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Working Paper

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Working Paper, No. 2003-08

Provided in Cooperation with:

Department of Economics, Rutgers University

Suggested Citation: Sigman, Hilary (2003) : Targeting Lead in Solid Waste, Working Paper, No. 2003-08, Rutgers University, Department of Economics, New Brunswick, NJ

This Version is available at:

<https://hdl.handle.net/10419/23166>

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Targeting Lead in Solid Waste

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September 2003

Prepared for OECD Workshop on the Economics of Waste, October 2003.

Abstract

This paper explores policy options for reducing lead in municipal solid waste. It focuses on policies that rely on economic incentives, such as taxes, deposit-refunds, and recycled content standards. The paper addresses the relative cost effectiveness of these approaches and also considers the overall desirability of government intervention to reduce lead disposal.

¹I am grateful to Huiying Zhang for research assistance.

Exposure to lead may present one of the most prevalent and serious environmental threats to human health. Recent research has shown that lead is toxic even at very low doses; indeed, there may be no blood lead level completely without adverse effects. With levels of lead exposure common in some U.S. cities, children and fetuses are at risk of IQ deficits, impaired reaction time, and other neurological problems (Banks et al., 1997). Studies have found that children exposed to lead have higher school drop-out rates, lower career achievement, and higher proclivity to crime, so the range of social harms and victims from lead pollution is great.

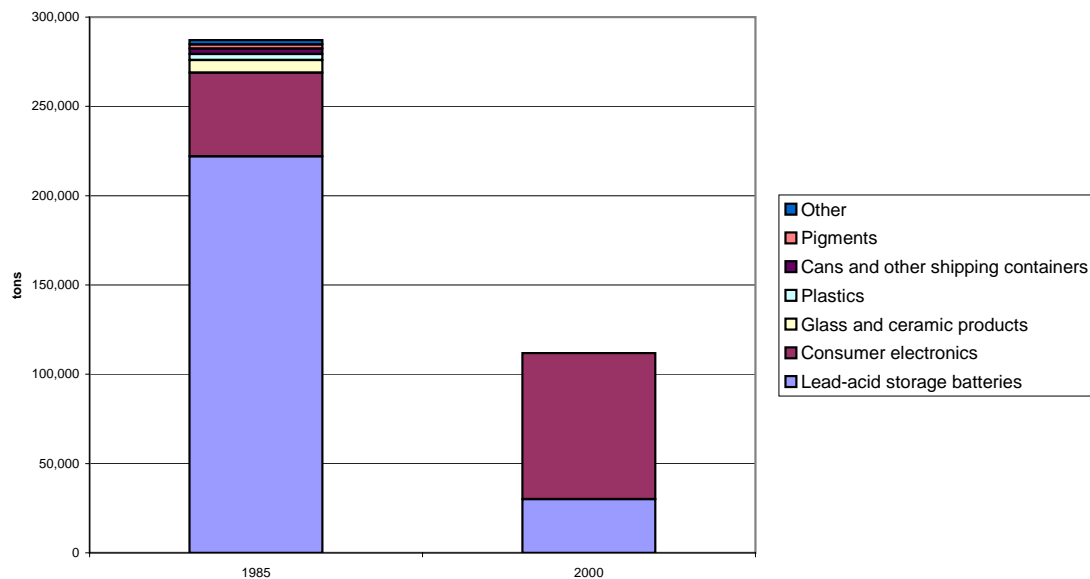
Many countries have made significant efforts to reduce exposure to lead, most importantly phasing out lead additives in gasoline. Other important policies target exposures from lead in paint, in food containers, and in drinking water from old lead-bearing pipes. There is also interest in reducing lead in solid waste because of the concern that lead waste may give rise to human and environmental exposure after wastes are incinerated or disposed in landfills. Different policy approaches have been suggested and adopted: this paper provides a comparison of some of the most prevalent and promising approaches.

The paper begins with background on the sources of lead in municipal solid waste and its implications. The second section discusses policy options to reduce lead in waste. In the third section, I quantify the effects and costs of several incentive policies for battery recycling in the U.S. and suggest how the results might differ if the policies were extended to other uses of lead. The empirical evidence suggests that incentive-based policies can be successful in encouraging recycling and reducing disposal, but there are substantial differences in the cost effectiveness of various incentive policies. The fourth section discusses earlier research that raises questions about the desirability of any policy to reduce lead in solid waste in the U.S. A final section briefly concludes.

1 Lead in municipal solid waste

Figure 1 presents the available information about lead in discards in the United States in 1985 and 2000 (Franklin Associates, 1989; U.S. Environmental Protection Agency (EPA), 2002).² The figure shows that two sources, lead-acid batteries and consumer electronics, dominated the sources in 1985, the most recent complete accounting of source of lead. In that year,

²There is considerable uncertainty about these figures. They are estimated based on sales figures from an earlier period, with the lag being determined by an average of the life-span of the relevant products. They assume that all of these defunct products enter municipal solid waste, when many households end up storing rather than disposing these products. A 1990 survey found that 20% of U.S. households had at least one old car battery stored in their house (EPA, 1991). Even more dramatically, a study in 1995 suggests that 75% of defunct cathode ray tubes are in storage (cited by Macauley et al. 2001). Although these stored products will probably ultimately be disposed, they suggest the difficulty in evaluating lead discard in any one year.

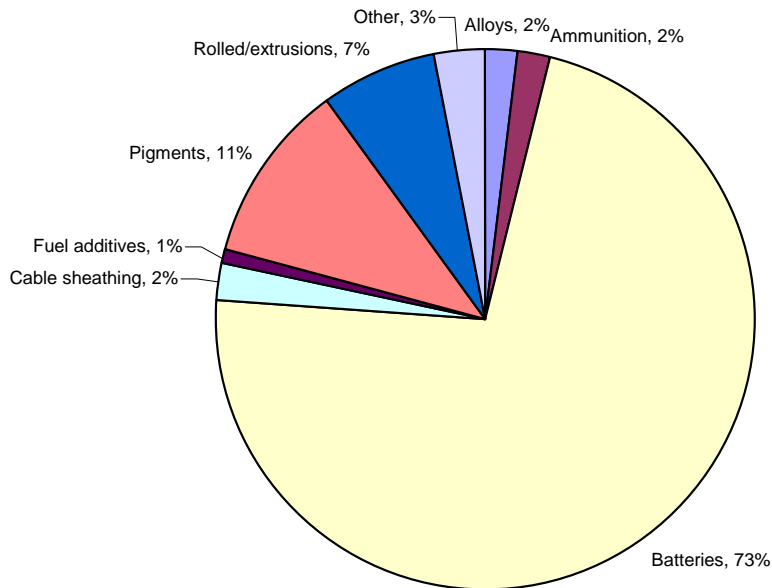


Sources: Franklin Associates (1989), all sources of lead, 1985; U.S. EPA (2002), batteries and consumer electronics only, 2000.

