A Comparison of Public Policies for Lead Recycling

by

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Abstract

Policies that encourage recycling may be used to reduce environmental costs from waste disposal when direct restrictions on disposal are difficult to enforce. Four recycling policies have been advanced: (i) taxes on the use of virgin materials; (ii) deposit/refund programs; (iii) subsidies to recycled material production; and (iv) recycled content standards. This study analyzes the structure of these policies and ranks them in terms of the private costs necessary to achieve a given reduction in disposal. The policies are then examined in the empirical context of the recycling of lead from automobile batteries. Elasticities for primary and secondary lead supply and demand are estimated in order to simulate the effects of lead recycling programs. The results suggest that price-based policy mechanisms can be successful in increasing lead recovery and that the difference in efficiency between the four approaches is substantial.

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A number of environmental policy issues concern costs from the disposal of materials. Although the direct policy response to such environmental costs is to impose fees or quantity restrictions on disposal, these measures may prove extraordinarily difficult to enforce and may even exacerbate environmental damage by encouraging illegal dumping. The government can respond to these difficulties with policies designed to increase recycling of materials. Possible targets for recycling policies include industrial waste and also wastes generated by households, which may pose greater enforcement difficulties. Hazardous household wastes, like used automobile oil, chlorofluorocarbons in refrigerators, or batteries, are of particular concern, but nonhazardous household wastes, such as newspapers and plastics, may also fall into this category.[2]

Several studies have examined the general problem of restricting undesirable emissions when a direct Pigouvian tax on these emissions is impossible.[3] In the case of a disposal externality with inexpensive recycling, a special set of taxes and subsidies become available that discourage disposal by inducing recycling. This paper examines several policies that incorporate these recycling incentives: (i) taxes on the use of virgin materials; (ii) deposit/refund programs, familiar from bottle bills; (iii) subsidies for recycling; and (iv) recycled content standards, which require recycled materials to compose a certain fraction of products. Although many policies of this kind are under consideration at both federal and state levels, few studies offer guidance to policy-makers in choosing among them.[4]

In this paper, I examine public policies for recycling of lead from automobile batteries. All four recycling policies have been advanced for lead-acid vehicle batteries in response to the health and environmental risks posed by battery disposal. Several states rely on deposit/refunds on vehicle batteries: Rhode Island implemented a $5 deposit/refund in 1988,

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[2] A survey of waste disposal issues and policy responses can be found in Stavins (1991) and Office of Technology Assessment (1989).

[3] For example, Diamond (1973), Green and Sheshinski (1976), and Sandmo (1976).

[4] Exceptions include analysis of deposit/refund systems by Bohm (1979) and Porter (1978, 1983). Studies comparing policies are rarer. Menell (1990) conducts a theoretical exploration of polices for disposal of nontoxic household waste. His study considers disposal fees to be enforceable and does not focus on differences between recycling policies. The Congressional Budget Office (1991) discusses informally the issues that arise in comparing these policies.

as did Connecticut in 1990, Washington and Idaho in 1991, and Michigan will begin a $6 deposit/refund in 1993. At the same time, the Environmental Protection Agency has considered a tax on virgin lead (*Environment Reporter*, 1991), and legislation for a national recycled content standard has been introduced into Congress.[5] Direct subsidies for proper disposal are often suggested for hazardous materials in general; for lead-acid battery recycling, none is currently under debate, although a number of states have indirect subsidies through tax privileges for secondary lead producers (OTA, 1989). My analysis of these approaches to recycling policy suggests that they are not interchangeable; the policies can be ranked on efficiency grounds.

A view commonly expressed in the environmental economics literature is that recycling policies like these, which are based upon altering prices, may have only limited success. Studies of waste paper recycling, which compose the majority of empirical work on recycling programs, have suggested that raising secondary materials prices will not change recycling behavior.[6] My results suggest the opposite conclusion for lead recycling: the recovery of lead from auto batteries is sensitive to prices. Thus, my empirical analysis suggests that price-based policy mechanisms can be successful in encouraging recycling of auto batteries.

The paper is divided into five sections. The first section provides background on the industry and on the problem posed by automobile battery disposal. Lead-acid automobile batteries are an important component of the hazardous waste generated by households, because lead is highly toxic and a large volume of it is disposed in batteries. Batteries are both the largest use of lead and, through recycling, one of the most important sources of lead supply.

The second section compares virgin-materials taxes, deposit/refunds, recycled content standards, and subsidies. The tax and deposit/refund are equivalent in the incentives they create, but the recycling subsidy and recycled content standard each have different effects. The tax-deposit approach achieves a given reduction in the environmental costs from waste

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[5] As HR 5359 in 1990.

[6] For example, Edwards and Pearce (1978) study waste paper recovery in the United States between 1947 and 1973. On the basis of their elasticity estimates they conclude that "if recycling is to be encouraged for whatever reason, the weapons must be other than direct manipulation of prices (p. 248)." A similar statement may be found in Edgren and Moreland (1989).

disposal with lower private costs than the other programs. Recycling subsidies are the most costly way of achieving the same target, whereas recycled content standards, which act as a hybrid of the tax and subsidy, have intermediate costs.

The third section calibrates this analysis for policies to encourage auto battery recycling. Primary and secondary lead supply and demand elasticities are estimated for this purpose. The results suggest that lead recovery has been moderately responsive to prices in the post-war period.

