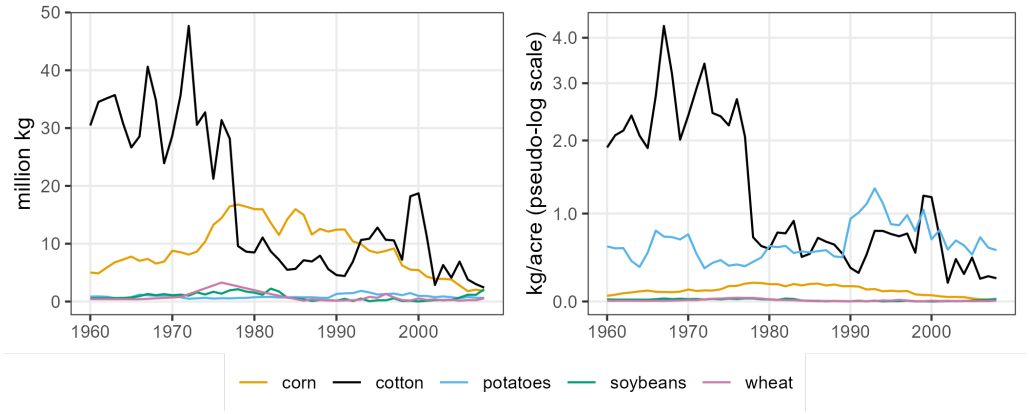
(d) Use by crop, 2008



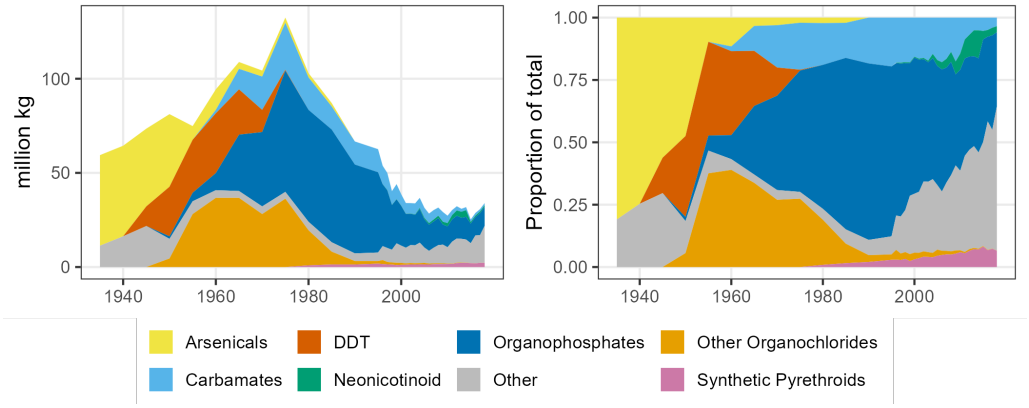
Notes: Data from USDA for the top 21 crops (Fernandez-Cornejo et al. 2014). Panel (a) aggregate pesticide use and intensity. Other pesticides include soil fumigants, desiccants, harvest aids, and plant growth regulators. Panel (b) use-weighted characteristics for corn, soybeans, cotton, and sorghum: application rate (pounds of active ingredient per acre per application times applications per year), toxicity (inverse of the average safe drinking-water threshold, Kellogg et al. 2002), and persistence (share of pesticides with a half-life greater than 60 days, Fernandez-Cornejo and Jans 1995). Panel (c) NASS pesticide and crop price indices. Panel (d) pesticide use by crop in 2008, measured by kg of active ingredient.

Figure A2: Insecticide Use and Composition over Time

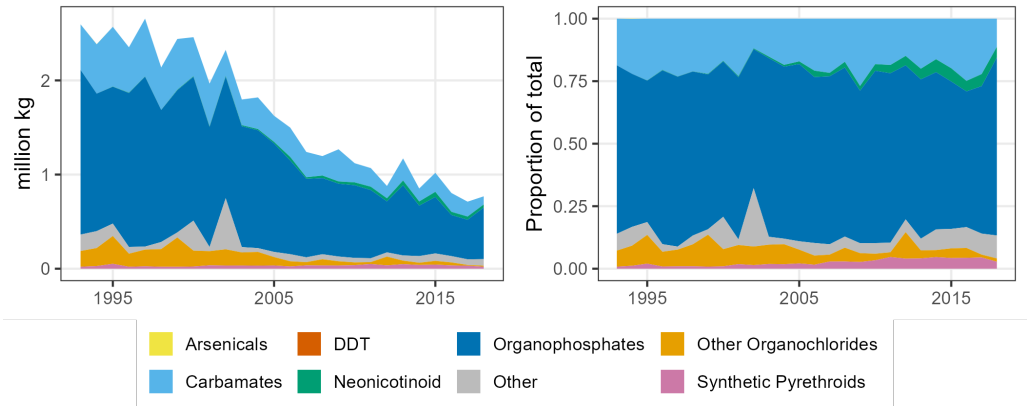
(a) Insecticide use by crop: quantity (left) and intensity (right)