Figure 1: Lead in municipal solid waste in the United States, 1985 and 2000

batteries composed 77% of lead in municipal solid waste and consumer electronics, 16%. A few other sources, including principally glass, plastics, and cans represented cumulatively about 6% of lead discarded. For a more recent perspective, Figure 1 presents data for batteries and consumer electronics in 2000, but the 2000 values exclude smaller sources so understate the total.

Figure 1 suggests a dramatic decline in lead in municipal solid waste during this 15 year period. The source of this decline is an increase in the recovery of lead from batteries. As a result, batteries have shrunk from 77% of lead discarded to less than 30%. At the same time, there has been an increase in lead disposed in consumer electronics, in particular the lead in cathode ray tubes (CRTs) from televisions and computer monitors. Although these represented only 16% of lead in 1985, they now far surpass batteries as a source of lead. These figures are for the U.S. which recycles more batteries and uses both more storage batteries and electronics than most other countries; however, the relative change over time is probably similar in other developed countries. Thus, any policy that aims to reduce lead in municipal solid waste should focus on these two sources, which this section discusses in more detail.



Source: Thorton et al. (2001), from ILZSG data.

Figure 2: International uses of lead, 1997

1.1 Lead-acid batteries

Lead's primary use is for lead-acid batteries. Figure 2 shows use in 1997 of lead in countries that are members of the International Lead and Zinc Study Group (ILZSG) and account for about 80% of global lead consumption. The figure shows that batteries account for 73% of lead use. In the U.S., batteries are even more predominant, consuming 1.4 million metric tons in 2001, 87% of the total consumption of lead. Most lead-acid batteries are used as starting-lighting-ignition batteries for motor vehicles (78% in 1992). The remaining batteries are used for motive power in electric vehicles, such as in-plant fork-lifts, and other industrial uses, such as uninterruptable power supply for large computer systems and standby power supply for emergency lighting.

Demand for batteries will probably continue to grow for a few reasons. First, increased vehicle use will raise demand. Second, battery lives are declining because of higher vehicle underhood temperatures, greater use of vehicles in warmer climates where their life-span is shorter (averaging 40 months in the U.S. South relative to 58 months in the North), and more electrical gadgets in cars (*Purchasing*, 1997). Finally, storage batteries are seeing expanded use in electric and hybrid vehicles and as backup power for computer and electrical systems.

Although batteries represent the dominant use of lead, their contribution to lead in

municipal solid waste is not as great because most batteries are recycled. Secondary lead accounted for 79% of refined lead production in the U.S. and possibly as much as 43% of production internationally in 2001 (Smith, 2003). Used batteries constitute most recovered scrap, 92% in the U.S. in 1998 (Smith, 2002). The remaining recovered lead includes 5% “new scrap” — scrap recovered from facilities that use lead as an input — and about 3% old scrap lead from metal sources such as casting, sheets, solder and fabricated metal products.

In the U.S., retail battery dealers collect used batteries from consumers and typically discount purchases of new batteries in exchange. Discounts have varied considerably: in general, they have been in the range of \$4 to \$7 (BCI, 2003). In the early 1980s when refined lead prices were especially low, however, some dealers charged a fee of \$.50 for the disposal of batteries (Putnam, Hayes and Bartlett, 1987). Scrap battery dealers purchase used batteries and sell them either to battery “breakers,” who remove the plastic cases and drain battery acid, or directly to the secondary smelters break the batteries and then re-refine the lead. Secondary lead from battery scrap is often used to make batteries (some battery manufacturers operate secondary smelters) and is a very close substitute for primary lead in some other uses.

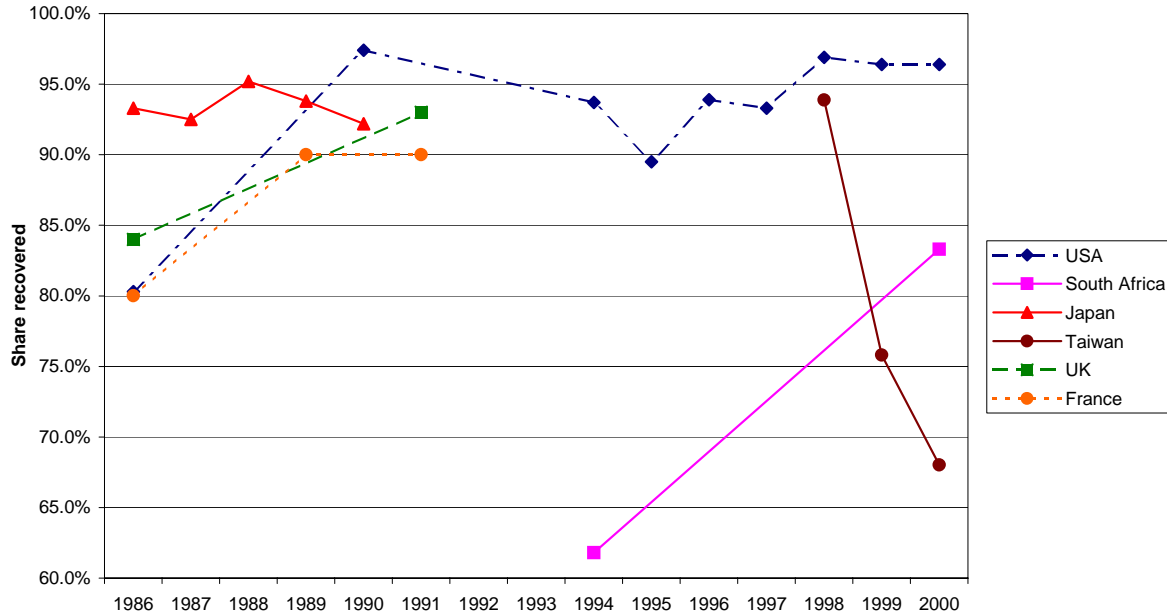
This recycling chain usually captures a large fraction of the used storage batteries. Figure 3 shows estimated recycling rates for batteries in a few countries for which recent estimates are available. These rates cannot be measured directly because the number of defunct batteries in a given year is unknown; it is estimated based on sales from a period of a few years prior, using a typical failure interval for different types of batteries. The rates in Figure 3 are share of batteries recycled, which includes not only the lead but also a substantial volume of plastic casings; however, the share of battery lead recovered is similar in magnitude and trend.³