In the fourth section, these estimates are used to simulate public policies for reduction in lead disposal. The results suggest that significant reductions can be achieved with moderate levels of government intervention. In addition, the disparities in the costs imposed by different policies may be substantial in magnitude. Finally, a brief conclusion considers other environmental issues to which this analysis is pertinent and suggests directions for future work.

1 The lead industry

To provide background for the discussion of recycling policies, this section describes some of the salient features of the lead industry, including historical trends that will be useful in interpreting the regression results in the third section. It also explains in more detail the nature of the environmental damage caused by disposal of lead-acid batteries.

1.1 Uses of lead

Table 1 illustrates recent trends in several important uses of lead. Policy initiatives have reduced lead use in many products in response to medical evidence of lead toxicity at low exposure levels. The Lead-Based Paint Poisoning Act of 1971 forbade use of leaded paint on surfaces accessible to children; the use of lead in indoor paint was banned entirely beginning in 1977. The EPA began a phase-out of lead additives in gasoline (tetraethyl and tetramethyl lead) in 1975. Recent efforts have focused on reducing the use of lead solder in plumbing.

Table 1: Some principal uses of lead in the U.S., 1960-88 (metric tons)

	1960	1970	1980	1985	1988
Ammunition	39,532	65,976	48,662	50,233	52,708
Cable covering	54,749	46,059	13,408	12,270	16,170
Solder	54,443	63,237	41,366	24,441	19,064
Storage batteries	320,414	538,371	645,357	840,940	955,623
Pigments (total)	89,387	89,571	78,430	74,852	62,524
Paints	–	–	45,361	44,146	–
Gasoline additives	148,621	252,656	127,903	45,694	–
Total (all uses)	926,392	1,234,272	1,057,967	1,133,737	1,230,732
Percent used for batteries	34.6%	43.6%	61.0%	74.2%	77.6%

Source: U.S. Bureau of Mines

However, not all the changes in lead use resulted from health regulations. Technological changes also explain some of the pattern of lead use in Table 1. For example, lead for cable covering has been replaced by plastic for some uses.

Despite these reductions, many products continue to use lead. Lead's primary use is for lead-acid storage batteries: 956,000 metric tons were used in 1988, representing 78% of the total consumption of lead in the U.S. for that year. Most lead-acid batteries are used as starting-lighting-ignition batteries for motor vehicles (84% in 1988). About 5% of the remaining lead is used for motive power in electric vehicles, like in-plant fork-lifts. Other industrial uses, such as uninterruptable power supply for large computer systems and standby power supply for emergency lighting, compose the remainder. Storage batteries made from alternative materials will probably remain much more expensive than lead-acid batteries, while use of batteries in electric vehicles and computer systems may rise substantially (Woodwell, 1985). Thus, the volume of lead in storage batteries should continue to grow and its disposal raise problems in the future.

1.2 Supply of lead

Primary supply Primary (or virgin) metal accounted for an average of 49% of U.S. supplies in the 1980s. Since the early 1970s, a single geologic formation, the Vibernum Trend in southeastern Missouri, has supplied about 90% of the U.S. mine production. Most production is from exclusively lead ores, but lead is also mined as co-product with zinc or silver. Firm concentration in the primary industry has become high since the 1970s.[7]

Secondary supply The secondary lead market accounts for the remaining half of refined lead. Used batteries constitute the bulk of the recovered scrap (80% in 1988). 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 $5; but in the early 1980s, 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 to battery "breakers," who remove the plastic cases and drain battery acid before transfering them to secondary smelters who 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. "Recovery rates" — the percentage of available scrap that is recycled — cannot be measured directly, because the number of defunct batteries is unknown, but Figure 1 shows estimates of these rates. The rates experienced a high of 86% in early 1960s, but they have since generally declined. The lowest rates were observed in the early 1980s, falling to less than 50%, but rates rebounded to near 70% later in the 1980s.[8]

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[7] The Census of Manufactures withholds the concentration ratios for primary lead; as an indication, however, four firms held 86% of the U.S. refining capacity in 1988. High concentration in primary production may affect secondary production, as analysis of the Alcoa case has suggested (e.g. Swan, 1980). However, the "Alcoa problem" is unlikely to be a factor in the lead industry. Both primary and secondary producers contribute to the next period's supply of scrap lead, because lead may be re-recovered multiple times. Thus, the relevant market for determining this intertemporal market power is both primary and secondary production, of which primary lead composed only 48% in 1988.

[8] Some experts claim that the rates have never dropped this low, barely dipping below 90%. However, a 1983 survey of generators of small quantities of hazardous waste suggests that such figures are too high (Abt