(b) National, all land uses, 1935–2018



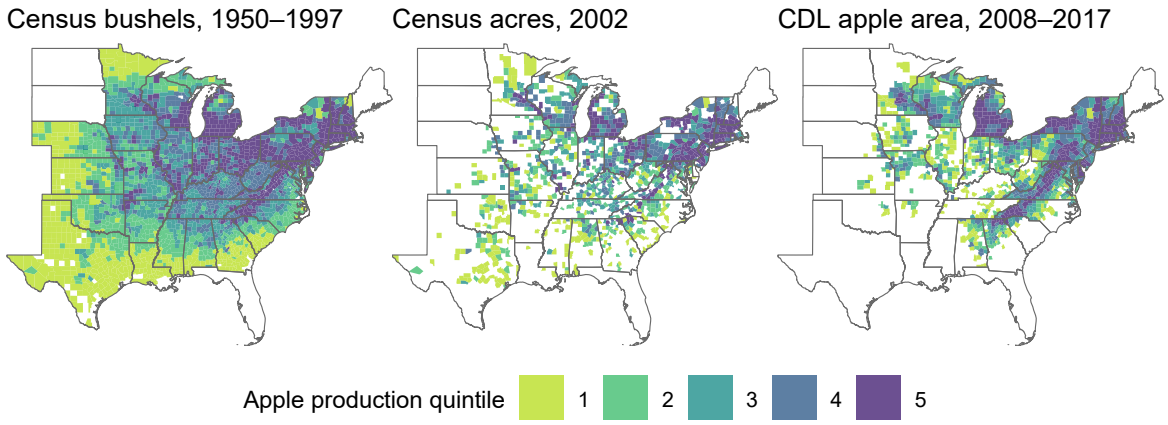
(c) Orchards and grapes, 1993–2018



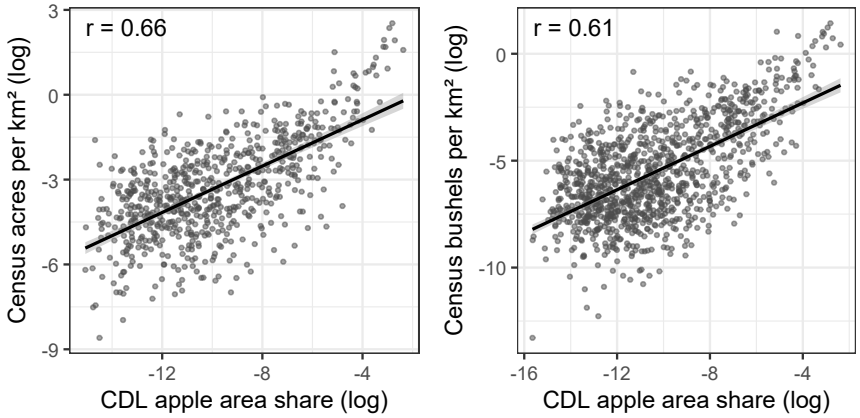
Notes: Panel (a): insecticide use for five major crops from USDA (Fernandez-Cornejo et al. 2014). Panels (b)–(c): insecticide use by active-ingredient class, national five-year-interval EPA data through 1995 (Aspelin 2003) combined with recent aggregated USGS county-level data (USGS 2019). Pesticide categorizations adjusted so the datasets are comparable. Panel (b) covers all land uses nationally. Panel (c) sums state-level "orchards and grapes" across the eastern sample states.

Figure A3: Apple Intensity: Census versus Cropland Data Layer

(a) Maps: Census bushels, Census acres, and CDL area



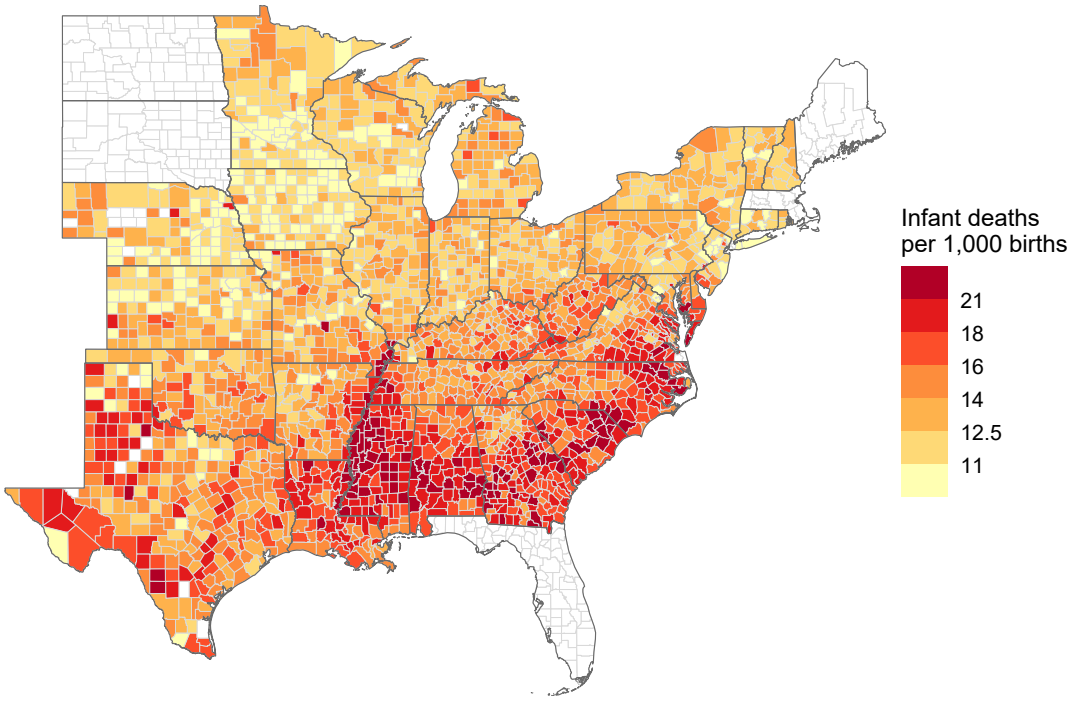
(b) CDL vs. Census: acreage (left) and bushels (right)



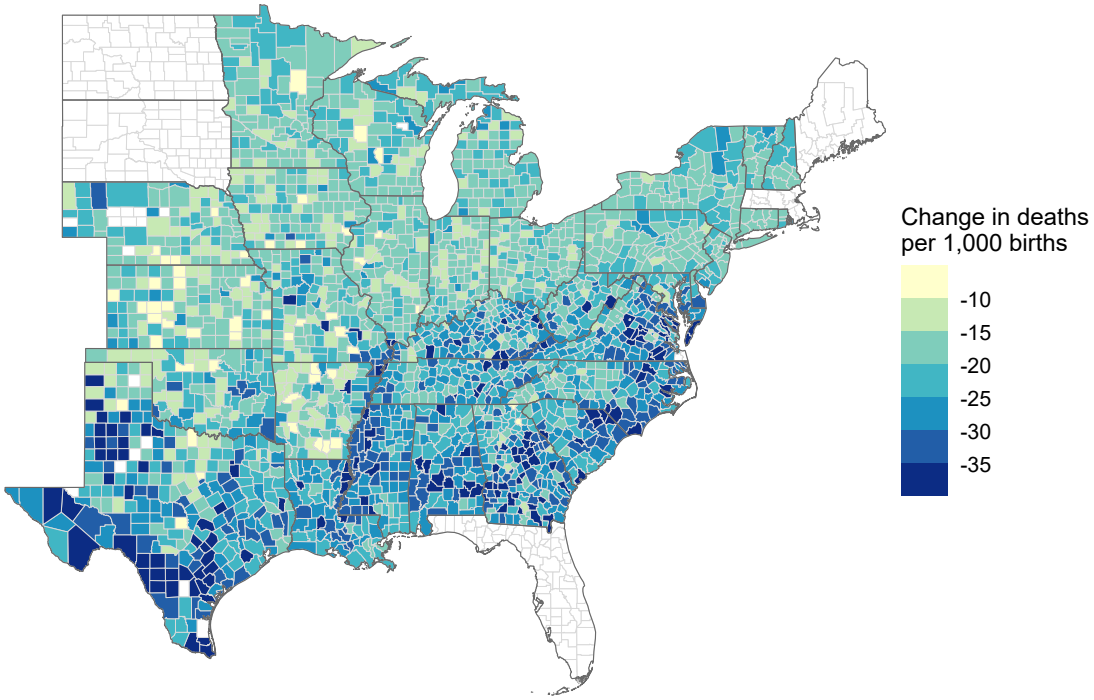
Notes: Panel (a): county apple intensity from three sources, left to right: USDA census apple production in bushels averaged across census years 1950 to 1997, the baseline measure, and apple acreage in 2002, each per km^2 of county land area, and the Cropland Data Layer apple share of total county area, 2008–2017. Counties are shaded by quintile of the nonzero values of each measure, and counties with none are white. Florida, Maine, North Dakota, and South Dakota are outside the sample and left blank. Panel (b): census apple acres (left) and bushels (right) per km^2 against the CDL apple share of total county area, in logs, for counties where both are nonzero.

