North-South Displacement Effects of Environmental Regulation: The Case of Battery Recycling

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This study examines the effect of a tightening of the U.S. air-quality standard for lead in 2009 on the relocation of battery recycling to Mexico and on infant health in Mexico. In the U.S., airborne lead dropped sharply near affected plants, most of which were battery-recycling plants. Exports of used batteries to Mexico rose markedly. In Mexico, production increased at battery-recycling plants, relative to comparable industries, and birth outcomes deteriorated within two miles of those plants, relative to areas slightly farther away. The case provides a salient example of a pollution-haven effect between a developed and a developing country.
JEL: F18, Q56, O15
Keywords: pollution-haven hypothesis, environmental regulation, infant health

One of the animating concerns of the trade-and-environment debate is the idea that tighter environmental regulation in richer countries may, through trade, lead to relocation of dirty production activities to poorer countries with weaker regulations. Although not always stated, the concern often includes worries about adverse health effects in the destination. This is one articulation of what is commonly referred to as the *pollution-haven hypothesis.*

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Despite the prominence of this idea in academic and policy discussions, the direct evidence that environmental regulation can displace polluting activities from developed countries (the "North") to developing countries with weaker regulations (the "South") remains thin. Several influential papers have documented displacement effects across regions within the U.S. (Henderson, 1996; Becker and Henderson, 2000; Greenstone, 2002), but there is less evidence for North-South displacement. A leading study by Hanna (2010) finds that the Clean Air Act Amendments in the U.S. increased outgoing foreign direct investment (FDI) but not disproportionately to developing countries. The reviews by Copeland and Taylor (2004), Levinson (2010), Karp (2011), Cherniwchan, Copeland and Taylor (2017), Dechezleprêtre and Sato (2017), Cole, Elliott and Zhang (2017), and Copeland, Shapiro and Taylor (2021) report some evidence that regulation leads to fewer exports and more imports of pollution-intensive goods and less inward and more outward FDI, but little direct evidence of displacement of pollution-intensive production from North to South.

It appears that the dominant view in the policy world is that such displacement effects, if they exist, are small and relatively innocuous. For instance, the World Bank's 2020 *World Development Report*, its flagship publication, asserts: "[E]mpirical evidence shows that strict environmental regulation of polluting industries *has not led to large relocations to countries with less-strict standards....* The association of falling trade costs and tighter environmental regulations could drive polluters to flee to developing countries. *But this has not happened*" (World Bank, 2020, p. 125, emphasis added).

Here we provide a counterexample to this anodyne view. Focusing on recycling of used lead-acid batteries (ULAB), we document a direct effect of tightened air-quality regulation in the U.S. on relocation of polluting activities to Mexico and on birth outcomes in Mexico. Battery recycling has a number of features that make it both salient and amenable to empirical study. First, the industry is an intensive emitter of lead, a particularly noxious pollutant. A recent UNICEF report lists battery recycling first among concerning sources of lead exposure for children (Rees and Fuller, 2020).[1] Lead exposure has been linked to retarded fetal growth, lower IQ, lower educational achievement, and several other adverse outcomes. Second, there was a sharp experiment: in early 2009, the U.S. tightened the National Ambient Air Quality Standard (NAAQS) for lead by a factor of 10, from 1.5 $\mu g/m^3$ to 0.15 $\mu g/m^3$; the standard in Mexico remained stable at 1.5 $\mu g/m^3$ over the period. Third, the data environment allows us to track the relocation of battery recycling. We observe the locations of battery-recycling plants in the U.S., ambient lead levels at monitoring stations nearby, ULAB trade flows from the U.S. to Mexico, industry output and the locations of ULAB recycling plants in Mexico, and birthweight of infants born to mothers who live

[1] A 2017 World Health Organization (WHO) report writes, "Lead recycling is an important source of environmental contamination and human exposure in many countries.... [T]he health impacts of lead exposure are significant... Young children, pregnant women and women of childbearing age are particularly vulnerable to the toxic effects of lead" (WHO, 2017, pp. 2–3, 14).

near them, a particularly well-measured and fast-responding health outcome.

We have five main findings. First, the revised air-quality standard reduced ambient lead concentrations around U.S. battery-recycling plants. Lead concentrations declined sharply in areas where the new regulation was binding relative to areas where it was not; we estimate that the new standard reduced concentrations by 0.242 $\mu g/m^3$ from a pre-reform mean in binding areas of 0.549 $\mu g/m^3$. Second, ULAB exports from the U.S. to Mexico rose markedly after the reform; after remaining roughly constant between 2005 and 2008, ULAB exports rose by a factor of four between Jan. 2009 and the end of 2014. Third, the growth of value-added and output in Mexican battery-recycling plants was sharply higher in 2008–2013 than in 2003–2008, relative to similar industries. Value-added in battery recycling grew by 62.2% over the 5-year period from 2003–2008 (i.e. approximately 12.4% per year) and by 243.2% from 2008–2013; the comparable numbers for non-battery plants in the same broad sector (averaging across 6-digit subsectors) are 77.5% and -2.2%. Fourth, the average incidence of low birthweight increased significantly near Mexican battery-recycling plants (within 2 miles) relative to areas slightly farther away (between 2 and 4 miles). Averaging over all hospital types, we estimate that the policy change increased the incidence of low birthweight by 0.020 on a pre-reform mean of 0.095. Fifth, the health effects were concentrated among mothers in hospitals run by the Mexican Ministry of Health, who tend to be of lower socio-economic status than mothers in other hospital types. For this disadvantaged group, we estimate that the incidence of low birthweight rose by 0.048 on a pre-reform mean of 0.128 in our preferred specification; we find no statistically significant effect for mothers in private or other public hospitals. Together, these findings suggest strongly that the tightening of the U.S. lead regulation induced the relocation of battery recycling and caused negative health spillovers in Mexico. They also reinforce the argument of a large environmental-justice literature that the poor are disproportionately affected by environmental hazards (Currie, 2011; Hsiang, Oliva and Walker, 2019; Banzhaf, Ma and Timmins, 2019).

We provide a short review of the literature on the displacement effects of environmental regulation in Appendix A. Our reading is that few studies using quasi-experimental designs have focused on relocation of dirty production activities from North to South, and that those few have found little evidence of such displacement. In addition, we are not aware of a study that has traced the effects of rich-country environmental regulation through to health outcomes in a destination country.

I. Background

Lead-acid batteries are a major use of lead, and much of the lead in new batteries is from recycling of used batteries. In the U.S. in 2009, for instance, nearly 90% of lead consumption was for new lead-acid batteries, and approximately 90% of refined lead production was from ULAB recycling (Guberman, 2012). More-over, 99% or more of ULABs are typically recycled, making the lead-acid battery industry nearly a "closed loop," in which almost all lead used in production is reused in the same sector (Davidson, Binks and Gediga, 2016).

The U.S. has made substantial reductions in lead in ambient air, reducing average airborne lead concentrations by more than 90% from 1980 to 2016 (U.S. EPA, 2014), mostly by phasing out lead in gasoline (starting in 1973) and banning lead in paint (in 1978). In Mexico, lead was largely unregulated until the early 1990s, when limits were imposed on the lead content of paints, toys, pens, cosmetics and several other products (Romieu et al., 1994).