Figure 1: Recovery rate of lead from auto batteries, 1954-88

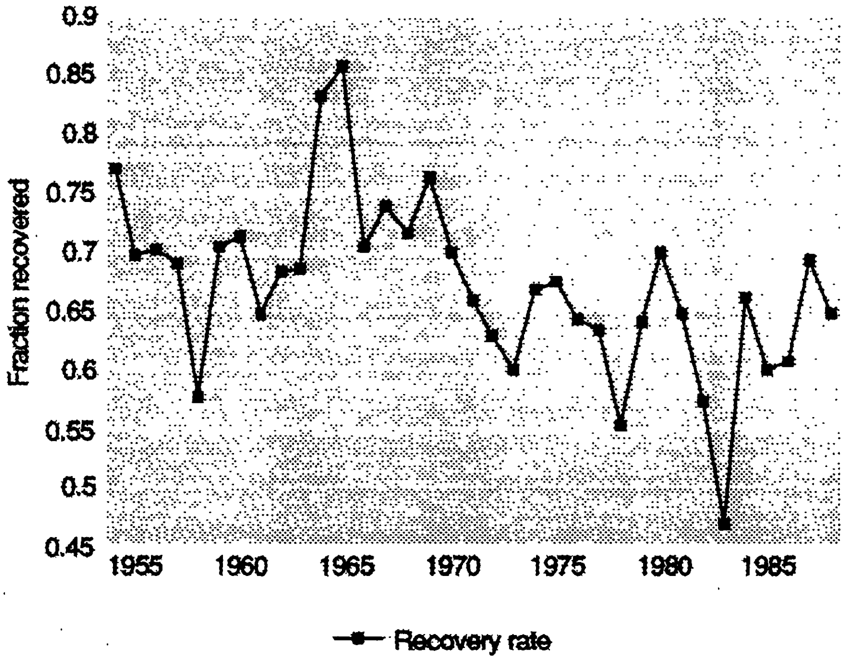
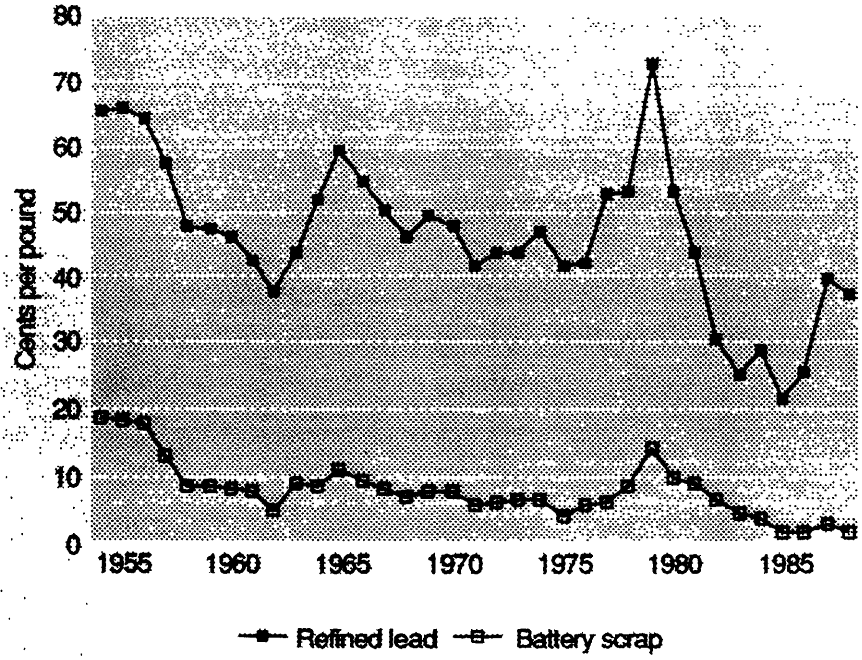


Figure 2: Prices of refined lead and battery scrap in 1988 dollars, 1954-88



The cause of the dramatic decline in recovery rates in recent years is not clear. The price of lead was low at this time (Figure 2) and a number of much stricter environmental regulations for lead were introduced, perhaps contributing to the low price. Many of these regulations affected primary producers at least as strongly as secondary producers and thus would not explain the decline in the fraction of recycled to total lead produced.[9] One regulation that might have affected recycling particularly strongly was the reclassification of used batteries as hazardous waste under the Resource Conservation and Recovery Act (RCRA). This classification imposes large costs on facilities which must store batteries before reclaiming them (dealers and transporters are exempt from these regulations). RCRA requires firms to acquire permits and liability insurance for hazardous waste sites, which may be costly. Another factor that may have contributed to the decline in recovery rates may be the growth in sales of replacement batteries through do-it-yourself auto parts outlets. These batteries are more likely to be disposed by households than batteries replaced by battery dealers who often sell defunct batteries to scrap dealers.

1.3 Lead-acid batteries in municipal solid waste

Despite generally high recycling rates, auto batteries represent the majority of lead in municipal solid waste (see Table 2). In 1985, about 220,000 tons of battery lead were disposed in municipal waste, accounting for 77% of the lead disposed there (Franklin Associates, 1989). This lead may pose a serious threat to public health as well as to environmental quality.

Medical evidence points to toxic effects from lead at low levels of exposure. Studies have gradually lowered the level of blood lead at which health harms are believed to arise; toxic

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Associates, 1985) The generators of used batteries (primarily vehicle maintenance establishments) sent 89% of batteries they collected to off-site recyclers, with the remainder mostly sent to solid waste landfills. This represents an upper bound on recycling rates because some household change and dispose their own batteries. In addition, firms wary of betraying illegal disposal of batteries may overreport the number they recycle.

[9] The new standards include: National Ambient Air Quality Standards for lead, fully implemented in 1988; OSHA standards for employee blood lead levels, effective 1983; OSHA standards for air exposure levels, effective in 1986 for secondary smelters and in 1991 for primary smelters-refiners; and effluent limits under the Clean Water Act in 1984 for smelters and battery producers. Engineering estimates suggest that all environmental regulations in effect in 1988 added 6.8 cents/lb (18% of its price) to the cost of secondary lead (OTA, 1989).