Figure A4: County Infant Mortality, 1950–2021

(a) Average infant mortality rate, 1950–2021

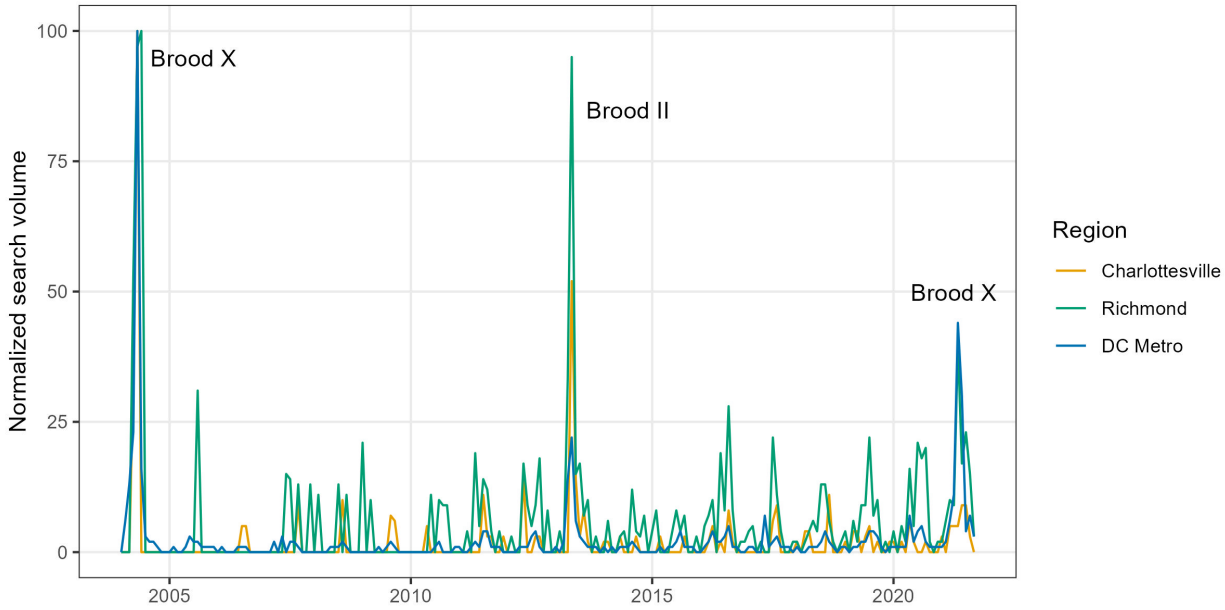


(b) Change, 1950–1959 to 2012–2021



Notes: County mean infant mortality rates for the 2,443 eastern-US counties used in the main analysis. Counties outside this sample are white. Infant mortality data from the National Center for Health Statistics and ICPSR (NCHS 2019; Bailey et al. 2018).

Figure A5: Public Salience of Cicada Emergences: Google Trends Search Interest



Notes: Average monthly Google Trends search interest for the term "cicada" in three Virginia metropolitan regions (Charlottesville, Richmond, and the Washington, DC area), normalized within each region. The Google Trends series begins in 2004.

B. Additional results

Figure A6: Timing Framework






	Years before emergence	Emergence, May–June	Rest of summer / fall	Following year
Cicadas	Nymphs feed on tree roots, then tunnel toward the surface in the final spring	Adults emerge, mate, and lay eggs in twigs for about a month	Eggs hatch after about six weeks, and nymphs burrow to the roots	Nymphs feed on roots underground for the next 13 or 17 years
Crop damage	Root feeding slows tree growth, which may lower yields	Egg-laying slits kill branch tips, most severely on young trees	Damaged branch tips die back	Lost branches and young trees can lower later yields
Insecticide use		Repeated spraying while adults are active and laying eggs		
Infant health		Exposure of pregnant women	*pregnancy continues* →	Births from first-trimester exposures; infant mortality measured

Table A2: Pesticide Response by Class and Compound

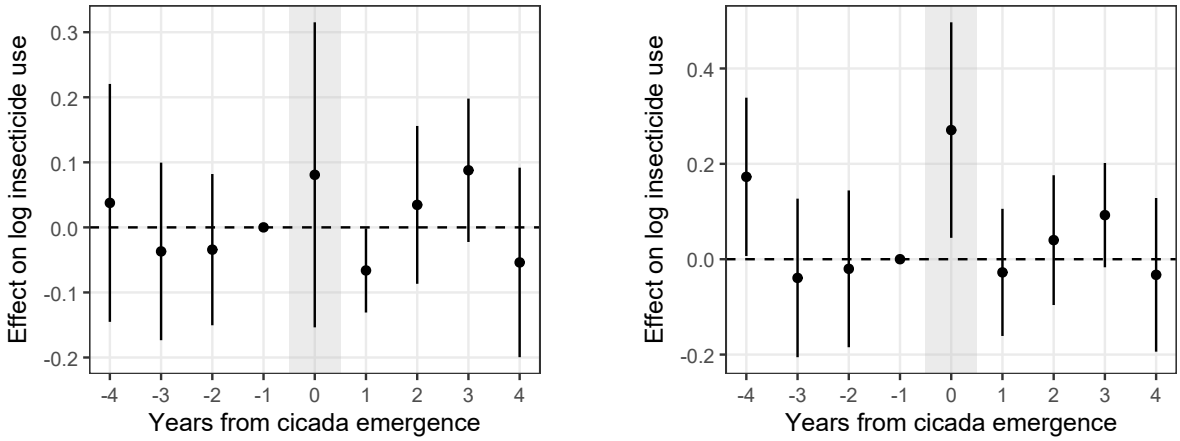
	Log pesticide use					
	Apple			Top decile of apple		
	(1)	(2)	(3)	(4)	(5)	(6)
Panel A. Insecticide						
Baseline	0.142*	0.156**	0.166**	0.182**	0.206**	0.215**
	(0.071)	(0.074)	(0.078)	(0.080)	(0.085)	(0.081)
Baseline, plus sulfur and oils	0.116**	0.131**	0.132**	0.127*	0.149**	0.154**
	(0.049)	(0.052)	(0.056)	(0.067)	(0.070)	(0.066)
Baseline, ex. soil fumigants	0.102*	0.112*	0.119*	0.051	0.069	0.078*
	(0.055)	(0.057)	(0.061)	(0.043)	(0.046)	(0.044)
Carbaryl	0.204	0.214*	0.201	0.282**	0.291***	0.284**
	(0.122)	(0.121)	(0.127)	(0.106)	(0.102)	(0.105)
Panel B. Herbicide						
Baseline	−0.028	−0.030	−0.015	0.068*	0.071	0.074*
	(0.033)	(0.034)	(0.027)	(0.039)	(0.042)	(0.038)
Baseline, ex. soil fumigants	−0.020	−0.021	−0.014	0.047	0.048	0.049*
	(0.031)	(0.029)	(0.027)	(0.029)	(0.032)	(0.028)
Panel C. Fungicide						
Baseline	0.164**	0.181**	0.187**	0.227**	0.255**	0.262***
	(0.070)	(0.077)	(0.072)	(0.086)	(0.095)	(0.084)
Baseline, ex. soil fumigants	0.126**	0.139**	0.147***	0.102	0.119*	0.134**
	(0.050)	(0.052)	(0.047)	(0.062)	(0.063)	(0.061)
Year FE	Yes	Yes	Yes	Yes	Yes	Yes
County FE	Yes	Yes	Yes	Yes	Yes	Yes
State linear trend	Yes	No	No	Yes	No	No
County linear trend	No	No	Yes	No	No	Yes