Despite the regulatory measures, lead continues to endanger public health. It is known to affect almost every organ and system in the human body. Children under six years old and fetuses are considered most susceptible, and exposure has been linked to learning disabilities, lower IQ, lower educational achievement, and later criminal activities (Needleman et al., 1990; Reyes, 2007; Aizer et al., 2018; Billings and Schnepel, 2018; Grönqvist, Nilsson and Robling, 2020). A number of studies have documented a relationship between maternal lead levels and birth outcomes, although the precise physiological mechanisms remain unclear (González-Cossío et al., 1997; Torres-Sanchez et al., 1999; Hernández-Ávila et al., 2002; Ettinger and Wengrovitz, 2010; Zhu et al., 2010; World Health Organization, 2017; Grossman and Slusky, 2019). Over time, the level of lead exposure considered to be safe has declined dramatically. The Centers for Disease Control and Prevention (CDC) progressively lowered its "level of concern" for blood-lead levels in children from 60 $\mu g/dl$ in 1960 to 10 $\mu g/dl$ in 2002; in 2012, it stopped using the "level of concern" terminology and concluded that "no safe blood lead level in children has been identified" (CDC, 2005, 2012).

In response to the growing body of evidence, the U.S. Environmental Protection Agency (EPA) reduced the NAAQS for lead from 1.5 $\mu g/m^3$ to 0.15 $\mu g/m^3$ in early 2009. It issued an Advance Notice of Proposed Rulemaking (ANPR), opening a period of debate, on Dec. 5, 2007. The new standard was signed on May 1, 2008 and took effect on Jan. 12, 2009. The standard is applied to three-month moving averages and enforced at the level of geographical areas, typically counties. An area found in violation is assigned "non-attainment status," which opens the door to substantially more stringent regulation.[2]

In Mexico, the air-quality standard remained at 1.5 $\mu g/m^3$ throughout our study period, and other dimensions of the regulatory regime were also largely unchanged. Awareness of the dangers of lead exposure from battery recycling grew over the period due to several reports and press accounts (OKI&FC, 2011; Commission for Environmental Cooperation (CEC) 2013; Rosenthal, 2011), and

[2]In 2012, the EPA also tightened a pollution standard at the point of production, the National Emission Standards for Hazardous Air Pollutants (NESHAP) for secondary lead smelting, which had been unchanged since June 1997. The new standard was implemented on Jan. 5, 2012, and existing plants were given until Jan. 6, 2014 to conform to the new rule (U.S. EPA, 2012). Given the timing of effects we document, we believe that the NAAQS change was the primary driver of production relocation, although the NESHAP change may have contributed in later years.

new point-of-production regulation was proposed in April 2014 and took effect in Jan. 2015 (Diario Oficial, 2014; 2015).

There exist technologies for reducing lead emissions from battery-recycling plants, including systems to filter exhaust through fabric ("baghouse systems"), to remove particles from exhaust through electrostatic precipitation, and to reduce fugitive dust emissions by enclosing production areas (CEC, 2016). But these systems are costly. For instance, Burr, Lazzari and Greene (2011) estimate that the annual costs for reducing lead concentrations to the new NAAQS standard for 14 plants active in 2009 together was $9.6 million per year.

One other important institutional detail is that the U.S. has not ratified the 1992 Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and Their Disposal, and ratifying countries (188 total) are in principle prohibited from trading with non-ratifying countries in the absence of a bilateral agreement. The U.S. has a bilateral agreement for export of hazardous waste (which includes ULABs) only with Canada and Mexico.[3] Thus in our context, the "South" effectively means Mexico only.

II. Data

Here we briefly describe our data sources; additional details are in Appendix B. We focus on the period 2002–2015.

We identify the geocoded locations of lead-emitting plants in the U.S. using the Toxic Release Inventory (TRI) (U.S. EPA, 2013). Among lead-emitting plants, we identify battery recyclers from a report by the Commission for Environmental Cooperation (CEC, 2013). The report lists 15 battery-recycling plants in the U.S., operated by 7 firms, in operation in 2007.

The EPA measures compliance with air-quality standards at approximately 4,000 monitors across the country, approximately 580 of which monitor lead (U.S. EPA, 2015). Appendix Figure A.1 plots their locations. The monitors are not located randomly: often they are placed where pollutant concentrations are expected to be high and measure the pollutants expected to be prevalent there. The monitors that measure lead tend to be located close to lead-emitting plants. We define distance for each monitor as the distance to the nearest lead-emitting plant. We focus on monitors within two miles of a lead-emitting plant, for reasons discussed below. To reduce the possibility that our estimates reflect the endogenous placement of monitors, we focus on monitors that were in place prior to the 2009 reform.[4] Appendix Table A.1 reports summary statistics for the 142 monitors in our main sample and the 22 monitors near battery-recycling plants.

We constructed a list of 26 authorized battery-recycling plants in Mexico from CEC (2013), from the Mexican counterpart of the TRI, the *Registro de Emisiones*

[3]See CEC (2013) and the EPA summary at https://www.epa.gov/hwgenerators/international-agreements-transboundary-shipments-hazardous-waste. (Accessed July 26, 2021.)

[4]In the empirical analysis, we include monitor fixed effects to absorb time-invariant differences across monitor locations.

y Transferencia de Contaminantes (RETC) (SEMARNAT, 2013), and from a separate list of battery recyclers published on the website of the ministry overseeing the RETC (SEMARNAT, 2011). Unfortunately, airborne lead concentrations were not systematically measured outside Mexico City during our study period. The location of Mexican battery-recycling plants are plotted in Appendix Figure A.2.

The data on exports and imports are from the U.S. Census Bureau, Foreign Trade Division, available on a monthly basis (U.S. Census Bureau, 2015). Trade of used lead-acid batteries is tracked in U.S. tariff codes 8548100540, 8548100580, and 8548102500, and we aggregate these three codes.

The Mexican production data are from the 2004, 2009 and 2014 Economic Censuses, each of which contains information from the previous calendar year (INEGI, 2014). We provided lists of battery-recycling plants active in 2008 and 2013 to INEGI, the Mexican statistical agency, and INEGI staff linked them to the Census microdata. To identify plants active in 2003, INEGI used the longitudinal links compiled by Busso, Fentanes and Levy (2018). It was possible to identify 8 battery recycling plants in 2003, 11 in 2008, and 15 in 2013.[5] Given the small number of plants, we are limited to using the censuses, rather than Mexico's monthly and annual industrial surveys, which have much less extensive coverage.

Our data on Mexican birth outcomes are from two sources. The first is discharge records of hospitals operated by the Mexican Ministry of Health (MH), which primarily serve a disadvantaged population not covered by the Mexican social security system (Secretaría de Salud, 2015*a*). The data report birthweight, gestation period, mother's age, and fetal death and are available on a consistent basis for 2005–2015. The second source is birth certificates issued by the Mexican National Health System, available beginning in 2008 (Secretaría de Salud, 2015*b*). While they are available for only one pre-reform year, they cover the universe of births and report detailed demographic characteristics of mothers. The broader coverage allows us to compare MH hospitals to private hospitals, whose patients tend to be significantly richer, and to other public hospitals, which cover mainly formal-sector workers and their families. Both sources report locality (*localidad*) of mother's residence, corresponding roughly to neighborhood. Appendix Table A.3 presents summary statistics for the two sources. Mothers in MH hospitals are younger, less likely to be married, and have lower completed schooling than those in private or other public hospitals.

[5]The linking process may have missed some recycling plants in 2003 that stopped producing before 2008 Census, but this would lead us to overstate the change in production between 2003 and 2008 and hence understate the acceleration in production between 2008 and 2013.