Table 2: Sources of lead in municipal waste, 1985

	Short tons	Percent
Lead-acid storage batteries	221,954	77.3%
Consumer electronics	47,011	16.4%
Glass and ceramic products	6,911	2.4%
Plastics	3,466	1.2%
Cans and other shipping containers	3,007	1.0%
Pigments	2,368	0.1%
Other	2,411	0.1%
Total	287,138	100%

Source: Franklin Associates, *Characterization of Products Containing Lead and Cadmium in Municipal Solid Waste in the United States, 1970 - 2000*, 1989

effects are recognized at blood lead levels that now characterize 17% of urban children under six (Agency for Toxic Substances and Disease Registry, 1988). With low levels of exposure, children and fetuses are at risk of IQ deficits, impaired reaction time, and other neurological problems; epidemiological studies find higher school drop-out rates and lower achievement long after initial exposure.

When disposed in a landfill, automobile batteries can expose the population to lead.[10] In the landfill, battery cases eventually break, releasing liquids tainted with lead that can contaminate groundwater. The extent of the danger is not well documented. Of the 158 municipal landfills on the Superfund National Priority List, 22% have released lead, suggesting that the potential exists for groundwater contamination at these sites and other municipal landfills (ATSDR, 1988).

In addition to health risks, households that dispose batteries as ordinary trash impose financial costs on other households. In recent years, landfill siting and design have been modified in an attempt to diminish environmental harms from landfill leachate. Reducing the toxicity of the leachate might relax constraints on landfill design and location and thus lower

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[10] An additional concern arises in areas where municipal solid waste is incinerated, because lead from batteries can escape in the air emissions from incineration. While state-of-the-art incineration technologies provide sorting too thorough for auto batteries to be incinerated, some existing incinerators have little or no sorting. Thus, some batteries may be incinerated (Franklin Associates, 1989) and give rise to substantial environmental costs.

the average costs of trash disposal.

2 Analysis of recycling policies

In response to these risks from lead in solid waste, public policies to reduce battery disposal have been proposed and implemented. Two general approaches can be taken. The first approach addresses the externality directly, by imposing requirements on battery disposal. In practice, state legislatures have chosen an extreme quantity restriction, banning disposal of batteries in municipal landfills.[11] In principle, the government could also charge fees for battery disposal reflecting its environmental costs. Unlike the more common quantity mechanism, this price-based policy would allow some battery disposal to continue if this outcome is socially efficient. The difficulty with such programs is the near impossibility of enforcing them. Consumers may dump batteries surreptiously rather than bear the costs of proper disposal.[12]

The second policy is to strengthen the incentives for auto battery recycling. Such policies can take several different forms, including virgin-materials taxes on primary lead, deposit/refunds on auto batteries, recycled content standards for batteries and subsidies to the secondary lead industry. This section analyzes these four policies. It begins with a framework for a market with recycling, then examines the simple analytics of the recycling programs, and finally compares their costs.

2.1 Framework for a market with recycling

Recycling policies can be compared in a partial-equilibrium framework in which there is a demand for lead in auto batteries and two sources of supply, virgin and recycled. Consumers purchase lead from both sources at a single price q, because virgin and recycled lead are nearly

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[11] 27 states had such laws in 1991. A national land disposal ban on vehicle batteries has been considered by Congress.

[12] There is no systematic evidence about whether households do obey disposal restrictions and the anecdotal evidence is mixed. Seattle has experienced virtually no increase in illegal disposal as a result of requiring payment of fees for garbage collection (Stavins, 1991), but the *New York Times* (1988) reports that after the imposition of fees, illegal disposal has been a problem in New Jersey.

perfect substitutes for many uses. The producer prices of virgin and recycled lead, denoted p_V and p_R respectively, may differ. Batteries are assumed to be the only use of lead for simplicity; an appendix shows the analytical results for the more general case where there are other uses of lead.

In the presence of recycling, demand depends not only on the consumer price of lead, but also on the expected price of scrap battery lead, p_S, when the battery wears out. For batteries that are recycled, the effective price of battery lead is $q - \delta p_S$ (where δ is a discount factor), but for batteries that are disposed, the price of lead is q. Then, the total demand (TD) for battery lead can be written as

$$TD(q, p_S) = \hat{r}(p_S)D(q - \delta p_S) + (1 - \hat{r}(p_S))D(q). \tag{1}$$

where $\hat{r}(p_S)$ is the fraction of batteries that are recycled and p_S is the expected price of scrap when returned.

An interpretation of this demand structure is that households vary in the time and effort required to recycle their batteries. Households choose to recycle if the effort cost of returning a battery is less than the scrap value of the battery. Then, the $\hat{r}(p_S)$ in equation (1) reflects the cumulative distribution of recycling costs in the population that purchases batteries.[13] Alternatively, costs of returning batteries may vary geographically across battery dealers, resulting in a similar demand pattern.[14]

For the market to clear, this demand must equal the supply of virgin and recycled lead,

$$TD(q, p_S) = V(p_V) + R(p_R). \tag{2}$$

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[13] Suppose each household has an average cost of recovery per unit of lead, e_i which is constant for the household, but varies across households. Household i recycles if $e_i \leq p_S$. Further, suppose that the lead content per battery is price sensitive, but the number of batteries demanded is not because batteries are essential to operating a car, but only a trivial fraction of its cost. If $F(e_i)$ is the cumulative distribution of effort costs in households that purchase batteries, $\hat{r}(p_S) = F(e_i \leq p_S D(q - p_S))$.