Notes: Each cell reports the cicada × apple interaction from a separate regression of log county use (USGS EPest-high, kg per km^2, 1992–2018), specified as in Table 1 and including cicada. Baseline aggregates are constructed as in Table 1, and the insecticide baseline row repeats that table. They include soil fumigants and exclude sulfur, petroleum oil and petroleum distillate, EPA's sulfur and oil category. The plus sulfur and oils row adds these three compounds back to the insecticide aggregate. Rows marked ex. soil fumigants remove methyl bromide, chloropicrin, methyl isothiocyanate, metam, metam potassium, dazomet, dimethyl disulfide, tetrathiocarbonate, methyl iodide and allyl isothiocyanate from the aggregate wherever they carry its class flag. Dichloropropene, also a soil fumigant, carries no class flag and is in no aggregate. The sample is that of Table 1, with 64,870 observations in every cell except carbaryl (59,943) and fungicide (64,514, and 64,508 ex. soil fumigants), which drop county-years with no recorded use. Standard errors are clustered at the state level. $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$

Figure A7: Timing of the Insecticide Response to Cicada Emergence

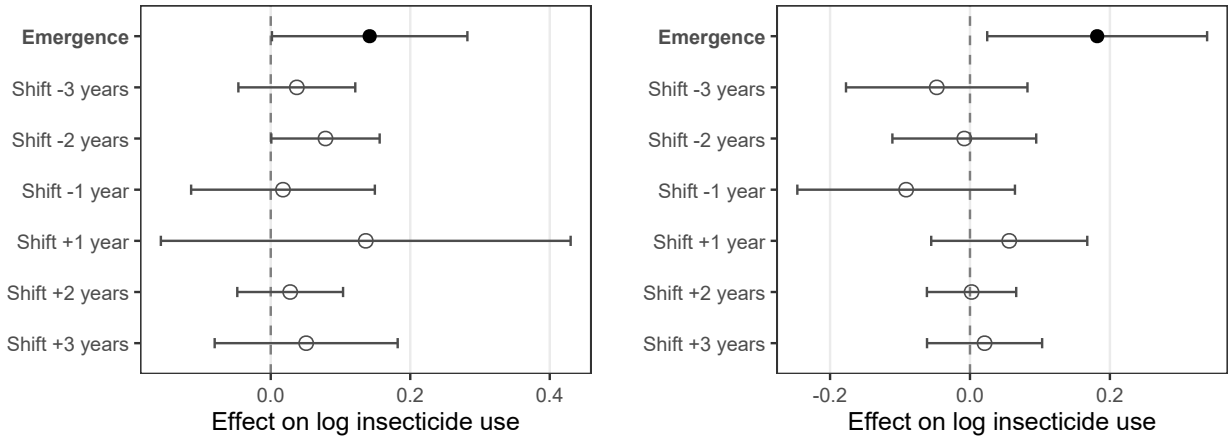
(a) Event study, apple intensity

(b) Event study, top decile



(c) Shifted calendar, apple intensity

(d) Shifted calendar, top decile

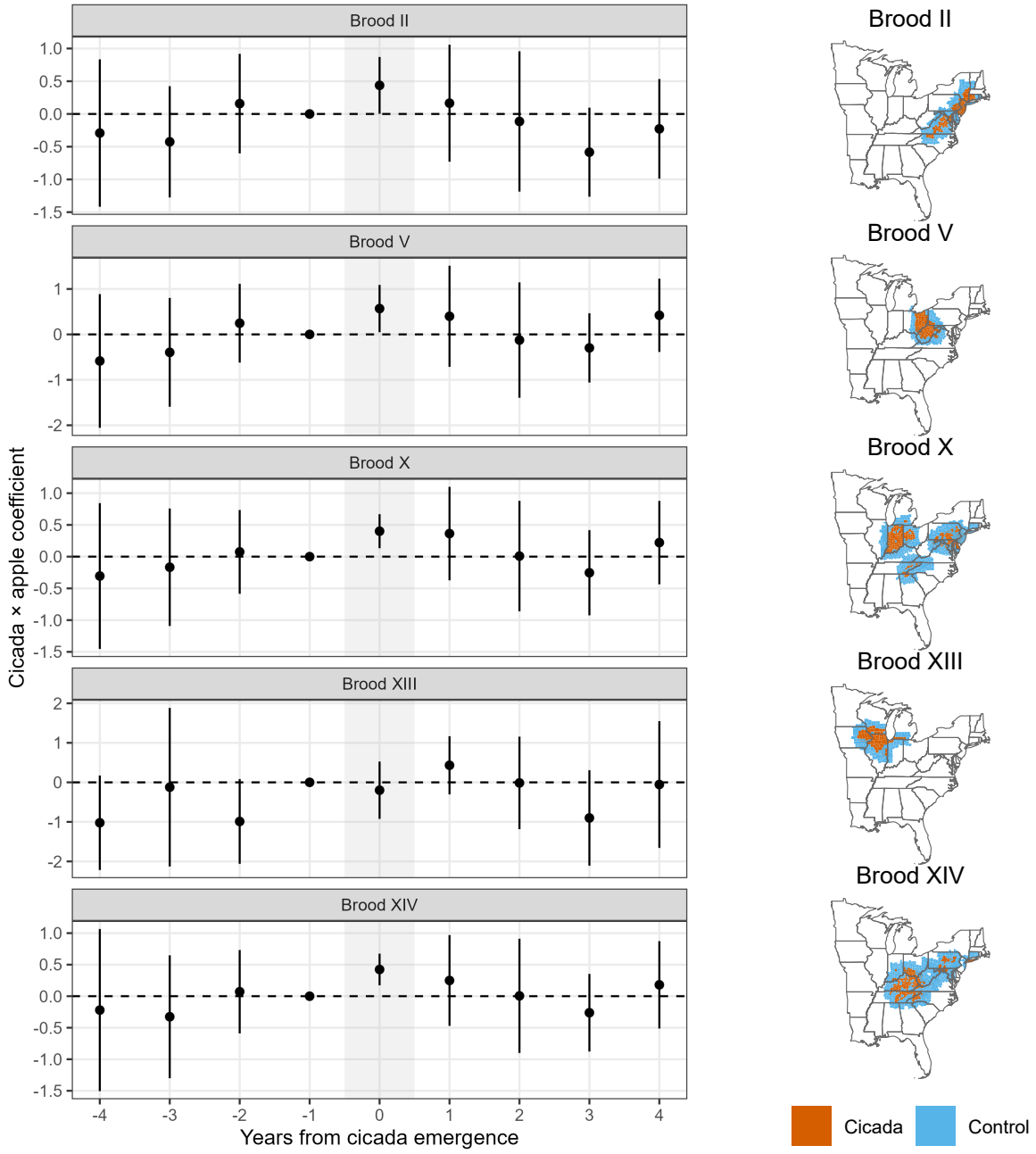


Notes: Coefficients on the interaction of cicada emergence with apple intensity (left) or a dummy for the top decile of apple intensity (right), for log insecticide use as defined in Table 1 (USGS county-level data, 1992–2018, USGS 2019). Panels (a) and (b) are event studies with leads and lags of four years, estimated on complete stacked nine-year windows around each emergence in cicada-endemic counties. Incomplete windows (such as those extending before 1992 or after 2018) are dropped, and a county-year enters once for each window. Coefficients are relative to $k = -1$, the year before cicada emergence, and the shaded band marks the emergence year $k = 0$. Joint tests