III. Results

A. Compliance with the New Lead Standard in the U.S.

In this section, we examine the effect of the tightening of the air-quality standard on airborne lead concentrations in the U.S. Because of data constraints, two empirical strategies that might seem natural in this setting are not feasible. One would be to compare ambient lead levels at monitors near and slightly farther away from plants affected by the reform. The difficulty here is that the number of monitors that measure lead more than two miles away from battery-recycling plants (or other lead emitters) is very limited. Another strategy would be to compare lead levels at monitors near lead-emitting plants to those near non-lead-emitting plants. The difficulty in this case is that few monitors near non-lead-emitting plants measure lead.

Given the data constraints, our strategy is to compare pollution levels at monitors near lead-emitting plants more and less affected by the regulation. We compare monitors near lead-emitting plants where the reform was binding, with ambient lead levels above 0.15 $\mu g/m^3$ prior to the reform, to monitors near lead-emitting plants where the reform was not binding. In implementing this strategy, a key decision is how to define "near." Our preferred definition is within two miles of a lead-emitting plant. As motivation for this choice, Appendix Figure A.3 plots lead concentrations by distance from plants for which average concentrations at nearby monitors were above the new standard, which we refer to as "binding" plants; the concentrations fall off quickly within the first two miles and remain roughly similar between 3 and 10 miles.[6] Below we also report results using a one-mile range, which we refer to as "very near."

Figure 1 plots average lead concentrations over time at monitors within two miles of a lead-emitting plant, separately for areas where the new standard was binding and where it was not. The vertical lines indicate the dates of the ANPR, the signing of the new standard, and the implementation of the new standard. For binding areas, there was no obvious trend pre-reform, but there was a clear decline in lead concentrations following the reform. As a further illustration, Appendix Figure A.4 plots the average lead concentration at monitors within two miles of a given lead-emitting plant in 2015 versus 2007, for plants with data in both years. Eight of the ten plants with average lead levels above the new standard in 2007 were battery-recycling plants, and all of the plants with initial average concentration levels above the new standard had average concentration levels below 0.15 $\mu g/m^3$ by 2015.

[Figure 1 here]

[6]The WHO (2017, p. 10) reports that a California battery-recycling plant was found to have contaminated the surrounding area up to 1.7 miles away. Given that battery-recycling plants are particularly intensive lead emitters, we believe that it is reasonable to focus on a slightly larger range than the one used by a leading previous study, Currie et al. (2015), which considered a range of pollutants.

Table 1 presents simple difference-in-difference and triple-difference estimates of the effect of the reform. Columns (1)–(3) use the sample of monitors within two miles of any lead-emitting plant. The coefficient on the $Binding \times Post$ interaction in Column (1) captures the differential decline in lead concentrations at monitors where the new standard was binding relative to monitors in non-binding areas.[7] The pre-reform mean at the binding monitors was 0.549 and the coefficient of -0.242 represents a decline of 44%. Column (2) shows that using an indicator for battery-recycling plant in place of the *Binding* indicator yields similar results, as would be expected given that it was mainly on such plants that the reform was binding. Column (3) adds the “very near” indicator. The coefficient on $Very\ Near \times Post \times Binding$ is identified by the comparison of the post-reform decline between monitors 0–1 and 1–2 miles from a lead-emitting plant in a binding area. The effect of the reform is statistically significantly stronger at very near monitors. At the same time, the $Binding \times Post$ interaction, which captures the effect on monitors 1–2 miles away, remains significant, providing support for our definition of “near” as within two miles. Column (4) reports the basic difference-in-difference specification for the subset of 22 monitors within 2 miles of a battery-recycling plant. The estimate is very similar to the estimate for the larger sample in Column (1). Overall, there is clear evidence that the reform reduced lead concentrations at monitors where it was binding, which were primarily those near battery-recycling plants.

[Table 1 here]

It is difficult to make definitive statements about U.S. lead output from battery recycling, because confidentiality rules for the U.S. Census of Manufactures would prevent the disclosure of information for such a small number of plants. But it appears from other sources that lead production from recycling of ULABs fell over the same period. The U.S. Geological Survey reports that lead output from battery recycling fell by 13% from 2007 to 2014, from 1.1 million to 0.96 million metric tons (Guberman, 2009, 2016).[8] Of the 15 battery-recycling plants in operation in 2007 listed in CEC (2013), only 10 were still in operation at the end of 2014 (Guberman, 2016, 2017).[9]

B. U.S. Exports of Used Lead-Acid Batteries to Mexico

Using the trade data from the U.S. Census Bureau, Figure 2 plots monthly ULAB exports from the U.S. to Mexico and to the rest of the world, primarily

[7]Standard errors are clustered at the monitor level to address the possibility of serial correlation (Bertrand, Duflo and Mullainathan, 2004).

[8]This decline came after output had risen by 10%, from 1.0 million to 1.1 million metric tons, between 2002 and 2007.

[9]The overall decline in output was slowed by the opening of a new Johnson Controls plant (the first new battery-recycling plant in the U.S. in 20 years) in Florence, SC, announced in June 2009.

Canada. There was a small increase in 2004, corresponding to the construction of a battery-recycling plant in Mexico by Johnson Controls, a major auto-parts producer. But the most notable feature of the graph is the trend break in early 2009. ULAB exports rose by a factor of four between Jan. 2009 and the end of 2014. In Appendix C.1, we test formally for a structural break using a Quandt likelihood ratio test, and we find clear evidence of a break in May–Aug. 2009. In Appendix C.2, we conduct a difference-in-difference analysis, comparing ULAB exports to exports in other 10-digit trade categories that map into the 3-digit sector in which battery recycling is typically considered to be located, Primary Metal Manufacturing (Sector 331 in the North American Industry Classification System (NAICS)).[10] We again find clear evidence that the reform increased U.S. ULAB exports. The fact that ULAB exports increased suggests that not all of the reduction in U.S. lead concentrations can be attributed to adoption of cleaner technologies by U.S. plants, which as noted above can be costly, especially for older plants.

[Figure 2 here]

C. *Growth of Battery Recycling in Mexico*

We turn now to the growth of battery-recycling plants in Mexico. The fact that we observe only two waves of pre-reform data limits our ability to apply synthetic-control methods (Abadie, 2021), which would otherwise be natural in this context.[11] Instead, we simply compare the growth of value-added for 2003–2008 and 2008–2013 for battery recycling and other 6-digit industries from NAICS Sector 331. To form the battery-recycling "sector," we aggregate the plants identified as battery-recycling plants. Figure 3 presents a scatterplot. Battery recycling is a clear outlier. Its value-added growth over 2003–2008 (62.2% over the 5-year period, or approximately 12.4% per year) was modest, below the median of 6-digit industries in Sector 331, and its growth over 2008–2013 (243.2%) was markedly greater than the other industries.[12] Appendix Figure A.7 presents a similar scatterplot for gross output. Growth in gross output was also high for industry 331520 ("Nonferrous Metallic Parts Molded by Casting") but, again, battery recycling's growth clearly accelerated in 2008–2013 relative to almost all other industries in the broad sector.

[10]Plants that engage in battery recycling are typically classified in NAICS 331419, Primary Smelting and Refining of Non-Ferrous Metal (except Copper and Aluminum), although in some cases they are classified in NAICS 335910, Battery Manufacturing.

[11]Given the small number of periods, this method would match battery recycling with industries based just on the 2003–2008 change in the outcome variable (e.g. value-added); it is not clear that industries that match on this single change are compelling comparators for battery recycling.