[14] In the early 1980s, there may have been few batteries recovered in the Pacific Northwest, after the closure of the only lead recycling plant in the region (Putnam, Hayes and Bartlett, 1987). Further, transporting batteries may constitute a large fraction of their recovery cost, resulting in low collection rates in rural areas (Schwartz and Pratt, 1990).

The two used lead prices, the price of scrap battery lead, p_S and recycled lead, p_R, should be closely related. Suppose they differ only by a constant marginal re-refining cost: $p_R = p_S + \alpha$. The demand equation (1) can then be written in terms of p_R, with the recovery rate $\hat{r}(p_S)$ redefined as a $r(p_R)$. In a steady-state, the demand by households which recycle is equal to the supply of recycled lead:[15]

$$r(p_R)D(q - \delta(p_R - \alpha)) = R(p_R)$$

Thus, the market-clearing condition (2) reduces to

$$(1 - r(p_R))D(q) = V(p_V). \tag{3}$$

This equation, representing the residual demand for virgin lead and its supply, is the basis for the analysis of price changes with policy interventions that follows.

2.2 The simple analytics of recycling policies

Virgin-materials tax An *ad valorem* virgin-materials tax of t, raises the price of lead to its users, $q = p_V(1 + t)$. Because primary and secondary lead are perfect substitutes in some uses, users will purchase recycled lead at the same price, $q = p_R$. This implies $p_R = p_V(1 + t)$. The market-clearing condition with the tax is

$$(1 - r(p_V T))D(p_V T) = V(p_V) \tag{4}$$

where $T = 1 + t$.

To find an approximate expression for the price change associated with a given tax, taking

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[15] This identity requires that no lead is lost in reprocessing. In fact, reprocessing losses amount to only 3-5% of re-refined lead, so this equation holds approximately.

derivatives yields

$$\frac{dD}{dq}(1 - r(p_V T))(T dp_V + p_V dT) + \frac{dr}{dp_R} D(p_V T)(T dp_V + p_V dT) = \frac{dV}{dp_V} dp_V.$$

Substituting a negative demand elasticity, ϵ_D, and positive primary and secondary supply elasticities, ϵ_V and ϵ_R, respectively, the expression for the change in p_V is:[16]

$$\frac{dp_V}{p_V} = \frac{(\epsilon_R \frac{r}{1-r} - \epsilon_D)}{\epsilon_D - \epsilon_R \frac{r}{1-r} - \epsilon_V} \frac{dT}{T}. \tag{5}$$

The producer price of virgin lead, p_V, decreases, but the price of recycled lead, p_R, and the price to purchasers, q, increase.

The tax imposes costs on consumers and producers that exceed the revenue it collects. These losses constitute private costs that weigh against the social benefits provided by a reduction in lead disposal. Households who do not recycle and primary lead producers bear burdens from the tax, but households who recycle bear no costs, if they do not discount the scrap price of batteries ($\delta = 1$).[17] When the tax revenue generated is subtracted from the households' costs the marginal private cost is

$$dPC^{tax} = -p_V t dV, \tag{6}$$

(yielding positive costs because dV is negative).

Deposit/refund A deposit/refund is analytically identical to a virgin-materials tax in this framework. With a virgin-materials tax, all consumers pay a higher price for lead, but those who recycle batteries recover the tax increment in the form of higher prices for their used batteries. Similarly, a deposit/refund is returned to those who recycle, so it taxes only lead

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[16] This expression is related to the familiar partial-equilibrium incidence of a tax $dp = \frac{\epsilon_D}{\epsilon_S - \epsilon_D} \frac{p}{T} dT$. Demand in the usual expression is re-defined as residual demand for virgin lead — the left-hand side of equation (4) — which has an elasticity of $\epsilon_D - \epsilon_R \frac{r}{1-r}$.

[17] In this framework, the secondary industry receives revenue just equal to its constant marginal reprocessing cost α and has no surplus to gain or lose.

use in batteries that are not recycled.[18]

Let level of the deposit/refund equal f, which is normalized by p_V for algebraic simplicity. The deposit raises the price of battery lead to users, so $q = p_V + p_V f$. However, the effective price of scrap to households that recycle remains the same: $q - p_S = \alpha$. These two equations imply $p_R = p_V(1+f)$. The market clears when

$$[1 - r(p_V(1+f))]D(p_V(1+f)) = V(p_V), \tag{7}$$

which is identical to equation (4), the market-clearing condition for the virgin-materials tax. The implied private costs are also the same.

Subsidy to recycled lead A recycling subsidy could take the form of a refund to consumers that is not funded by a deposit or a credit to firms per pound of lead they recover. More indirect subsidies may also be used; for example, many states provide investment tax credits to firms that produce recycled goods (OTA, 1989). Such subsidies may be effective in reducing the marginal cost of recycled lead, but alter secondary firm behavior in other ways. This analysis concentrates on the simplest form, which creates the incentives desired without additional distortions.