of the null hypothesis that the three pre-emergence coefficients are equal to zero give $p = 0.45$ in panel (a) and $p = 0.03$ in panel (b). Panels (c) and (d) shift the entire emergence calendar by one to three years and re-estimate columns (1) and (4) of Table 1 on the full 1992–2018 panel, so each row is a separate regression ($N = 64{,}870$). County and year fixed effects and state-specific linear trends are included. Standard errors are clustered at the state level. Bars show 95% confidence intervals.

Figure A8: Brood-Specific Event Study of Infant Mortality

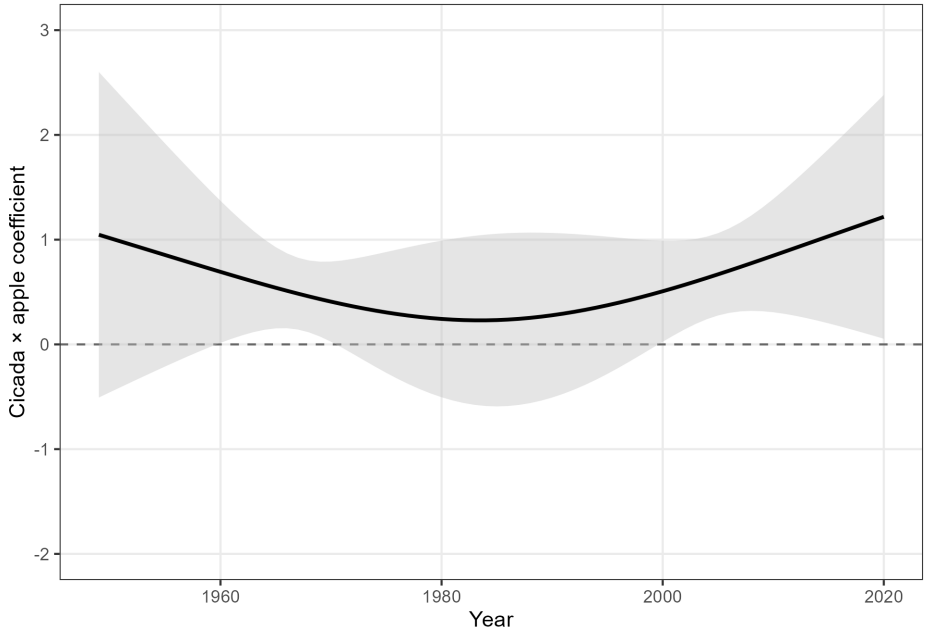
(a) Event studies by brood

(b) Treatment and control counties



Notes: Panel (a): event-study coefficients on the cicada × apple interaction from brood-specific regressions, in deaths per 1,000 births per thousand bushels per km^2. Coefficients are relative to $k = -1$, the year before emergence, and the shaded band marks $k = 0$. Each brood's sample includes counties in the brood's range plus neighboring controls within approximately 100 km. Events are defined as any cicada emergence within this geographic subset, stacked over complete nine-year windows around each emergence such that a county-year enters once for each window. County and year fixed effects and state-specific linear trends are included. Standard errors are clustered at the state level. Bars show 95% confidence intervals. Panel (b): treatment (orange) and control (blue) counties for each brood.

Figure A9: Infant Mortality Estimate over Time (Restricted Cubic Spline)



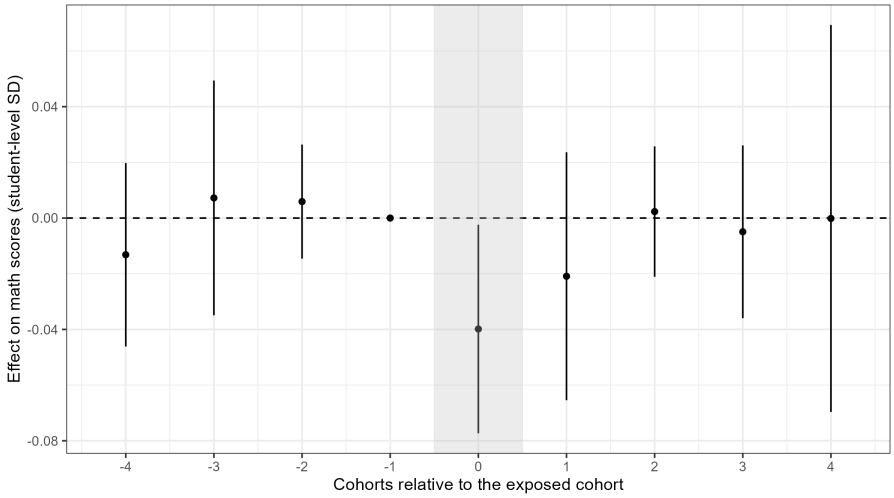
Notes: Marginal effect of the cicada-apple interaction on next-year infant mortality, in deaths per 1,000 births per thousand bushels per km^2, allowed to vary by year through a restricted cubic spline (three knots), as in Model (2) of Table 2 with the apple-cicada interaction interacted with the year spline. Shaded bands are 95% confidence intervals computed by the delta method. County and year fixed effects and state-specific linear trends are included. Standard errors are clustered at the state level.