[12]The value-added growth rates for non-battery plants reported in the Introduction were calculated by taking an unweighted average of 6-digit industries in Sector 331.

[Figure 3 here]

D. Infant Health in Mexico

To estimate the health effects in Mexico, we compare birth outcomes for mothers living in localities near battery-recycling plants to those for mothers living in localities slightly farther away. Consistent with our approach in the U.S., we define "near" as within two miles of a battery-recycling plant and "slightly farther away" as between two and four miles away. Appendix Figure A.8 illustrates the assignment of localities to distance bins. We focus on births to mothers residing in localities in one of these two bins.

Our preferred model is the following:

$$
\begin{aligned}
(1) \qquad Health_{ijmht} \quad = \quad & \alpha + \beta Near_j \times Post_t + \rho X_{it} + \phi Z_{j,2005} \times Post_t \\
& + \mu_{mt} + \gamma_j + \lambda_{ht} + \varepsilon_{ijmht}
\end{aligned}
$$

where i, j, m, h, and t denote individual, locality, municipality, hospital, and year, respectively. $Health$ denotes a birth outcome, e.g. an indicator for low birthweight (< 2.5 kg) or birthweight itself. $Near$ is an indicator for mother's locality being 0–2 miles from the nearest battery-recycling plant. The $Post_t$ indicator takes the value 1 in 2009 and thereafter and 0 otherwise. The X_{it} vector contains mothers' characteristics. The $Z_{j,2005}$ vector contains initial values of locality characteristics, listed in Appendix B.6. The variables μ_{mt}, γ_j, and λ_{ht} are fixed effects for municipality-year, locality, and hospital-year. The coefficient of interest is β, which captures the differential effect of the U.S. reform on birth outcomes for mothers living in localities 0–2 miles from a battery-recycling plant relative to those living 2–4 miles away. The fact that we can control for hospital-year effects is a notable advantage over previous studies (e.g. Currie et al. (2015)), since there is extensive sorting of mothers across hospitals based on observable socio-economic characteristics and most likely on unobservable characteristics as well.

Table 2 presents estimates of equation (1). Panel A uses the Ministry of Health (MH) hospital-discharge records. These data contain limited information on mothers; X_{it} here includes only mother's age and age squared. Across columns, we include progressively richer sets of controls, using the same sample. Column (1) includes just municipality-year and locality effects, Column (2) adds the $Z_{j,2005} * Post_t$ locality controls, Column (3) adds hospital effects, and Column (4) adds hospital-year effects. The dependent variables are an indicator for low birthweight (< 2.5 kg) — the primary outcome considered in the literature (e.g. Currie et al. (2015)) and our preferred outcome — in Panel A.1 and birthweight itself in Panel A.2.[13] The results are reasonably stable across columns and con-

[13]Appendix Tables A.5–A.6 report all coefficients for the Panel A regressions, and include an additional

sistent across outcomes. Our preferred specification is the most stringent one, Column (4), with hospital-year effects. These estimates indicate that the share of low-birthweight births mothers rose by 0.048 (i.e. 4.8 percentage points) and birthweight declined by 38.5 g on average for mothers living in a locality within 2 miles of a battery-recycling plant relative to mothers living in a locality 2–4 miles away who gave birth in the same hospital in the same year.

[Table 2 here]

Panel B uses the birth certificates, which are only available for one pre-reform year but are available for all hospitals. We estimate the Panel A Column (4) specification for the different types of hospitals. To facilitate comparison we include the same covariates as in Panel A, i.e. mother's age and age squared.[14] In Column (1), for MH hospitals, the low birthweight indicator estimate is very similar to the Panel A.1 Column (4) estimate and again highly significant. The birthweight estimate is larger than the Panel A.2 Column (4) estimate — Appendix B.5 discusses differences in the data sources that give rise to this difference in magnitudes — but is again negative and highly significant. Pooling across hospital types, we see an increase of 0.02 in the incidence of low birthweight on average. But this effect is driven entirely by the MH hospitals. Strikingly, there is little evidence of a negative impact on birth outcomes in other public or private hospitals. For the low birthweight indicator, the estimates for other public and private hospitals are both very close to zero. For birthweight, the point estimate for other public hospitals is in fact positive, although not statistically significant, and the point estimate for private hospitals, although negative, is an order of magnitude smaller than the estimate for MH hospitals, and again not statistically significant.

To illustrate the timing of the impacts, Appendix Figure A.9 plots coefficients from a specification similar to our preferred one, Table 2 Panel A.1 Column (4), but interacting $Near$ with dummies for each year (with 2008 as the omitted reference year). There was no obvious trend prior to 2009; to the extent that there is a pattern, it suggests that the incidence of low birthweight was declining in areas closer to battery-recycling plants. But there is an evident and statistically significant increase in 2009. We cannot reject that the effect is constant thereafter. As mentioned above, awareness of the health consequences of battery recycling increased in Mexico over the period, culminating in new regulation imposed in early 2015; public pressure may be in part responsible for the slight decline in the effect over 2010–2015.

We have also considered the effects on other outcomes, in particular the incidence of very low birthweight (<1.5 kg), the length of gestation period, the

specification with municipality controls in place of municipality-year effects.

[14] Appendix Tables A.11 and A.12 report all coefficients for the Panel B regressions. Regressions using a richer set of mothers' characteristics are reported in Appendix Tables A.13 and A.14. The results are very similar.

incidence of premature birth (<37 weeks), and the probability of live birth. Appendix Tables A.7–A.10 report the results for the MH hospital-discharge data.[15] For very low birthweight (a rare occurrence), the effect is marginally significant in our preferred specification, but not robust across specifications. We do not find robust effects on gestation length, the incidence of premature birth, or the probability of live birth. The latter result suggests that selection into birth is not a major source of bias in our main estimates.

Our estimates for birth outcomes in Ministry of Health hospitals — an 0.048 (4.8 percentage point) increase in the incidence of low birthweight and a 38.5 gram (1.3%) decline in birthweight in our preferred specification — are large relative to many existing estimates of the effect of pollution on infant health, but not out of line with evidence on concentrated exposure among disadvantaged populations. Two leading related studies find comparatively small effects. Currie and Schmieder (2009) relate U.S. firms' self-reported releases of lead in the TRI to infant health outcomes at the county level, and find that a one-standard-deviation increase is associated with just a 0.00002 increase in the incidence of low birthweight and a decline in birthweight of 0.9 g. Currie et al. (2015) consider the effects of openings of industrial plants (which emit lead and other pollutants) on births to mothers living within 1 mile and find an increase in low-birthweight incidence of 0.002 and a decline in birthweight of 3.9 g. But other recent studies have found larger effects. Currie and Walker (2011) find that the introduction of E-ZPass in New Jersey and Pennsylvania was associated with a reduction of low-birthweight incidence of approximately 0.01 for mothers living within 2 miles of toll plazas, with a larger reduction for African-American mothers (0.024). Both estimates are about half the size of ours (0.02 on average for all hospitals, and 0.048 for disadvantaged mothers in MH hospitals). A study of the switch to a contaminated water source (containing lead and other pollutants) in Flint, Michigan, estimates a 175 g (5.4%) decrease in birthweight after adjusting for selection into birth (Grossman and Slusky, 2019). Currie, Neidell and Schmieder (2009) find that reductions of carbon monoxide in some areas of New Jersey were associated with birthweight increases of approximately 60 g, roughly what would be expected for a mother going from 10 cigarettes a day to zero. The fact that our estimates are on the high side of the range of existing estimates may be explained by the facts that battery-recycling plants are particularly intensive emitters, that lead is a particularly toxic pollutant, and that mothers in MH hospitals are a particularly vulnerable population, with limited access to high-quality health care.