Although the data are somewhat patchy, Figure 3 suggests a few conclusions.⁴ First, battery recycling rates are high in high income countries. By the early 1990s, rates in the UK, Japan, and the U.S. exceeded 90%. Second, rates have generally risen over time. Public policies, discussed below, may have partly driven the increase in recycling rates.

Another source of information about battery recycling for the U.S. is a survey of generators of small quantities of hazardous waste in 1983 (Abt Associates, 1985). In that survey, the generators of used batteries (primarily vehicle repair shops) sent 89% of batteries they collected to off-site recyclers, with the remainder mostly sent to solid waste landfills.

³Recovery rates for batteries exceed the ratio of secondary to total lead use for two reasons. First, other uses in addition to batteries are in the denominator in the later concept. Second, growth in lead use over time makes the quantity of scrap lag behind new demand.

⁴Some of the volatility in the data, especially for Taiwan, may be noise rather than rapid shifts in actual conditions.



Sources: Japan, UK, France, OECD (1993); USA, U.S. EPA (2002); South Africa, Joseph and Verwey (2001); Taiwan, Lee (2003).

Figure 3: Lead-acid battery recycling rates in selected countries, 1986–2000

This percentage is probably higher than total recovery because some households change and dispose their own batteries. Thus, the survey appears to be broadly consistent with the estimated recovery rates.

There is substantial trade in scrap batteries. The US exported 152,000 tons of scrap battery lead (either in drained or undrained batteries) and imported 27,000 tons per year on average from 1997 through 2001 (BCI, 2003b). The export figure represents about 14% of the battery lead recovered domestically.

1.2 Consumer electronics

U.S. EPA (2002) estimates that consumer video products contain 7% lead by weight and information products contain about 3% lead by weight. The vast majority of this lead is in CRTs used for televisions and computer monitors, although electronics also contain some lead in solder on circuit boards. Figure 1 suggests that the total lead in municipal waste from consumer electronics has grown dramatically since 1985. This change results from increased use of these products, especially computers, and also changes in their lead composition. In particular, monochrome computer monitors contain substantially less lead

than color monitors (Macauley et al., 2001).

In contrast to lead-acid batteries whose role and design are fairly stable, this technology is in flux, so historical data provide less of a guide for public policy. A few technological changes may be especially important in the immediate future. First, flat panel display monitors and televisions do not have the lead content of CRTs. Flat panel displays are still considerably more expensive than traditional technologies, but if their sales rise, current lead disposal in monitors may turn out to be a temporary spike. On the other hand, the advent of high definition television may cause widespread replacement of older televisions and, at least temporarily, shorten the life-span of these products.

The recycling chain for electronics is more complicated than for batteries because of a range of possible end-of-life fates. Used electronics may be immediately reused (as with some charitable donations), refurbished and resold, demanufactured for spare parts, or sent to facilities that recover raw materials, or experience some combination of these fates. The National Safety Council (NSC, 1999) provides the best data on recycling of electronics in the United States. The report finds a substantial discrepancy between the number of CRTs reported processed by recyclers and the number reported received by glass recovery facilities and secondary smelters.⁵ Despite this discrepancy, it is clear that recycling rates are low. The NSC report concludes that 1.5 million computer monitor CRTs were recovered in 1998, compared with 15.8 million that became obsolete, for a recovery rate of under 10%. The reason for this low recovery is the high cost of recycling: the report indicates that smelters charge \$.10 per pound to accept intact CRTs and \$.07 per pound for crushed CRTs (about \$2 to \$3 per unit), distinguishing them from used batteries which are a valuable commodity.

1.3 Consequences of lead in municipal solid waste

The environmental harms from lead in municipal solid waste depend on whether waste is sent to a landfill or a combustion facility, such as an incinerator or waste-to-energy facility. In a landfill, the concern is that battery cases break and CRT glass becomes pulverized, so that liquids moving through the landfill can become contaminated. Contaminated leachate may reach groundwater when leachate containment systems fail or in places where wastes are landfilled without such safeguards.

The extent of the danger from landfilling lead is not well documented. Past history does suggest some risk. Of the 158 municipal landfills on the U.S. priority list for contaminated sites, 22% have released lead (ATSDR, 1988). However, these landfills predate modern leachate containment and monitoring and the decline in lead discards. Macauley et al.

⁵The NSC report hypothesizes the discrepancy arises because firms fail to report all the used CRTs they export (or sell to brokers for export).

(2001) cite releases of lead to leachate from .0035 pound to under a billionth of a pound per CRT (with the range being determined by whether glass is pulverized or just broken in the landfill). Given leachate containment and inexpensive alternatives to use of any contaminated groundwater, they conclude that land disposal of CRTs imposes few health costs in the U.S.

Combustion may pose a bigger risk. Both batteries and consumer electronics can potentially be separated out from other wastes before the wastes enter a municipal waste combustion facility. However, many facilities do not have suitable sorting areas before materials are placed into furnaces and even those that may require visual inspection of the waste, presumably a leaky process. Thus, it seems likely that a substantial share of lead in municipal waste does enter combustion facilities.

When lead does enter the facilities, it elevates the toxicity of ash and, more harmfully, may be emitted to the air. According to estimates by Pacnya and Pacnya (2001), global air emissions of lead from waste disposal were 821 metric tons per year in the mid-1990s. This is a tiny share of total air emissions of lead because the vast emissions of lead additives to fossil fuels. However, excluding lead from fossil fuels, municipal waste disposal accounts for about 4% of lead emissions to air.