Table A3: Cicada Emergence and Elementary School Test Scores

	Overall		Math		English	
	(1)	(2)	(3)	(4)	(5)	(6)
Cicada	−0.009	−0.004	−0.015	−0.010	−0.0004	0.003
	(0.009)	(0.010)	(0.012)	(0.012)	(0.009)	(0.011)
Cicada × apple	−0.031*		−0.039**		−0.010	
	(0.016)		(0.015)		(0.012)	
Cicada × apple, top decile		−0.030***		−0.038***		−0.021*
		(0.011)		(0.011)		(0.012)
R^2	0.801	0.801	0.761	0.761	0.773	0.773
Observations	22,724	22,724	25,447	25,447	23,010	23,010
Year FE	Yes	Yes	Yes	Yes	Yes	Yes
County FE	Yes	Yes	Yes	Yes	Yes	Yes
State linear trend	Yes	Yes	Yes	Yes	Yes	Yes

Notes: Dependent variable is county-level averages of Stanford Education Data Archive third-grade test scores, matched to whether the cohort was exposed to a cicada event in the year before the average birth year (test year minus 10). Annual scores available 2009–2019. Scores are in student-level standard deviations. Each column re-estimates Model (2) of Table 2 with the test score as the outcome. Columns (1), (3), and (5) use the same continuous apple measure, and columns (2), (4), and (6) the top-decile indicator. Overall is the average of the county's math and English scores. Observation counts differ across columns because SEDA coverage varies by subject and Overall requires both. County and year fixed effects and state-specific linear trends are included. Standard errors are clustered at the state level. $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$

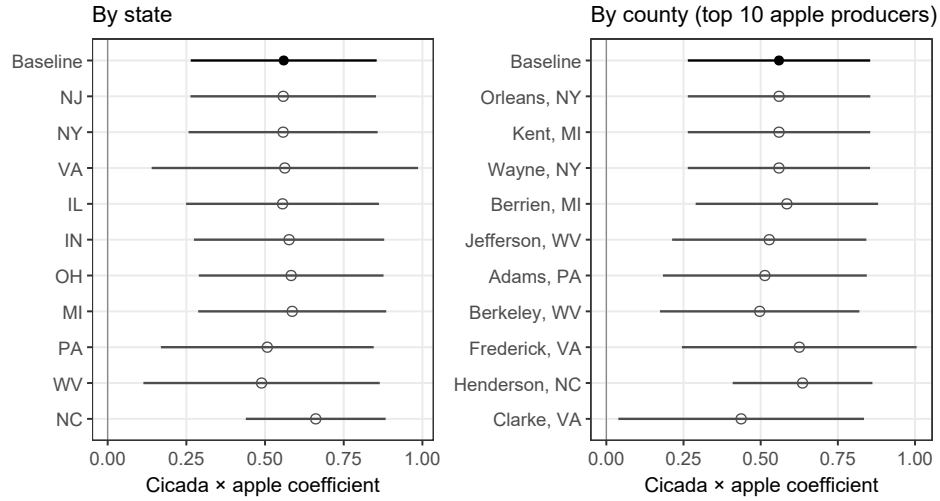
Figure A10: Cicada Emergence and Third-Grade Math Scores: Event Study



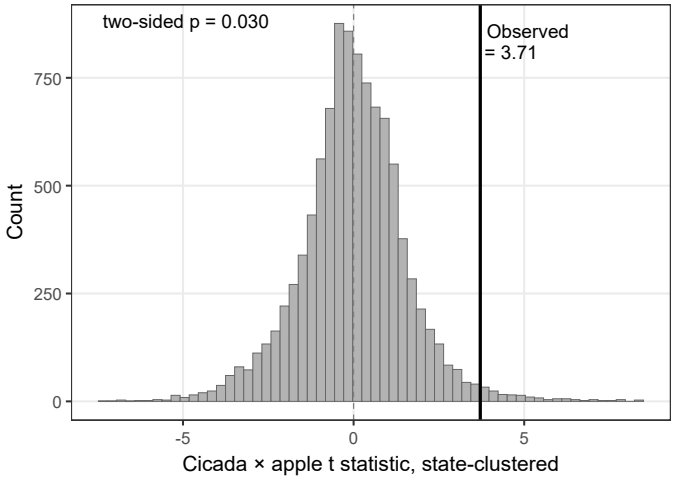
Notes: Event study of third-grade math scores (Stanford Education Data Archive) on cicada emergence interacted with the apple measure of Model (2) of Table 2, in student-level standard deviations per thousand bushels per km^2. Cohorts are matched to cicada exposure (test year minus ten) and stacked over all available cohorts within four years of each emergence in cicada-endemic counties, such that a county-cohort enters once for each window. Coefficients are relative to the cohort before the exposed cohort ($k = -1$), and the shaded band marks the exposed cohort ($k = 0$). County and year fixed effects and state-specific linear trends are included. Standard errors are clustered at the state level. Bars show 95% confidence intervals.

Figure A11: Inference: Leave-One-Out and Randomization

(a) Leave-one-out: by state (left) and county (right)



(b) Randomization inference



Notes: Panel (a): coefficient on Cicada × apple, in deaths per 1,000 births per thousand bushels per km^2, from the baseline infant mortality specification (county and year fixed effects and state-specific linear trends, with standard errors clustered at the state level) after dropping a single state (left) or county (right). The 10 largest apple-producing states and counties are dropped in turn, and the solid point marks the baseline estimate with its 95% CI. Panel (b): distribution of the state-clustered t statistic on Cicada × apple across 10,000 placebo calendars, each shifting every brood's emergence schedule by a uniform draw from $\{0, \ldots, L-1\}$ years (L the brood's 13- or 17-year cycle), preserving the spatial footprint and periodicity of treatment. The solid line marks the observed t statistic. Appendix Table A4 reports the two-sided p-values for the t statistic and for the coefficient.

Table A4: Inference for the Infant Mortality Estimate

	Standard error	p-value	Clusters
Cluster-robust and spatial standard errors			
State (baseline)	0.151	0.001	33
State, jackknife (CV3)	0.164	0.002	33
County	0.191	0.003	2,443
Conley, 100 km	0.263	0.033	
Conley, 200 km	0.285	0.049	
Design-based inference			
Randomization inference, t statistic		0.030	
Randomization inference, coefficient		0.087	

Notes: Each row is a different measure of uncertainty for the cicada × apple interaction in Model (2) of Table 2 (175,896 observations). The state jackknife (CV3) uses the 33 leave-one-state-out estimates, with the p-value from a t distribution with 32 degrees of freedom (MacKinnon et al. 2023). Conley standard errors (Conley 1999) allow for correlation between counties within the stated distance in the same year, but not across years. Randomization inference (Young 2019) compares the observed state-clustered t statistic, or the coefficient, with its distribution across 10,000 placebo calendars that shift each brood's emergence schedule by a random number of years. Figure A11 shows the placebo t distribution. All p-values are two-sided.