IV. Conclusion

This short paper has provided evidence that the 2009 tightening of the U.S. airborne lead standard led battery recycling to shift from the U.S. to Mexico

[15] The probability of live birth cannot be examined in the birth-certificate data, since they only record live births.

and negatively affected infant health near Mexican battery-recycling plants. The data have limitations: for instance, we are not able to track year-to-year changes in output of Mexican battery-recycling plants, nor are air-monitor data on lead available outside of Mexico City over the study period. But the findings provide reasonably strong evidence of a pollution-haven effect in this industry, with adverse health consequences in the destination. The fact that the health impacts are concentrated among disadvantaged mothers echoes the findings of the environmental-justice literature that the costs of environmental hazards are disproportionately borne by the poor (Currie, 2011; Hsiang, Oliva and Walker, 2019; Banzhaf, Ma and Timmins, 2019).

Two important questions remain unanswered and merit further investigation. First, what were the health effects in the U.S. of the policy change? As noted above, two leading studies, Currie and Schmieder (2009) and Currie et al. (2015), suggest that the positive impact on U.S. birth outcomes was likely small. But neither study focuses specifically on the link between airborne lead concentrations and infant health. In addition, people living near "binding" plants tend to be disadvantaged relative even to people near other lead-emitting plants.[16] Given the income gradient in effects of pollution, one might reasonably expect larger impacts on the vulnerable population living near U.S. battery recyclers.

Second, to what extent is the case of battery recycling representative of broader patterns? We have focused on the case because there was a sharp regulatory change and a conducive data environment, but the technological characteristics of the sector may be special. Ederington, Levinson and Minier (2005) argue that evidence for pollution-haven effects has been mixed in part because industries with high pollution-abatement costs tend to be less mobile than those with lower costs. Battery recycling may be an unusual case in which abatement costs are high and both inputs (used batteries) and output (lead) are relatively transportable. More research on North-South displacement in other sectors is needed. But at a minimum, the case of battery recycling provides a clear example that environmental regulation in the North *can* displace polluting activities to the South.

From a policy perspective, a key contribution of this paper is to document an environmental production externality between the U.S. and Mexico — a particular sense in which the environmental fates of the two countries are linked. The externality points to a need for greater North-South coordination of environmental policy, in the same way that terms-of-trade externalities provide a motivation for trade agreements (Bagwell and Staiger, 2004). Such coordination is sure to be complicated both by differences in bargaining power between countries and by unequal distribution of impacts and influence within countries. There are also important questions about the extent to which trade and environmental negotiations should be linked (Copeland and Taylor, 2004; Limão, 2005). But it

[16] Appendix Table A.15 reports summary statistics from the American Community Survey indicating that the former group has lower household income and educational attainment and is much more likely to be Hispanic. See Appendix D for details.

seems clear that North-South displacement effects make environmental policy a legitimate subject for North-South policy negotiations.

REFERENCES

Abadie, Alberto. 2021. "Using Synthetic Controls: Feasibility, Data Requirements, and Methodological Aspects." *Journal of Economic Literature*, 59(2): 391–425.

Aizer, Anna, Janet Currie, Peter Simion, and Patrick Vivier. 2018. "Do Low Levels of Blood Lead Reduce Children's Future Test Scores?" *American Economic Journal: Applied Economics*, 10(1): 307–341.

Bagwell, Kyle, and Robert W Staiger. 2004. *The Economics of the World Trading System.* MIT press.

Banzhaf, H. Spencer, Lala Ma, and Christopher Timmins. 2019. "Environmental Justice: Establishing Causal Relationships." *Annual Review of Resource Economics*, 11(1): 377–398.

Becker, Randy, and Vernon Henderson. 2000. "Effects of Air Quality Regulations on Polluting Industries." *Journal of Political Economy*, 108(2): 379–421.

Bertrand, Marianne, Esther Duflo, and Sendhil Mullainathan. 2004. "How Much Should We Trust Difference-in-Differences Estimates?" *Quarterly Journal of Economics*, 119(1): 249–276.

Billings, Stephen B., and Kevin T. Schnepel. 2018. "Life after Lead: Effects of Early Interventions for Children Exposed to Lead." *American Economic Journal: Applied Economics*, 10(3): 315–344.

Burr, Mike, Donna Lazzari, and Danny Greene. 2011. "Draft Summary of the Technology Review for the Secondary Lead Smelting Source Category." Url: https://www.regulations.gov/document/EPA-HQ-OAR-2011-0344-0055 (accessed on June 23, 2021).

Busso, Matías, Oscar Fentanes, and Santiago Levy. 2018. "The Longitudinal Linkage of Mexico's Economic Census 1999-2014." IADB Technical Note IDB-TN-1477.

Centers for Disease Control and Prevention. 2012. *CDC Response to Advisory Committee on Childhood Lead Poisoning Prevention Recommendations in 'Low Level Lead Exposure Harms Children: A Renewed Call of Primary Prevention'.* U.S. Department of Health and Human Services.

Centers for Disease Control and Prevention, CDC. 2005. *Preventing Lead Poisoning in Young Children.* U.S. Department of Health and Human Services.

Cherniwchan, Jevan, Brian R. Copeland, and M. Scott Taylor. 2017. "Trade and the Environment: New Methods, Measurements, and Results." *Annual Review of Economics*, 9: 59–85.

Cole, Matthew A., Robert J. R. Elliott, and Liyun Zhang. 2017. "Foreign Direct Investment and the Environment." *Annual Review of Environment and Resources*, 42: 465/487.

Commission for Environmental Cooperation. 2013. *Hazardous Trade? An Examination of US-Generated Spent Lead-Acid Battery Exports and Secondary Lead Recycling in Canada, Mexico, and the United States,.*

Commission for Environmental Cooperation. 2016. *Environmentally Sound Management of Spent Lead-Acid Batteries in North America: Technical Guidelines.* Montreal, Canada.

Copeland, Brian R., and M. Scott Taylor. 2004. "Trade, Growth, and the Environment." *Journal of Economic Literature*, 42: 7–71.

Copeland, Brian R., Joseph S. Shapiro, and M. Scott Taylor. 2021. "Globalization and the Environment." National Bureau of Economic Research Working Paper 28797.

Currie, Janet. 2011. "Inequality at Birth: Some Causes and Consequences." *American Economic Review*, 101(3): 1–22.

Currie, Janet, and Johannes F. Schmieder. 2009. "Fetal Exposures to Toxic Releases and Infant Health." *American Economic Review*, 99(2): 177–83.

Currie, Janet, and Reed Walker. 2011. "Traffic Congestion and Infant Health: Evidence from E-ZPass." *American Economic Journal: Applied Economics*, 3(1): 65–90.

Currie, Janet, Lucas Davis, Michael Greenstone, and Reed Walker. 2015. "Environmental Health Risks and Housing Values: Evidence from 1,600 Toxic Plant Openings and Closings." *American Economic Review*, 105(2): 678–709.

Currie, Janet, Matthew Neidell, and Johannes F. Schmieder. 2009. "Air Pollution and Infant Health: Lessons from New Jersey." *Journal of Health Economics*, 28(3): 688–703.

Davidson, Alistair J, Steve P Binks, and Johannes Gediga. 2016. "Lead Industry Life Cycle Studies: Environmental Impact and Life Cycle Assessment of Lead Battery and Architectural Sheet Production." *International Journal of Life Cycle Assessment*, 21(11): 1624–1636.