Nonetheless, Macauley et al. (2001) find fairly low costs from combustion of computer monitor CRTs in the United States. Using air emission rates of .00026 pounds per CRT incinerated and assuming that the share of consumer's monitor CRTs incinerated equals the national average share of all municipal solid wastes incinerated, they conclude that the health costs have a dollar value of \$2.67 million annually. This value may be an overestimate because it gives no role to sorting at the facility. On the other hand, it captures only costs from computer monitor CRTs and reflects U.S. air pollution controls, so harms in less stringently regulated countries may be greater.

In addition to health risks, households that dispose lead impose financial costs on other households by raising the average cost of waste management. Reducing the toxicity of the municipal waste inputs might relax constraints on the design and location of management facility. Thus, it might improve efficiency to discourage discards of these products, even if environmental standards ensure that lead discards do not harm anyone's health.

2 Analysis of public policies

In response to these costs and risks, several public policies have been proposed and implemented to reduce lead disposal.⁶ Two general approaches can be taken, one more direct than the other.

2.1 Direct approaches

Direct policies place restrictions on lead disposal. The most common direct response is a ban. In the U.S., 42 states have banned lead acid-battery disposal by households (BCI, 2003). One state, Massachusetts, banned disposal of CRTs in 2000. Similarly, large business users of batteries and CRTs may fall under hazardous waste regulations, which require that they dispose the products in special hazardous waste landfills.

Another direct approach has the government charge fees for disposal of lead-containing products that reflect its environmental costs. Unlike bans, this price-based policy would allow some battery disposal to continue if the cost of alternatives is higher than the fee. Such fees do not appear to have seen widespread use.

The difficulty with these direct approaches is high costs they may impose with imperfect enforcement. If users dump wastes surreptitiously because the policy precludes legal disposal and they wish to avoid the costs of recycling, the environmental implications of a ban may be far worse than the harm it seeks to remedy. There are no estimates of how much dumping occurs in response to current restrictions on auto batteries. However, I studied reported incidents of dumping of used oil — another waste largely handled by auto repair shops — in the United States from 1987 through 1994 (Sigman, 1998). States that banned disposal of used oil experienced a 28% increase in the number of dumping incidents, suggests a substantial adverse effect of disposal bans.

2.2 Indirect approaches

More indirect policy approaches may reduce disposal without these adverse consequences. These approaches tend to discourage use of the lead in products and promote recycling. Although there are a large number of possible policies, I will consider four incentive approaches in particular: (i) deposit-refunds; (ii) taxes on lead; (iii) subsidies for recycled lead and (iv) recycled content standards. For clarity, this section ignores nonrecyclable uses of lead; the implications of these uses are discussed later.

⁶For economic comparisons of alternative waste management policies, see Dinan (1993), Fullerton and Kinnaman (1995), Palmer and Walls (1997) and Porter (2002).

Deposit-refund. Deposit-refunds are second to disposal bans as the most common type of public policy in place to curb lead disposal. Nine states place deposit-refunds on vehicle batteries, at a rate of \$5 or \$10 per battery (BCI, 2003).⁷ Several countries, including Mexico, Denmark, and South Africa, also have deposit-refunds on car or lead-acid batteries (OECD/EEA, 2003; Johnson and Verwey, 2001).

The deposit-refund requires a fee for the initial use of the lead that is rebated when the lead is recycled (Bohm, 1981). It raises the cost of lead to consumers and also the value of used batteries. Thus, it provides incentives for both the reduction of lead use and for recycling.⁸

A deposit-refund may be imposed at different levels. The most common approach — placing the deposit and refund on the consumer — may help make the public aware of the government’s interest in battery recycling, but has significant drawbacks. A better deposit-refund would be imposed at the producer level, with a charge for lead use in production and subsidy for recovered lead.⁹ Assuming well-functioning markets, the deposit would be passed forward to consumers in the form of higher prices for goods containing lead. In addition, because secondary lead producers receive a subsidy, they would provide higher rewards for (or at least greater ease in) the return of scrap products.

This producer-based deposit-refund has a number of benefits. First, it is likely to lower the administrative costs of the deposit-refund because the government does not need to assure that all retail transactions have the deposit and refund properly administered. Second, it provides greater incentives to assure that lead scrap collected from consumers is recovered because the refund is not payable otherwise. A consumer-level deposit-refund may result only in the collection of batteries and their subsequent disposal by retail stores or collection centers — with some benefit in perhaps allowing them to be disposed as hazardous waste rather than in general municipal waste — but without the full effect of reducing lead disposal.

Taxes on lead. An alternative policy is to tax lead. One approach applies the tax on to lead products. By raising the cost of lead, such a tax may discourage its use, but would not have a direct effect on recycling. Although in principle such a tax could apply to all

⁷The states have mandatory deposit-refunds are: Arizona, \$5 per battery; Arkansas, \$10 ; Connecticut \$5; Idaho, \$5; Maine, \$10; Minnesota, \$5; New York, \$5; South Carolina, \$5; and Washington, \$5. Wisconsin and Rhode Island have voluntary charges. All these policies were implemented by 1989 and 1991.

⁸Not only does a deposit-refund not create incentives for illegal disposal, it may even encourage the collection of illegal dumped and stored lead products because consumer can collect refunds by returning products. Collection of the illegally dumped products creates substantial environmental benefits; collection of stored products, especially old computers, probably does not and may add to the cost of program by requiring handling of products that would not face environmental release.

⁹See Fullerton and Wolverson (1999) for their discussion of “two-part” instruments in environmental policy, of which this producer deposit-refund is an example.

refined lead or to all products containing lead, in practice most of these taxes are restricted to batteries. Four U.S. states and several European countries have charges on batteries, ranging in the U.S. between \$1 and \$3 per battery (BCI, 2003; OECD/EEA, 2003). These charges are typically fixed per battery and thus do not provide incentives for use of lower lead batteries, unlike a pure lead tax which would vary with the lead content of the product.¹⁰

Another tax approach — a virgin material tax — applies only to primary lead. The U.S. has considered such a tax (*Environment Reporter*, 1991). A virgin material tax raises the price of primary lead to its users. Because primary and secondary lead are perfect substitutes, users will purchase recycled lead at the same price, implying that the tax will also raise the price of recycled lead. As a result, the virgin material tax is similar to a deposit-refund in its effect. Under both programs, recycling consumers receive a rebate. The rebate is the direct refund with a deposit-refund; with a virgin materials tax, the tax increment is returned in the form of higher prices for lead scrap.