Table A5: Specification Sensitivity for Infant Mortality

Panel A. Outcome, estimator, and sample

	(1) Rate, OLS	(2) Rate, weighted	(3) Rate, births $\geq$ 25, OLS	(4) Deaths, Poisson	(5) log(deaths), OLS
Cicada × apple	0.559***	0.283*	0.525***	0.025**	0.047**
	(0.151)	(0.159)	(0.151)	(0.013)	(0.018)
Observations	175,896	175,896	174,230	175,896	152,877

Panel B. Fixed effects and trends

	(1) State trends	(2) No state trends	(3) State × decade FE	(4) State × year FE	(5) County trends	(6) Year × apple
Cicada × apple	0.559***	0.552***	0.625***	0.592***	0.569***	0.432**
	(0.151)	(0.140)	(0.150)	(0.140)	(0.148)	(0.210)
Observations	175,896	175,896	175,894	175,824	175,896	175,896

Notes: Each column re-estimates the main infant mortality specification, Model (2) of Table 2. Apple is county apple production in thousands of bushels per km^2, averaged across census years. All specifications include county and year fixed effects, and column (1) of each panel is the baseline with state-specific linear trends. Standard errors are clustered at the state level. Panel A varies the outcome, estimator, and sample. Columns (1) to (3) use the next-year infant mortality rate (deaths per 1,000 live births). Column (2) weights by number of births, and column (3) drops the 1,666 county-years with fewer than 25 births. Column (4) is a Poisson model of infant deaths with log(births) as an offset, which imposes a births elasticity of one. Column (5) is OLS on log(infant deaths) with log(births) as a control, restricted to the 87% of county-years with at least one infant death. Coefficients in columns (4) and (5) are semi-elasticities. Panel B keeps the baseline specification but varies the temporal trends and fixed effects: no state trends, state × decade fixed effects, state × year fixed effects, a separate linear trend for each county, and year × apple interactions along with state trends. $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$

Table A6: Alternative Treatment Measures and Land-Use Placebos

Panel A. Alternative apple-intensity measures and functional forms

	(1) Average bushels	(2) 1964 bushels	(3) 1997 bushels	(4) Time varying	(5) Apple acreage
Baseline (level)	0.559***	0.609***	0.434**	0.426***	0.223***
	(0.151)	(0.125)	(0.168)	(0.137)	(0.075)
Not normalized by area	0.392***	0.455***	0.288**	0.243**	0.164***
	(0.138)	(0.144)	(0.109)	(0.105)	(0.052)
Log (excl. zeros)	0.060	0.033	−0.005	−0.031	0.037
	(0.067)	(0.037)	(0.037)	(0.035)	(0.050)
Top decile	0.177	0.298	0.294*	0.144	0.171
	(0.174)	(0.183)	(0.158)	(0.178)	(0.167)

Panel B. Other land-use measures

	(1) Apple bushels	(2) Apples (CDL)	(3) Forest (CDL)	(4) Corn and soy	(5) Corn and soy, with apples
Cicada	0.029	0.039	0.151	0.068	0.033
	(0.122)	(0.123)	(0.255)	(0.167)	(0.169)
Cicada × land measure	0.559***	23.9*	−0.196	−0.027	−0.008
	(0.151)	(12.3)	(0.541)	(0.195)	(0.194)
Cicada × apple bushels					0.558***
					(0.158)
R^2	0.495	0.495	0.495	0.495	0.495
Observations	175,896	175,896	175,896	175,896	175,896

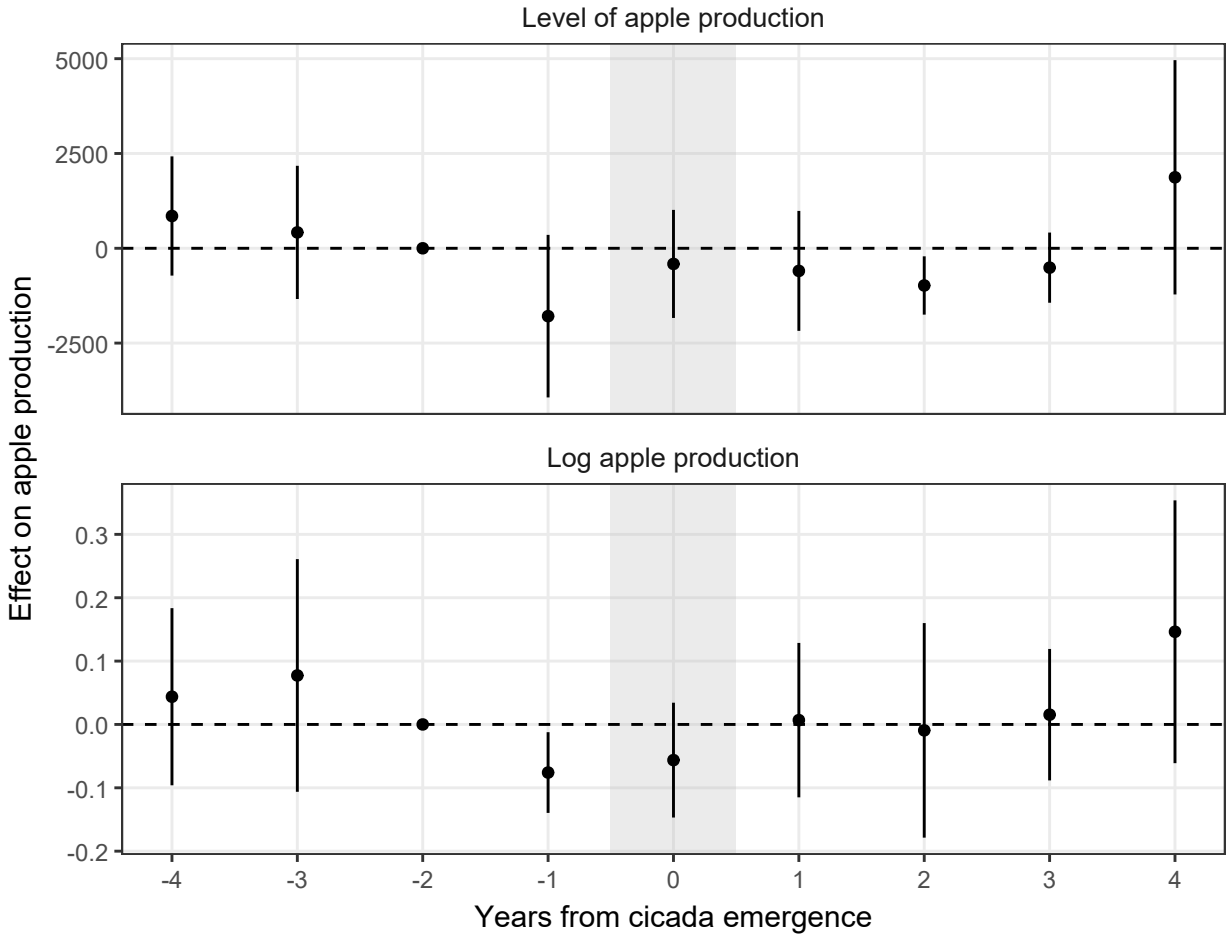
Notes: The dependent variable is the next-year infant mortality rate (deaths per 1,000 live births). Each cell or column re-estimates Model (2) of Table 2 with a different treatment measure, with county and year fixed effects, state-specific linear trends, and standard errors clustered at the state level. In Panel A, columns measure apple intensity as average, 1964, 1997, and time-varying bushels and planted acreage, each per km^2, and rows vary the functional form. The second row does not divide by county land area and is in millions of bushels per county in columns (1) to (4) and thousands of acres per county in column (5). The log row drops county-years with no recorded production, leaving 42,624 to 170,496 observations, and all other rows use the full sample of 175,896. In Panel B, column (1) repeats the baseline census bushel measure. Columns (2) and (3) use the share of total county area that the USDA Cropland Data Layer (CDL, 30m) classifies as apple orchard or forest in any year of 2008–2017. Columns (4) and (5) use average NASS county corn and soybean harvested acres from 1992 on, in hundreds of acres per km^2 (cross-county standard deviation 0.56), and column (5) adds the census bushel measure. Years 1950–2020. $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$