Let the subsidy equal a ($A = 1 + a$) which is normalized by p_V. Recycled lead now receives an implicit subsidy: the price to users is $q = p_R - ap_V$. But $q = p_V$ continues to hold, so $p_R = p_V(1+a)$. The market now clears when

$$1 - r(p_V A)]D(p_V) = V(p_V). \tag{8}$$

Taking derivatives yields an equation comparable to equation (5):

$$\frac{dp_V}{p_V} = \frac{\epsilon_R \frac{r}{1-r}}{\epsilon_D - \epsilon_R \frac{r}{1-r} - \epsilon_V} \frac{dA}{A}. \tag{9}$$

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[18] The equivalence between these two programs will not hold, however, if there are other uses of lead in addition to batteries. In this case, the deposit/refund will dominate the virgin-materials tax as a means of reducing disposal of lead in batteries. This difference is illustrated in table 10 in section 4.

Not only does p_V decline, but $q = p_V$ so the user price of refined lead declines with it. Thus, the price of lead to its users declines with a refund, in contrast to a virgin-materials tax or deposit/refund.

As a result, the subsidy also differs from the tax and deposit/approach in terms of who bears its costs. Unlike the tax and deposit in which households that do not recycle lose, these households gain from a subsidy because the cost of lead to them declines. Recycling households also gain because their scrap is more valuable. On the other hand, the subsidy imposes burdens on virgin material producers and dissipates government revenue. The net marginal private cost is analogous to equation (6):

$$dPC^{subs} = ap_V dR.$$

To compare a subsidy to the cost of the tax-deposit approach, the marginal private cost per unit reduction in virgin lead is more useful:

$$dPC^{subs} = -ap_V \frac{1}{\epsilon_V}[\frac{1}{\epsilon_R}(\epsilon_R - \epsilon_D)(\epsilon_D - \epsilon_V) + \epsilon_D \frac{r}{1-r}]dV. \tag{10}$$

Recycled content standard A recycled content standard stipulates a ratio r^* of recycled lead to total lead in batteries. This ratio can be set as a standard that individual firms must meet. Alternatively, trading between firms can be allowed; 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. The permit system accomplishes a given r^* for minimum cost, if the permit market is competitive (Dinan, 1990). A marketable permit system is analyzed here because this requires less information about the substitution opportunities for different battery producers than a firm-by-firm standard.

Suppose that a permit entitles its holder to use one unit of virgin lead. This permit must be traded for enough units of of recycled lead that the aggregate r^* for the industry continues to conform to the standard. This tradeoff holds if a permit can be created by the use of an additional $\frac{r^*}{1-r^*}$ units recycled lead. [19] Policy-makers choose the level of r^* and it generates an equilibrium permit price π (where π is normalized by the price of primary lead).

With this system, users of lead perceive the price of virgin lead to be equal to its price plus the price of the permit sacrificed to use it, $q = p_V + \pi p_V$. On the other hand, recycled lead costs p_R but creates $(\frac{r^*}{1-r^*})^{-1}$ permits with value πp_V, so its net cost is only $q = p_R - (\frac{1}{r^*} - 1)\pi p_V$. The price of recycled lead in terms of p_V is:

$$p_R = p_V(1 + \pi + (\frac{1}{r^*} - 1)\pi) = p_V(1 + \frac{\pi}{r^*}).$$

Thus, the recycled content standard can be thought of as a revenue-neutral combination of a virgin-materials tax and a subsidy to recycled lead.[20]

Now, the lead market will clear when

$$[1 - r(p_V(1 + \frac{\pi}{r^*}))]D(p_V(1 + \pi)) = V(p_V), \tag{11}$$

and the permit market will clear when

$$r^* = r(p_V(1 + \frac{\pi}{r^*})).$$

The change in lead price can be found for a dr^* and the associated $d\pi$:

$$\frac{dp_V}{p_V} = \frac{1}{\epsilon_D - \epsilon_R\frac{r^*}{1-r^*} - \epsilon_V}[(\frac{\epsilon_R\frac{1}{1-r^*}}{1 + \frac{\pi}{r^*}} - \frac{\epsilon_D}{1 + \pi})d\pi - \frac{\epsilon_R\pi}{1 + \frac{\pi}{r^*}}\frac{dr^*}{r^{*2}}]. \tag{12}$$

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[19] This tradeoff is the number of units x of recycled lead necessary to maintain r^*:

$$r^* = \frac{R}{R + V} = \frac{R + x}{V + 1 + R + x}.$$

[20] Anderson et al (1989) suggest a policy they refer to as "recycling credits" for waste oil, in which a subsidy to recyclers is funded by a tax on the primary industry. If this policy is to be revenue-neutral, the ratio of the subsidy to tax must be $\frac{1-r}{r}$ at the final recovery rate. Thus, this "recycling credits" program is equivalent to the recycled content standard with permit trading; all the results for a recycled content standard apply to either program.

The expression is simpler if there is assumed to be no standard in place at the outset, so the initial $\pi = 0$ and $r^* = r$:[21]

$$\frac{dp_V}{p_V} = \frac{\epsilon_R \frac{1}{1-r} - \epsilon_D}{\epsilon_D - \epsilon_R \frac{r}{1-r} - \epsilon_V} d\pi. \tag{13}$$

The producer price of virgin lead declines as with all the programs, but the effect on q, the price of lead to users, is ambiguous.[22] It is unclear whether a recycled content standard discourages lead use, like the tax and deposit, or encourages it like a subsidy.