Subsidy to recycled lead. An alternative program would provide a direct subsidy to recycled lead production. This exact approach does not seem to have been used in practice. However, there are indirect subsidies to recycled lead. For example, many U.S. states provide investment tax credits (which offset income taxes) to firms that produce recycled goods (U.S. Office of Technology Assessment, 1989). Such indirect subsidies may have similar effects to a direct subsidy in encouraging lead recovery; however, they will also have additional effects (not analyzed here) on the recyclers' production decisions that raise the costs of these policies relative to direct subsidies.

The effect of a subsidy is to lower the costs of recycled lead relative to virgin lead. This should have the effect of reducing virgin lead production. However, the price of lead to its users declines because recycled lead has become cheaper. Thus, although the recycling subsidy may encourage recycling, it also creates an incentive for *increased* consumption of the lead.

Recycled content standard. A recycled content standard stipulates a ratio of recycled lead to total lead used. In the United States, Congress has considered a recycled content standard for batteries before recent increases in the recovery rate. Early versions of the EU's Directive on Waste Electrical and Electronic Equipment (WEEE) seem to have contained a recycled content standard that had been dropped by the time of the final version.

¹⁰Texas does distinguish high and low voltage batteries, which will reflect their lead content. Below, I discuss whether the difference between fixed charges per battery and charges based on the lead content of the battery will make much difference empirically.

The government may implement the recycled content standard on a variety of scales. One approach would require that each individual product has a given recycled lead content, for example 80%. This policy would give a firm no flexibility to choose product lines for which recycled lead was most appropriate and would require the manufacturer to engage in costly monitoring of the use of specific shipments of lead. A more flexible approach is to set the requirements at the firm level, so a firm would have to assure that its products overall contained 80% secondary lead, but some products or batches might have higher or lower recycled content.

The most flexible approach applies the recycled content standard not to a firm but to all users of lead. The government would set a requirement that 80% of all lead consumed be secondary lead and leave it to firms to assure that this level was obtained. Firms could meet this requirement by trading recycled content: a firm using a larger fraction of recycled material than the standard can trade its surplus with firms using too little, so the standard applies to the industry as a whole. If the permit market operates well, this tradable recycled content standard accomplishes the 80% level (or whatever level is chosen) at least cost.¹¹

It turns out this tradeable recycled content standard also has an equivalent tax-subsidy combination. Suppose that a permit entitles its holder to use one unit of virgin lead. This permit must be traded for enough units of recycled lead that the recycled content for the industry conforms to standard, call it r^* . This trade-off holds if a permit can be created by the use of an additional $\frac{r^*}{1-r^*}$ units of recycled lead. Thus, for example, with the 80% standard, a firm could use 1 ton of virgin lead if it found other firms willing to use 4 tons of recovered lead. This arrangement means there is a cost equal to the price of a permit (whatever that turns out to be) for using virgin lead, like a tax. There is an effective subsidy for using secondary lead, equal to the value of the permits that are created: in this 80% content standard case, the subsidy for each unit of secondary lead would be 25% of the price for a permit to use virgin lead. Thus, it is as though the government collected a virgin materials tax equal to the permit price and used all that the money it collected to give a recycling subsidy. This tax/subsidy combination is revenue-neutral, unlike the deposit-refund, which raises revenue from the government on unrecovered lead.

The effect of a recycled content standard, whether tradeable or not, is to increase the recycling rate. The policy is also likely to increase the cost of lead overall. However, because all the money paid by firms to use primary lead is returned as a subsidy to secondary lead, the overall effect on the price of lead to users is smaller than for a deposit-refund or virgin materials tax. As a result, the reduction in lead use will be smaller than for these other

¹¹For more discussion of tradeable recycled content standards, see Dinan (1993) for paper and Macauley et al. (2001) for CRTs.

policies.

Producer responsibility requirements. Under producer responsibility, the government sets rules requiring that producers, individually or through proxies, take back products. These government may set varying types of requirements for the nature of the take-back, the share of the products the producer successfully take-back, and the required fate of products collected. For lead, the principal example of this sort of policy is the EU’s WEEE Directive, which sets take-back and recovery requirements for various white goods and consumer electronics, including CRTs.

Given the complexity of producer responsibility, it deserves more extensive discussion than I can accord here. However, it is worth mentioning that the requirements have some features of the tax/subsidy combinations discussed here, with the levels of the effective taxes and subsidies set by market conditions and the stringency of the government’s quantitative requirements.¹² The policies create large effective subsidies to recycling because of the funds the producers must spend on collection of scrap products and their recovery. The revenue for these effective subsidies comes from the producers, who presumably increase the prices of their products to reflect the expected collection and recovery costs. Thus, there is also an effective tax on the purchase of the product. Under the WEEE Directive, producers are allowed to collect this cost in “visible fee” for the first 8 to 10 years; thereafter, the price premium will continue to exist, even without the explicit label.

2.3 Comparison of private costs

Reducing lead in municipal solid waste will require firms and consumers to bear costs and exert effort. Such costs include the time spent returning products for recycling, costs of transporting, storing and disassembling scrap goods, and of rerefining lead. It also includes costs from lost opportunities as a result of source reduction. These costs may be worthwhile to avoid the environmental consequences of battery disposal, but one goal of good policy is to make them as low as possible for any given reduction in lead disposal. The costs will be lowest for a deposit-refund and higher for a tax on all lead, a recycling subsidy, or a recycled content standard.

Table 1 shows the reason for these cost differences. By encouraging users to take advantage of both options for source reduction and recycling, the deposit-refund and virgin material tax will reduce lead with the least cost to the society. However, approaches such as a subsidy or tax on all lead leave some low cost options for lead disposal reduction op-

¹²Unlike an implicit tax or subsidy, these effective charges may vary by producers.