Dechezleprêtre, Antoine, and Misato Sato. 2017. "The Impacts of Environmental Regulations on Competitiveness." *Review of Environmental Economics and Policy*, 11(2): 183–206.

Diario Oficial de la Federación Mexicana. 2014. "Proyecto de Norma Oficial Mexicana PROY-NOM-166-SEMARNAT-2014, Control de Emisiones Atmosféricas en la Fundición Secundaria de Plomo." Published April 22, 2014.

Diario Oficial de la Federación Mexicana. 2015. "Norma Oficial Mexicana PROY-NOM-166-SEMARNAT-2014, Control de Emisiones Atmosféricas en la Fundición Secundaria de Plomo." Published Jan. 9, 2015.

Ederington, Josh, Arik Levinson, and Jenny Minier. 2005. "Footloose and Pollution-Free." *Review of Economics and Statistics*, 87(1): 92–99.

Ettinger, Adrienne S., and Anne M. Wengrovitz, ed. 2010. *Guidelines for the Identification and Management of Lead Exposure in Pregnant and Lactating Women.* Centers for Disease Control.

González-Cossío, Teresa, Karen E Peterson, Luz-Helena Sanín, Eugenia Fishbein, Eduardo Palazuelos, Antonio Aro, Mauricio Hernández-Ávila, and Howard Hu. 1997. "Decrease in Birth Weight in Relation to Maternal Bone-Lead Burden." *Pediatrics*, 100(5): 856–862.

Greenstone, Michael. 2002. "The Impacts of Environmental Regulations on Industrial Activity: Evidence from the 1970 and 1977 Clean Air Act Amendments and the Census of Manufactures." *Journal of Political Economy*, 110(6): 1175–1219.

Grönqvist, Hans, J. Peter Nilsson, and Per-Olof Robling. 2020. "Understanding How Low Levels of Early Lead Exposure Affect Children's Life Trajectories." *Journal of Political Economy*, 128(9): 3376–3433.

Grossman, Daniel S., and David J. G. Slusky. 2019. "The Impact of the Flint Water Crisis on Fertility." *Demography*, 56(6): 2005–2031.

Guberman, David E. 2009. *2007 Minerals Yearbook: Lead.* U.S. Geological Survey.

Guberman, David E. 2012. *2010 Minerals Yearbook: Lead.* U.S. Geological Survey.

Guberman, David E. 2016. *2014 Minerals Yearbook: Lead.* U.S. Geological Survey.

Guberman, David E. 2017. *2015 Minerals Yearbook: Lead.* U.S. Geological Survey.

Hanna, Rema. 2010. "US Environmental Regulation and FDI: Evidence from a Panel of US-based Multinational Firms." *American Economic Journal: Applied Economics*, 2: 158–189.

Henderson, Vernon J. 1996. "Effects of Air Quality Regulation." *American Economic Review*, 86(4): 789–813.

Hernández-Ávila, Mauricio, Karen E. Peterson, Teresa González-Cossío, Luz H. Sanín, Antonio Aro, Lourdes Schnaas, and Howard Hu. 2002. "Effect of Maternal Bone Lead on Length and Head Circumference of Newborns and 1-Month-Old Infants." *Archives of Environmental Health: An International Journal*, 57(5): 482–488.

Hsiang, Solomon, Paulina Oliva, and Reed Walker. 2019. "The Distribution of Environmental Damages." *Review of Environmental Economics and Policy*, 13(1): 83–103.

Instituto Nacional de Estadística y Geografía (INEGI). 2014. "Censos Económicos 2004, 2009, 2014." https://www.inegi.org.mx/programas/ce/ (accessed on Dec 9, 2020).

Karp, Larry. 2011. "The Environment and Trade." *Annual Review of Resource Economics*, 3(1): 397–417.

Levinson, Arik. 2010. "Offshoring Pollution: Is the United States Increasingly Importing Pollution Goods?" *Review of Environmental Economics and Policy*, 4(1): 63–83.

Limão, Nuno. 2005. "Trade Policy, Cross-Border Externalities and Lobbies: Do Linked Agreements Enforce More Cooperative Outcomes?" *Journal of International Economics*, 67(1): 175–199.

Needleman, Herbert L., Alan Schell, David Bellinger, Alan Leviton, and Elizabeth N. Allred. 1990. "The Long-term Effects of Exposure to Low Doses of Lead in Childhood. An 11-year Follow-up Report." *New England Journal of Medicine*, 322(2): 83–88.

Occupational Knowledge International (OKI), and Fronteras Comunes (FC). 2011. *Exporting Hazards: U.S. Exports of Used Lead Batteries to Mexico Take Advantage of Lax Environmental and Worker Health Regulations.*

Rees, Nicholas, and Richard Fuller. 2020. "The Toxic Truth: Children's Exposure to Lead Pollution Undermines a Generation of Future Potential." UNICEF.

Reyes, Jessica W. 2007. "Environmental Policy as Social Policy? The Impact of Childhood Lead Exposure on Crime." *The B.E. Journal of Economic Analysis and Policy*, 7(1).

Romieu, Isabelle, Eduardo Palazuelos, Mauricio Hernandez Avila, Camilo Rios, lda Muñoz, Carlos Jimenez, and Gisela Cahero. 1994. "Sources of Lead Exposure in Mexico City." *Environmental Health Perspectives*, 102(4): 384–389.

Rosenthal, Elisabeth. 2011. "Lead From Old U.S. Batteries Sent to Mexico Raises Risks." *New York Times*. Dec. 9.

Secretaría de Medio Ambiente y Recursos Naturales (SEMARNAT). 2011. "Directorio de Empresas de Reciclaje de Residuos Peligrosos Industriales." http://tramites.semarnat.gob.mx/images/stories/menu/empresas/rubro1.pdf (accessed on Dec. 30, 2011).

Secretaría de Medio Ambiente y Recursos Naturales (SEMARNAT). 2013. "Registro de Emisiones y Transferencia de Contaminantes, 2004–2013." http://app1.semarnat.gob.mx/retc/index.html (Accessed on July 27, 2015).

Secretaría de Salud. 2015*a*. "Egresos Hospitalarios, 2005–2015." http://www.dgis.salud.gob.mx/contenidos/basesdedatos/std_egresoshospitalarios.html (accessed on July 15, 2018).

Secretaría de Salud. 2015*b*. "Sistemas de Información en Salud: Certificado de Nacimiento, 2008–2015." http://www.sinais.salud.gob.mx/basesdedatos/std_nacimientos.html (accessed on July 21, 2015) and http://www.dgis.salud.gob.mx/contenidos/sinais/subsistema1.html (accessed on July 15, 2018).

Torres-Sanchez, L. E., G. Berkowitz, L. Lopez-Carrillo, L. Torres-Arreola, C. Rios, and M. Lopez-Cervantes. 1999. "Intrauterine Lead Exposure and Preterm Birth." *Environmental Research Section*, 81(4): 297–301.

U.S. Census Bureau. 2015. "USA Trade Online, 2002–2015." https://usatrade.census.gov/ (accessed on Oct 21, 2018).

U.S. Environmental Protection Agency. 2012. "National Emissions Standards for Hazardous Air Pollutants from Secondary Lead Smelting." *Federal Register*, 77(3). Jan. 5.

U.S. Environmental Protection Agency. 2013. "Toxics Release Inventory (TRI), 2002–2013." https://www.epa.gov/toxics-release-inventory-tri-program (accessed on July 2, 2015).