The private costs of a recycled content standard are:

$$dPC^{RCS} = \pi p_V(\frac{1-r}{r} dR - dV).$$

This expression is clearer if it is translated into a a revenue-neutral combination of tax and subsidy

$$dPC^{RCS} = -tp_V dV + ap_V dR,$$

where the tax rate $t = \pi$ and the effective subsidy is $a = (\frac{1}{r} - 1)\pi$. Again, the expression can be rewritten in terms of the change in virgin lead use:

$$dPC^{RCS} = \pi p_V(\frac{\epsilon_R(\epsilon_D(1-r) - \epsilon_V) + \epsilon_D(1-r)(\epsilon_D - \epsilon_R \frac{r}{1-r} - ev)}{\epsilon_V r(\frac{\epsilon_R}{1-r} - \epsilon_D)} - 1)dV. \tag{14}$$

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[21] Equation (13) is phrased in terms of a change in the equilibrium price of permits because this makes the analogy with tax and subsidy clearest, but in fact policy operates through a choice of dr^* which in turn determines $d\pi$ and dp_V. The change in price in terms of dr^* is:

$$\frac{dp_V}{p_V} = \frac{\frac{1}{1-r^*} - \frac{\epsilon_D}{\epsilon_R}}{\epsilon_D(1-r^*) - \epsilon_V} dr^*.$$

[22] The change in price to users is:

$$\frac{dq}{q} = \frac{\epsilon_R - \epsilon_V}{\epsilon_D - \epsilon_R \frac{r}{1-r} - \epsilon_V} d\pi$$

which is positive iff $\epsilon_R < \epsilon_V$.

2.3 Comparison of the policies

Table 3 summarizes the results of the foregoing analyses of the recycling policies. The four policies can be compared when they are implemented to achieve the same reduction in lead disposal. This section first compares the level of the policy intervention (that is, how large the tax or subsidy must be) and then uses this information to compare the private costs of the various policies.

For this comparison, the reduction in lead disposal must be expressed in terms of the variables used in this analysis. The amount of lead disposed is the difference between the stock of scrap and the amount recovered. If demand and supply of lead do not shift over time, then once the policy intervention has been in place for some time the available scrap will equal the amount demanded in the current period. In this steady state, disposal is $(V(p_V)+R(p_R))-R(p_R)=V(p_V)$, so policies can target the amount of virgin lead produced.[23] Although the popular view is that the goal of these recycling policies is to raise recovery rates, the amount of virgin lead is a better target, because environmental costs may be reduced not only by recycling ("pollution abatement") but also by "pollution prevention" through discouraging consumption of the pollution-generating good.

Intervention levels Thus, policies should be compared at level necessary to achieve given dV, which requires equal dp_V under all policies. For a virgin-materials tax (or deposit/refund) versus a recycling subsidy, equating (5) and (9) yields

$$\frac{da}{dt}=1-\frac{\epsilon_D(1-r)}{\epsilon_R r}>1 \tag{15}$$

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[23] More generally, disposal equals $V_{t-4}+R_{t-4}-R_t$, so V_t is at best an approximate target when the market is not in steady state. In practice, the market for battery lead may not be static for several reasons, the most significant being growth in demand for automobiles, electric vehicles, and back-up power for computers. Further, the introduction of a recycling policy itself will induce dynamics in the market. With a tax or deposit, more battery scrap is available at the outset of the program than a few years later, because these programs reduce demand for lead. It makes less sense to phrase the problem as achieving a disposal reduction target when the market is not in steady-state; the level of disposal as well as the level of the tax or subsidy should be allowed to vary with time.

Table 3: Summary of results for recycling policies

	Virgin materials tax and deposit/refund	Recycling subsidy	Recycled content standard
Decrease in virgin lead price ($\frac{dp_V}{p_V}$)	$(\epsilon_R \frac{r}{1-r} - \epsilon_D)\gamma \frac{dt}{1+t}$	$\epsilon_R \frac{r}{1-r} \gamma \frac{da}{1+a}$	$(\epsilon_R \frac{1}{1-r} - \epsilon_D)\gamma d\pi$
Change in consumer price (dq)	> 0	< 0	$>$ or < 0
Marginal private cost	$-tp_V dV$	$ap_V dR$	$\pi p_V(\frac{1-r}{r} dR - dV)$

Note: The derivation of these expressions is described in the text. The symbols are:

Policy interventions — t is the level of the virgin materials tax, a is the subsidy level, and π is the price of permit to use one unit of recycled lead under a recycled content standard with trading.

Quantities — V is the amount of virgin lead, and R the amount recycled lead. The recovery rate is $r = \frac{R}{V+R}$.

Elasticities — the supply elasticity of recovered battery lead is ϵ_R, and for virgin lead, ϵ_V, and the demand elasticity for lead in batteries is ϵ_D.

$$\gamma = \frac{1}{\epsilon_D - \epsilon_R \frac{r}{1-r} - \epsilon_V}$$

The expression for $\frac{dp_V}{p_V}$ for the recycled content standard assumes no initial intervention in the market; the more general case is shown as equation (13) in the text.

when there is no intervention initially. Expression (15) implies a higher subsidy a than tax t for the same level of disposal. This result has a simple explanation: a subsidy to recycled lead does not discourage lead use like a virgin-materials tax or deposit/refund, so a stronger intervention is necessary to accomplish the same reduction in disposal.