Table 1: Summary of effects of indirect incentive policies

| Policy | Effect on: | |
|---------------------------|------------|----------------|
| | Use | Recycling rate |
| Deposit-refund | ↓ | ↑ |
| Tax on all lead | ↓ | — |
| Virgin materials tax | ↓ | ↑ |
| Recycling subsidy | ↑ | ↑ |
| Recycled content standard | ↓ | ↑ |

Note: Narrow arrows indicate effects that will typically be smaller than thicker arrows.

tions unexploited and, thus, require higher costs to accomplish the same disposal reduction. This ranking will hold regardless of whether the policies are restricted to batteries or also encompass consumer electronics and other recyclable lead uses.

3 Empirical effects of battery recycling policies

To illustrate the magnitude of differences in the costs, this section will focus on application of the policies only to lead in batteries. As discussed above, batteries are the most common application of these policies so far.

3.1 Responsiveness to price

To estimate the magnitude of these effects, we also need some information on how participants in the market respond to changes in price — that is, price elasticities. The first question is how sensitive recycling rates are to prices. In Sigman (1995), I estimate the responsiveness of battery recovery to lead scrap prices in the United States from 1954 through 1992. In this research, a 10% increase in the scrap lead price increases recovery rates by about 2% (which would be about a 1 percentage point increase in the recovery rate in 1988).¹³ Thus, there is evidence that recovery of lead from batteries is responsive to price, providing support for

¹³That is, the recovery rate elasticity is .2. The long-run elasticity is estimated at .1, lower perhaps because of depletion of stored batteries. For comparison, Anderson and Spiegelman (1976) find supply elasticity for all secondary lead of .48 for the period 1954–1972 and Fisher, Cootner, and Bailey (1972) find a long-run supply elasticity for secondary copper of .31–.33.

incentive-based policies as an effective approach to encourage recycling.

A second price elasticity that is important is the responsiveness of primary lead supply to prices. In Sigman (1995), a 10% increase in refined lead prices increases primary lead supply by 8% in the long run.

Finally, the calculation requires an estimate of the sensitivity of demand for lead in batteries to price. In Sigman (1995), I did not find a statistically significant elasticity of lead in batteries to price. This insensitivity may reflect the absence of any good substitute for lead and is consistent with the lack of attention devoted to the lead content of batteries in engineering papers like Salkind et al. (1984). However, it also suggests that consumers do not adjust their consumption of batteries to lead costs. One might expect more rapid replacement of old batteries when they are cheaper. Earlier studies do estimate some sensitivity of battery lead demand to its price. Moroney and Trapani (1981) estimate a 2% reduction in demand for lead in batteries for a 10% increase in price and Anderson and Spiegelman (1976) estimate 2.1% reduction in demand for all lead (not just lead in batteries) for 1949–72. The calculations below use a 1% reduction in demand for the 10% price increase because it is a midpoint of the studies and reflects the likelihood of at least some response in battery demand.

Considering lead demand sensitive to price is critical to comparison of the policies. If demand is insensitive to price, then there are no opportunities for source reduction. In this case, the tax on lead is ineffective as a way of getting rid of lead in waste. In addition, recycling subsidies do not have any perverse effect on lead use, thus they are not any more costly than the other policies.

3.2 Empirical effects of the policies

Table 2 calculates the impacts of three policies for reducing disposal of lead in batteries in the United States. The table uses 1988 price and quantity data: the price of refined lead is \$.3714 per pound and the recovery rate is .665.¹⁴ I use data from 1988 because it predates the state taxes and deposit-refunds and thus shows the change from the market with and without lead disposal policies. Table 2 compares the policies when they are implemented to achieve the same reduction in lead disposal. It is worth highlighting that the appropriate target is not the recycling rate, but rather the amount of lead incinerated or disposed because it is these activities that potentially create health and environmental harms.

¹⁴This recovery rate is my estimate (Sigman, 1995). It is somewhat lower than the recovery rate estimated by EPA described above.

Table 2: Estimated effects of policies in the United States in 1988

| | Decrease in lead disposal | | |
|--------------------------------------|----------------------------------|------------------|------------------|
| | 10% reduction | 20% reduction | 50% reduction |
| Deposit-refund | | | |
| Tax level (per pound of lead) | \$.121 | \$.242 | \$.606 |
| Deposit per battery | \$2.82 | \$5.64 | \$14.11 |
| Revenue (million 1988 dollars) | 77 | 136 | 214 |
| Private costs (million 1988 dollars) | 8.5 | 34 | 214 |
| Recycling subsidy | | | |
| Subsidy level (per pound of lead) | \$.152 | \$.303 | \$.758 |
| Revenue cost (million 1988 dollars) | 227 | 483 | 1,430 |
| Private costs (million 1988 dollars) | 15 | 59 | 367 |
| Recycled content standard | | | |
| Price of permits(per pound of lead) | \$.086 | \$.173 | \$.432 |
| Level of standard (r^*) | .700 | .735 | .840 |
| Private costs (million 1988 dollars) | 9.5 | 38 | 239 |

Calculations use a recovery rate elasticity of .2, primary lead elasticity of .8, and lead demand elasticity of -.1, and 1988 prices and quantities. See Sigman (1995) for the equations used to calculate the values.

Deposit-refund. Table 2 indicates that the deposit-refund rates necessary to achieve moderate reductions in lead disposal are consistent with those currently in use. If implemented correctly, a \$5 per battery deposit-refund could result in almost a 20% reduction in battery lead disposal. The results suggest that the private costs of a deposit-refund to reduce lead disposal by 20% would be about \$34 million.

However, this figure may underestimate the true costs of accomplishing disposal reductions through these policies because of implementation issues that are omitted from the model. First, the deposit-refunds used in the U.S. do not vary with the lead content of the battery, unlike the deposit-refund shown here. A fixed deposit-refund per battery provides weaker incentives for source reduction than a variable one because the policy does not encourage consumers to choose lower lead batteries. Thus, the costs of achieving the reduction may be greater. However, the discussion of price responsiveness above suggests that reducing the lead content of batteries does not have much potential as an avenue of source reduction, so lack of these incentives may not be that important.

Second, Table 2 fails to consider the administrative and compliance costs of the policies. These costs may be substantial for a deposit-refund if it is administered at the retail level because large numbers of firms must comply. However, administering the program at the level of battery producers could make these costs much lower.