U.S. Environmental Protection Agency. 2014. "Policy Assessment for the Review of the Lead National Ambient Air Quality Standards." U.S. Environmental Protection Agency document EPA-254/R-14-001.

U.S. Environmental Protection Agency. 2015. "Air Quality System, 2002–2015." https://www.epa.gov/aqs (accessed on June 22, 2015).

World Bank. 2020. *World Development Report 2020: Trading for Development in the Age of Global Value Chains.* Washington DC: World Bank.

World Health Organization. 2017. *Recycling Used Lead-Acid Batteries: Health Considerations.* Geneva: World Health Organization (WHO).

Zhu, Motao, Edward F Fitzgerald, Kitty H Gelberg, Shao Lin, and Charlotte M Druschel. 2010. "Maternal Low-Level Lead Exposure and Fetal Growth." *Environmental Health Perspectives*, 118(10): 1471–1475.

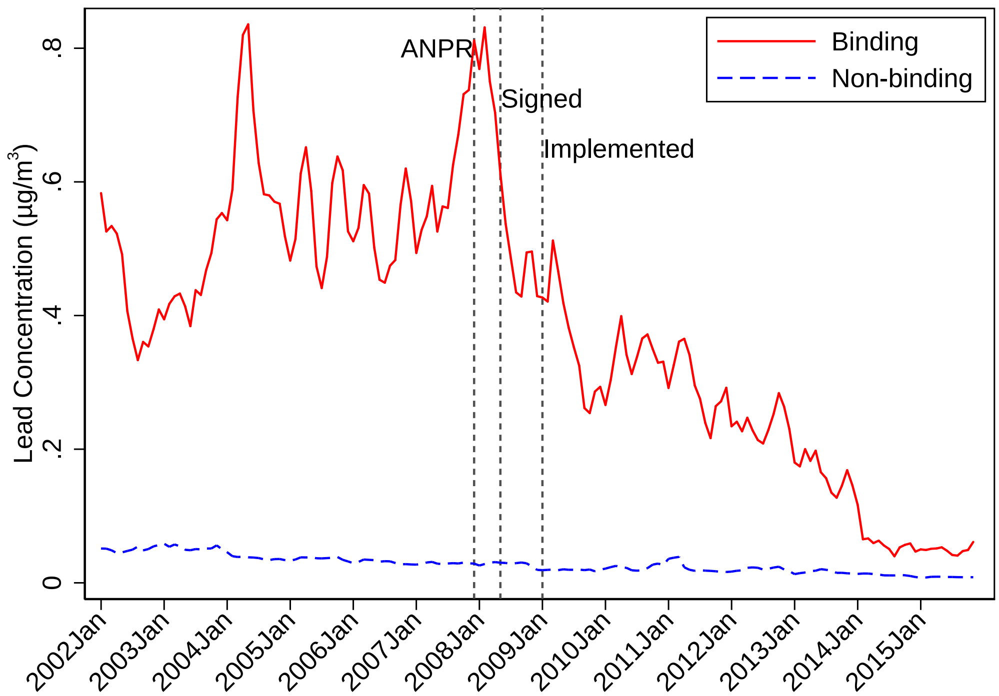

FIGURE 1. LEAD CONCENTRATIONS IN U.S., BINDING VS. NON-BINDING AREAS

Note: Sample is monitors within two miles of lead-emitting plants for which we observe both pre-reform and post-reform lead concentrations. Figure plots three-month moving averages in lead concentration levels in ambient air at monitoring stations between 2002 and 2015, separately for binding areas (concentration $> .15\ \mu g/m^3$ pre-2008) and non-binding areas. The leftmost vertical line indicates the date of the Advance Notice of Proposed Rulemaking (ANPR), Dec. 5, 2007; the middle line the signing of the revised standard (NAAQS), May 1, 2008; the rightmost the implementation of the new standard, Jan. 12, 2009.
Source: U.S. EPA (2015).

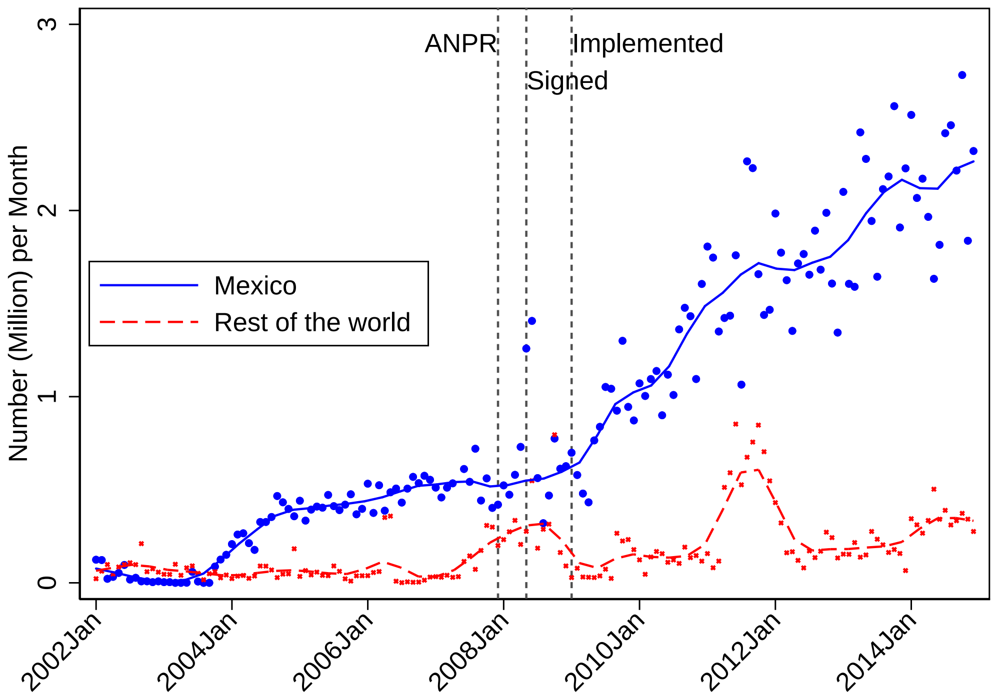

Figure 2. U.S. Monthly Exports of Used Lead-Acid Batteries

Note: Each dot indicates monthly exports (measured in millions of batteries) from the U.S. to Mexico (in blue) and the rest of the world (in red). The fitted trend lines indicate smoothed local polynomial trends with the bandwidth of three months. The leftmost vertical line indicates the date of the Advance Notice of Proposed Rulemaking (ANPR), Dec. 5, 2007; the middle line the signing of the revised standard (NAAQS), May 1, 2008; and the rightmost line the implementation of the new standard, Jan. 12, 2009. The trend for Mexico omits May and June, 2008, just after the new standard was signed, when exports spiked temporarily.
Source: U.S. Census Bureau (2015). Exports are the sum for U.S. tariff codes 8548100540, 8548100580, and 8548102500.