Table A7: Alternative Cicada Brood Map, Mid-Atlantic States

	(1) Original map, full sample	(2) Original map, Mid-Atlantic	(3) New map, Mid-Atlantic
Cicada	0.029	0.332	0.048
	(0.122)	(0.162)	(0.200)
Cicada × apple	0.559***	0.612***	0.682**
	(0.151)	(0.061)	(0.207)
R^2	0.495	0.357	0.357
Observations	175,896	19,152	19,152
Clusters	33	4	4

Notes: The dependent variable is the next-year infant mortality rate (deaths per 1,000 live births), and each column re-estimates Model (2) of Table 2. Column (1) is the full-sample baseline with the original century-old brood map. Columns (2) and (3) restrict the sample to the four Mid-Atlantic states covered by the new map (MD, PA, VA, WV) and use the original map and the 2021 map, respectively. County and year fixed effects and state-specific linear trends are included throughout, and standard errors are clustered at the state level. $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$

Figure A12: Cicada Emergence and Apple Production: Event Study



Notes: Event study of county apple production on cicada emergence event-time indicators, not interacted with apple intensity, in thousands of pounds (level and log), from annual USDA NASS data for 168 counties in VA, SC, KS, PA, and NJ, 1972–2012. The event study stacks every available year within four years of each emergence, and the estimation sample includes 103 counties. Coefficients are relative to two years before emergence ($k = -2$), and the shaded band marks $k = 0$. County and year fixed effects and state-specific linear trends are included. Standard errors are clustered at the state level (five clusters). Bars show 95% confidence intervals.

Table A8: Cicada Emergence and Farm Income

	Farm income ($ millions) (1)	Log farm income (2)
Cicada	−0.986	−0.057*
	(0.583)	(0.028)
Cicada × apple	0.252	0.031
	(0.738)	(0.037)
R^2	0.614	0.729
Observations	126,152	111,934
Year FE	Yes	Yes
County FE	Yes	Yes
State linear trend	Yes	Yes

Notes: Dependent variable is BEA county-level farm income, 1969–2020, in millions of dollars in column (1) and in logs (requiring positive farm income) in column (2). Cicada is 1 if a cicada emergence occurs in the county-year, interacted with average apple production in thousands of bushels per km^2. County and year fixed effects and state-specific linear trends are included. Standard errors are clustered at the state level. $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$

Table A9: Maternal Composition and Fertility

Panel A. Balance of maternal characteristics

	Cicada year	Non-cicada year	t-value	p-value
Mother's education (years)	12.583	12.603	0.484	0.629
Share Black	0.067	0.068	0.301	0.763
Share Hispanic	0.055	0.052	−0.972	0.331
Mother's age (years)	26.316	26.345	0.395	0.693

Panel B. Infant mortality with maternal controls

	(1)	(2)	(3)	(4)	(5)
Cicada	0.029	−0.016	−0.012	0.072	0.072
	(0.122)	(0.131)	(0.132)	(0.148)	(0.147)
Cicada × apple	0.559***	0.443*	0.445*	0.223	0.234
	(0.151)	(0.237)	(0.234)	(0.153)	(0.149)
Mother's age	No	No	Yes	No	Yes
Shares Black and Hispanic	No	No	Yes	No	Yes
Mother's education	No	No	No	No	Yes
Sample years	1950–2021	1978–2016	1978–2016	1978–2008	1978–2008
Observations	175,896	88,694	88,694	66,114	66,114

Panel C. Next-year birth rates

	Birth rate (crude)		Birth rate (women 15–44)	
	(1)	(2)	(3)	(4)
Cicada	0.148***	0.155***	0.677***	0.707***
	(0.033)	(0.034)	(0.162)	(0.167)
Cicada × apple		−0.132		−0.641
		(0.081)		(0.421)
Mean of dependent variable	16.7	16.7	83.5	83.5
R^2	0.828	0.828	0.829	0.829
Observations	140,621	140,621	140,621	140,621

Notes: Maternal characteristics are annual county averages from NCHS natality microdata. Panel A compares maternal characteristics for births in the year following a cicada emergence versus other years, among counties in the top decile of apple intensity (averaged across census years) with endemic cicadas. Each row uses the years in which the characteristic is recorded (age and race 1968–2016, education 1969–2008, Hispanic origin 1978–2016). The t- and p-values are for the difference in means and treat county-years as independent. Panel B re-estimates Model (2) of Table 2 on next-year infant mortality (deaths per 1,000 live births), adding the maternal controls marked in the table for the same next-year births. Columns (2)–(5) drop county-years without Hispanic origin, which some states did not record until 1991, and columns (4)–(5) also require education, recorded through 2008. Sample years refer to the birth year of the outcome. Panel C regresses the next-year birth rate, per 1,000 people or per 1,000 women aged 15–44, on cicada emergence and its interaction with apple intensity. It covers births in 1950–2007, since the ICPSR county birth and population data end in 2007, and excludes 16 Virginia counties and independent cities without separate ICPSR populations in 1969–1981. County and year fixed effects and state-specific linear trends are included in Panels B and C, and standard errors are clustered at the state level. $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$

Table A10: Cicada Emergence and Infant Mortality, by Cause of Death, 1968–2020

	All	External	Non-external	Perinatal	Congenital	Respiratory	SIDS	Cardio-vascular
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Cicada	−0.054	−0.009	−0.045	−0.008	−0.034	−0.025	0.013	0.015
	(0.128)	(0.028)	(0.120)	(0.085)	(0.053)	(0.020)	(0.023)	(0.017)
Cicada × apple	0.317	−0.041	0.357	−0.082	0.077	0.087	−0.028	0.124**
	(0.343)	(0.042)	(0.311)	(0.362)	(0.122)	(0.076)	(0.039)	(0.047)
Share of deaths	100.0%	4.3%	95.7%	51.0%	18.7%	3.1%	2.5%	2.1%
R^2	0.254	0.038	0.250	0.193	0.047	0.162	0.036	0.027
Observations	129,479	129,479	129,479	129,479	129,479	129,479	129,479	129,479
Year FE	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
County FE	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
State linear trend	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes

Notes: Dependent variable is the next-year infant mortality rate (deaths per 1,000 live births) by cause of death, 1968–2020. Columns report all causes, external causes (accidents and violence, a placebo), non-external causes, and the leading specific causes: perinatal conditions, congenital anomalies, respiratory causes, SIDS, and cardiovascular causes. 'Share of deaths' is each category's share of total infant deaths. Cicada is 1 if a cicada emergence occurs in the county-year, interacted with average apple production in thousands of bushels per km^2. County and year fixed effects and state-specific linear trends are included. Standard errors are clustered at the state level. $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$