Recycling subsidy. Table 2 suggests that a subsidy has significant disadvantages relative to a deposit-refund. For a reduction in disposal of 20%, a subsidy costs almost twice as much as the least cost approach, \$59 million compared to \$34 million. In practical terms, however, the revenue required to support a subsidy may present a more formidable obstacle than this difference in private costs. While the deposit-refund would collect \$136 million in revenue for a 20% reduction, the subsidy would require expenditures of \$483 million. Subsidy expenditures are high even for small reductions because of the large amount of lead recovered initially.

On the other hand, the administrative costs of this program are not likely to be substantial. A limited number of firms reprocess battery lead or purchase this lead. In addition, reported quantities of recycled lead are easily verified, making this subsidy less vulnerable to sham operations than other pollution abatement subsidies.

Recycled content standard. In Table 2, the recycled content standard costs only slightly more than a tax or deposit program. For a 20% reduction, it costs \$38 million compared to \$34 million for the best approach. The recycled content standard implies only a small subsidy to recycled lead (because initial recovery rates are high) and therefore does not differ

dramatically from a deposit-refund.

However, Table 2 shows the best possible case for the recycled content standard by allowing trading of permits.¹⁵ Costs without trading will probably be higher because firms have different opportunities for substitution of recycled lead for virgin lead.

Even if the government does allow trading to meet a recycled content standard, the costs may exceed the estimates in Table 2. Substantial transactions costs appear to have hampered other environmental permit markets. However, recycled content permits are likely to be subject to fewer transactions obstacles than other permit markets. Firms are already familiar with their patterns of reliance on recycled and virgin lead, unlike pollutants to which permits have been applied previously. In addition, there should be little need for intrusive regulatory oversight of the market given these ease with regulators should be able to monitor lead usage. Thus, transactions cost may not greatly elevate the costs of a tradeable recycled content standard.

3.3 Results with other uses of lead

Although the figures above are designed to address policies restricted to batteries, this section makes some qualitative observations about how the effects and comparison of the policies would change if the policies were applied to additional uses of lead. First, I consider extending the policies to other recyclable uses of lead, such as consumer electronics and wheel weights in motor vehicles. The overall ranking of the policies will not change with this extension, but the relative costs of the policies may change. Second, I discuss how the ranking may change if policies apply to nonrecyclable uses of lead, such as gasoline additives or ammunition.

Additional recyclable uses of lead. In comparing the policies, the principal effect of adding other recyclable uses of the lead will be to change the price elasticities of lead demand and recycling. Demand for other uses of lead, such as consumer electronics, is likely to be more sensitive to price because of more available substitutes. For example, higher lead costs may raise the price of CRTs and encourage the substitution of flat panel displays. Consumers may also choose to extend the life of televisions and computer monitors, unlike batteries for which replacement is usually forced by the demise of the old battery. A somewhat dated estimate supports the notion of greater price sensitivity in other uses: Wise (1979) estimates an reduction in demand of 3% for a 10% increase in price for lead uses other than batteries,

¹⁵The inefficiency of the tradeable permit system may come as a surprise. The result arises because of the specific permit system analyzed, namely one based on recycled content standards. It would be possible to design a tradeable permit system more comparable to the deposit-refund. Such a system would require permits for lead disposal rather than virgin lead use. It would, however, face the enforcement problems of other direct approaches.

ammunition, and gasoline additives in the United States. If demand is more sensitive to price, the relative costs of policies that do not take advantage of demand reductions will rise: the recycling subsidy and the recycled content standard will be more costly relative to the other policies.

In addition, there may be fewer recycling opportunities for sources of lead other than batteries. In particular, it seems that lead prices will have to rise dramatically for recycling of CRTs to become economical. The lack of recycling options also favors policies, such as the deposit-refund and lead taxes, that can take advantage of source reduction relative to recycling subsidies and recycled content standards.

Another issue that arises in considering other uses of lead is the loss from using a fixed deposit-refund relative to a deposit-refund that varies with the lead content of the product. For batteries alone, this loss might not be very great because changes in product design will account for little source reduction. However, with other products, product design changes may be important ways to reduce lead. A deposit-refund that is invariant with the lead content of batteries provides no incentive for this substitution and thus will be much substantially more costly for any reduction in disposal than deposit-refund that varies with lead content.

Nonrecyclable uses of lead. Expanding the policies to encompass nonrecyclable uses of lead complicates the policy comparison. With these other uses, we can no longer justify a simple cost-effectiveness analysis, which compares the policies for a given reduction in lead in municipal solid waste. To compare the broader policies, we also need an assessment of other environmental exposures to lead.

For example, a deposit-refund for which the deposit was paid on all uses, but the refund for recycling only, would not only decrease disposal of lead but also other releases to the environment. This breadth could be an advantage of the policy if it reduced the air pollution from leaded fuel or wildlife exposures to lead ammunition. However, it could also be a disadvantage if it unnecessarily discouraged use of lead in applications, such as construction or X-ray shielding, for which there was little risk of environmental exposure. A broad policy could also be excessive if non-municipal solid waste releases of lead are already adequately addressed by other environmental policies, such as policies for air pollution. For example, a recycling subsidy could dominate an excessively broad deposit-refund because the subsidy will affect only lead in municipal solid waste. If so, restricting the deposit-refund only to recyclable uses of lead would reestablish its superiority.

Non-recyclable uses of lead also affect the best design of a recycled content standard. Recycled content standards may apply only to the lead content of recyclables or to all uses

of lead. A standard solely for recyclable products also may be ineffective, however. It may shift recycled lead used by other industries into manufacture of batteries and other recyclable products. Such a standard may not reduce disposal at all. To accomplish reductions, therefore, it may be necessary to apply the recycled content standard to all lead.

4 Desirability of reducing lead in municipal solid waste

Finally, we should consider the overall desirability of policies to reduce lead in municipal solid waste. Two previous studies raise questions about whether even the most cost-effective policies would have a beneficial effect. First, Walker and Wiener (1995) summarize a 1991 U.S. EPA assessment of an increase in the recycling of lead-acid batteries.¹⁶ As reported in Table 3, EPA found that increasing the battery recycling rate from 85 to 95 percent dramatically reduced air emissions of lead because a great reduction in lead emitted from municipal waste combustion facilities and a smaller reduction in emissions at primary smelters.¹⁷ Offsetting these gains is an increase in emissions from secondary smelters.¹⁸

When the EPA examined the number of children predicted to have high blood lead, however, the gains from increased recycling look less clear. The reduction in lead emissions from combustion has little effect on the number of children with high blood lead for two reasons. First, the populations near the facilities had low levels of blood lead, making the number of children to cross the 10 $\mu\text{g}/\text{dl}$ threshold small. Second, although many tons of air pollution are avoided, the reduction is spread over 186 facilities, so the change in exposure at any single location is small.