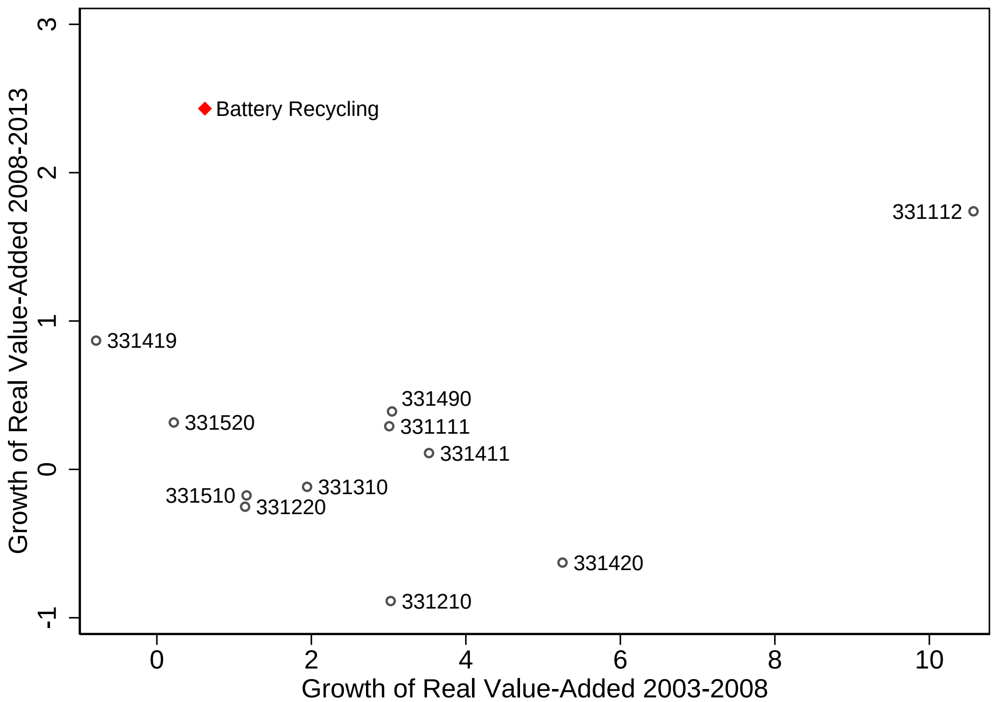

FIGURE 3. VALUE-ADDED IN BATTERY RECYCLING VS. SIMILAR INDUSTRIES IN MEXICO

Note: Value-added by industry is from 2004, 2009, and 2014 Mexican Economic Censuses (data for 2003, 2008, and 2013), for North American Industry Classification System (NAICS) sector 331 (Primary Metal Manufacturing) and for plants identified as battery recyclers. Real value-added is defined as gross output minus intermediate input consumption, deflated by the Mexican CPI. Growth of real value-added is defined as $(va_t - va_{t-1})/va_{t-1}$. The 6-digit industries are: 331111 – Iron and steel mills; 331112 – Primary roughs and ferroalloy manufacturing; 331210 – Iron and steel pipe and tube manufacturing; 331220 – Other iron and steel product manufacturing; 331310 – Aluminum production; 331411 – Copper smelting and refining; 331412 – Precious metals smelting and refining; 331419 – Other nonferrous metals smelting and refining; 331420 – Secondary lamination of copper; 331490 – Secondary lamination of other nonferrous metals; 331510 – Iron and steel parts molded by casting; 331520 – Nonferrous metallic parts molded by casting.
Source: INEGI, 2014.

Table 1—Effect of Tightened Lead Standard on Airborne Lead in the U.S.

	Dep. var.: Lead concentration ($\mu g/m^3$)			
	Monitors near lead-emitting plants			Monitors near battery-recycling plants
	(1)	(2)	(3)	(4)
Binding × Post	-0.242		-0.142	-0.252
	(0.047)		(0.003)	(0.067)
Battery × Post		-0.165		
		(0.050)		
Very Near × Post			-0.020	
			(0.007)	
Very Near × Post × Binding			-0.102	
			(0.050)	
N (observations)	16,858	16,858	16,858	3,133
N (monitors)	142	142	142	22
Pre-Reform Mean (binding monitors)	0.549	0.549	0.549	0.506
Monitor Effects	Y	Y	Y	Y
Year-Month Effects	Y	Y	Y	Y

Note: "Near" is defined as ≤ 2 miles. Sample in Columns (1)–(3) is monitors near any lead-emitting plant. Sample in Column (4) is monitors near a battery-recycling plant. In all columns, monitors are included only if they report lead emissions both before and after Jan. 1, 2009. Data are for 2002–2015. "Binding" means that lead concentration levels were above new standard at the most recent reading prior to Jan. 2009. "Post" takes the value 0 prior to Jan. 2009 and 1 thereafter. "Battery" means near a battery-recycling plant. "Very near" means ≤ 1 mile from a lead-emitting plant. Pre-reform mean is calculated for available years prior to 2009. Robust standard errors, clustered at the monitor level, are in parentheses.

Table 2—Effects on Birthweight in Mexico

Panel A. Hospital-Discharge Records	Ministry of Health (MH) Hospitals			
	(1)	(2)	(3)	(4)
1. Outcome: 1(Birthweight < 2.5 kg)				
Near × Post	0.022	0.043	0.049	0.048
	(0.0081)	(0.011)	(0.012)	(0.011)
Pre-Reform Mean (Near=1)	0.128	0.128	0.128	0.128
2. Outcome: Birthweight (grams)				
Near × Post	-35.0	-32.3	-40.4	-38.5
	(10.2)	(16.0)	(16.2)	(16.3)
Pre-Reform Mean (Near=1)	3,006.6	3,006.6	3,006.6	3,006.6
Observations	319,165	319,165	319,165	319,165
Locality Effects	Y	Y	Y	Y
Municipality-Year Effects	Y	Y	Y	Y
Locality Chars.×Post	N	Y	Y	Y
Hospital Effects	N	N	Y	N
Hospital-Year Effects	N	N	N	Y

Panel B. Birth-Certificate Data	Hospital Type			
	MH (1)	Other public (2)	Private (3)	All (4)
1. Outcome: 1(Birthweight < 2.5 kg)				
Near × Post	0.052	0.0020	0.0024	0.020
	(0.014)	(0.019)	(0.015)	(0.0081)
Pre-Reform Mean (Near=1)	0.124	0.100	0.071	0.095
2. Outcome: Birthweight (grams)				
Near × Post	-71.5	28.6	-8.19	-23.5
	(23.6)	(33.4)	(27.7)	(17.4)
Pre-Reform Mean (Near=1)	3,011.4	3,078.8	3,095.1	3,068.3
Observations	226,458	187,684	139,818	553,960
Locality Effects	Y	Y	Y	Y
Municipality-Year Effects	Y	Y	Y	Y
Locality Chars.×Post	Y	Y	Y	Y
Hospital-Year Effects	Y	Y	Y	Y

Note: Table reports 16 separate regressions. Post indicates year≥2009. All include a quadratic in mother's age; locality fixed effects (locality is area smaller than municipality); interactions of Post indicator with locality characteristics (share of households with access to water, electricity, and sewer, share of population below age 5, log total population, and share of population with social security); and interactions of Post with indicators for 1–5, 6–10, or ≥ 11 other lead-emitting plants ≤ 2 miles from mother's residence locality. Panel A sample is live births in Ministry of Health (MH) hospitals with mother's residential locality ≤ 4 mi. from battery-recycling plant, 2005–2015. Panel B sample is selected similarly but from birth certificates for 2008–2015, for MH, other public, and private hospitals. "Near" equals 1 if mother's locality is ≤2 mi. from battery-recycling plant, 0 otherwise. Pre-reform means are for near localities over available years (2005–2008 in Panel A, 2008 in Panel B). See Appendix B.6 for details on locality characteristics. Panel B uses specification from Panel A, Column (4), and varies sample. Robust standard errors, clustered at the locality level, are in parentheses.