In addition, the trade-off between primary and secondary smelting does not appear favorable. Secondary smelting puts more children over the threshold because secondary smelters are located near people (who supply their raw materials), whereas primary smelters tend to be located near mines and are thus more remote. As a result, increased recycling may have negative overall implications for children's health.¹⁹

¹⁶The documents are U.S. EPA "Lead Acid Battery Recycling Risk Assessment" and "Cost Analysis of Options to Enhance Lead Acid Battery Recycling," both drafts from 1991, but my discussion is based on Walker and Wiener's summary.

¹⁷It also considered reduced exposure to contaminated groundwater from recycling more batteries. However, the EPA concluded that this pathway did not pose a significant health risk even when batteries are sent to older landfills.

¹⁸These emissions occur despite existing U.S. air pollution regulations and might be worse in countries with less stringent regulations. Engineering estimates suggest that all environmental regulations in effect in 1988 added \$0.068 per pound (18% of its price) to the cost of secondary lead (U.S. Office of Technology Assessment, 1989).

¹⁹The figures shown here presume that the increased secondary smelting occurs in the U.S. The EPA also presented a variant in which this secondary smelting occurs abroad, in which case the figures look more favorable for reducing lead, but exposures of children outside the U.S. are not counted.

Table 3: U.S. EPA estimates of lead air emissions and children at risk from increased battery recycling

| Pollution source | 85% recycling | 95% recycling | Change |
|--|---------------|---------------|-------------|
| Air emissions of lead (tons) | | | |
| Primary smelters | 498 | 398 to 473 | -100 to -25 |
| Secondary smelters | 509 | 560 | +51 |
| Municipal waste combustion | 736 | 282 | -454 |
| Children with blood lead $\geq 10\mu\text{g}/\text{dl}$. | | | |
| Primary smelters | 226 | 198 to 219 | -28 to -7 |
| Secondary smelters | 525 | 563 | +38 |
| Municipal waste combustion | – | – | -2 |

Source: Walker and Wiener (1995).

Note: The range for primary smelters reflects different assumptions about the change in primary production. The low emissions/children at risk values assumes primary lead production falls by the amount of the increase in secondary production. The high emissions and risks assumes primary lead smelters reduce production by only 25% of that increase (and perhaps export the rest).

Table 4: Population densities near lead smelters around the world

| | Primary smelters | Secondary smelters |
|---|------------------|--------------------|
| Number of smelters | 74 | 84 |
| Number located for density calculation | 46 | 55 |
| Population density (people/km ²) within 20 km | 393 | 915 |

Source: Author's calculations.

An important caveat in interpreting these results is that the EPA does not include any source reduction in the change it analyzes. Indeed, total U.S. lead output increases in the scenario that leads to the worst case for primary smelting. Such an increase might occur with a recycling subsidy, but would not characterize the other policies discussed here. A policy such as a deposit-refund that also has the potential to reduce lead use might result in greater health gains.

As a first step toward extending this analysis to other countries, Table 4 presents my calculations of the population near secondary and primary lead smelters around the world.²⁰ The results do suggest substantially higher population densities near secondary smelters than primary smelters. Average 393 people per square kilometer live near primary lead smelters, but more than twice as many, 915, near secondary smelters. Thus, there is reason for concern that the trade-off identified by the EPA applies internationally as well.

Macauley et al. (2001) provide another previous evaluation of the overall desirability of policies to reduce lead in waste. They study the overall costs and benefits of several policies that address end-of-life computer monitor CRTs. For example, they find that a complete ban on land disposal and incineration of monitors (assuming it is enforceable) saves \$4 million per year, but costs nearly \$300 million per year. Some of these costs are from increased recycling; however, recycling is so expensive that Macauley et al. estimate that most users would store their old monitors, unless recycling receives substantial subsidies. None of the policies Macauley et al. consider pass a cost-benefit test. The most favorable policy in their analysis is the most targeted to the health harms: excluding CRTs from incineration. Even this limited policy costs \$38 million for benefits of only \$3 million.

²⁰Alliance to End Childhood Lead Poisoning (1994) provides the names of the town in which lead smelters are located. We were able to attach latitude-longitude coordinates to these names for 46 primary and 55 secondary smelters, representing 62% and 65% of the total respectively. Population densities within 20 kilometers were then calculated using ArcGIS and data from the Gridded Population of the World, Version 2 (Center for International Earth Science Information Network, 2000).

5 Conclusion

Governments wishing to reduce lead in waste have a variety of policy options. This paper considers policies that can reduce waste disposal when direct restrictions are too difficult to enforce. Successful policies address disposal at two levels, encouraging recovery of lead and discouraging its consumption. The deposit-refund and virgin material tax are low cost policies because they create both types of incentives. In contrast, a subsidy for recycling is more costly because it decreases the price of lead to users and thus encourages lead consumption.

The policy analysis applied to programs aimed at the recovery of lead from batteries and used supply and demand parameters estimated from the U.S. lead market. This analysis suggests two conclusions. First, price-based recycling policies can effectively increase lead recycling. Second, this analysis reveals substantial empirical differences among the policies in the costs of accomplishing the same reduction of lead in waste. A recycling subsidy entails nearly twice the private costs of a deposit-refund. The general ranking presented also applies to policies aimed at other sources of lead in municipal solid waste, including consumer electronics. Indeed, expanding the sources covered probably increases the importance of source reduction and the cost differences among the policies.

Despite the effectiveness of price-based lead reduction, earlier studies surveyed here suggest the need for caution before deciding to pursue policies that reduce lead in solid waste. For countries with already high recovery rates of lead from batteries, such as the U.S., it may be that the environmental gains of reduced lead disposal are not high enough to merit the cost and environmental consequences.

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