On the other hand, the price of a permit is less than the level of the tax for changes in the neighborhood of the no-intervention equilibrium. Comparing the price of permits with the tax yields

$$\frac{d\pi}{dt} = \frac{\epsilon_R \frac{r}{1-r} - \epsilon_D}{\epsilon_R \frac{1}{1-r} - \epsilon_D} < 1. \tag{16}$$

The price of the permit does not capture the complete effect of the program: π is the rate of the effective tax on virgin lead use, but there is also an effective subsidy to recycled lead. Because of this additional subsidy, a smaller π than t is necessary to achieve a small reduction in V.

Private costs Comparing the private costs of the four policies at the same dV provides an assessment of their cost-effectiveness in reducing environmental damages.[24] Using the expressions for private cost in terms of dV in equations (6), (10), and (14), the policies can be ranked for a given dV:

$$dPC^{subs} > dPC^{RCS} > dPC^{tax}.$$

That is, the tax-deposit provides the least costly means of reducing lead disposal, and a subsidy to recycled goods the most costly. A recycled content standard with trading of permits is intermediate in costs.

The virgin-materials tax and deposit/refund are not just the most cost-effective of these four policies but the best of all possible policies. Two arguments support this claim. First, it is argued above that accomplishing a reduction in disposal is equivalent to accomplishing a

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[24] In the presence of multiple environmental externalities, it is not necessarily true that more recycling will decrease the total external costs (Baumol, 1977). In this analysis, I assume disposal cost are the only environmental costs that have not been internalized so the policies can be compared in terms of their private costs alone. However, Stavins (1991) expresses doubts about whether battery recycling does result in a net reduction in environmental damages.

reduction in virgin lead use. But, the least costly means of reducing virgin lead use is to tax the good itself — not subsidize its substitutes as the subsidy and recycled content standard do. Second, with a deposit/refund and virgin-materials tax, only lead that is disposed (and thus generates externalities) is taxed. Although they are not waste-end taxes, these policies effectively tax disposal and thus act as Pigouvian taxes.[25]

Although the policies considered here differ from the usual instruments of environmental policy, the results are consistent with the literature on environmental policy design (e.g. Baumol and Oates, 1988; Bohm and Russell, 1985). One apparent inconsistency is the inefficiency of the tradeable permit system. 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 that would be no more costly than the virgin-materials tax. In the steady-state, such a system would simply require a permit per unit of primary lead used, with the number of permits set by the government in order to achieve a given reduction in disposal, rather than allowing permits to be created by use of recycled lead.

By contrast, the subsidy considered here is the most efficient form of subsidy. A recycling subsidy provides the correct incentives for pollution abatement, when the pollution in question is waste disposal. The problem arises with the use of subsidies in general, rather than the particular design of the subsidy considered here. In the presence of recycling, subsidizing pollution abatement also encourages consumption of the offending good. Subsidies are not a preferred policy for regulating disposal when recycling is an option.[26]

3 Empirical analysis of the market for lead

To estimate the empirical effects of these policies and the magnitude of their costs requires information about the lead supply and demand elasticities (ϵ_R, ϵ_V, and ϵ_D). Following a discus-

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[25] Menell (1990) makes this argument for the deposit/refund.

[26] Studies of environmental policy design have argued that taxes should be preferred to subsidies, although the marginal incentives are the same, because the subsidy may adversely affect firms' entry and exit decisions (see Baumol and Oates, 1988, ch. 8). In this analysis, taxes are preferred to subsidies even if the number of firms in the industry not responsive to the programs. Taxes dominate subsidies more generally because of the effect of subsidies on supply of the good.

sion of previous empirical work, this section presents estimates of the relevant elasticities. In the following section, these estimates are applied to the incidence and private cost expressions above to simulate the consequences of the three policies.

Previous empirical work on supply and demand relationships for recyclable materials falls into two general categories. First, a number of authors construct econometric models of various world metals markets to project the effects of supply disruptions. Wise (1979) builds this kind of model for the lead industry. However, he focuses on projecting lead supply and price and does not make an effort to estimate structural relationships. Thus, many of his regression do not include price variables or do not correspond to meaningful supply functions. A study of world copper markets by Fisher, Cootner and Bailey (1972) provides elasticities from another non-ferrous metal market for comparison.

Second, a few empirical studies concentrate on recycling activity *per se*. The largest group of studies examines paper markets (for example, Deadman and Turner, 1981; Edgren and Moreland, 1989; Edwards and Pearce, 1978; Lahiri and Kinkley, 1984). All of these studies ignore the substantial change in the composition of waste paper during the time studied which leads to error in their price variables. Some of these studies also fail to take into account the endogeneity of paper prices. Perhaps as a result of these econometric problems, most of these studies do not find significant price effects on recovery. The authors are skeptical about the potential of price-based policies to alter paper recycling behavior, and this skepticism has occasionally cast doubt on recycling programs in general.

A small number of other studies examine public policies for metal recycling. Anderson and Speigelman (1976) estimate supply and demand equations for lead in order to study the impact of the tax code on secondary material use. The equations they estimate are the most similar to those estimated here, but they use short time series (1949 to 1967 or 1972) that precedes the opening of the major Missouri mines and the recent declines in recycling rate.[27] Further, their demand elasticities fail to take into account the effect of scrap prices on lead

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[27] In addition, the wartime price controls on lead were not lifted until 1951, making their use of observations prior to 1952 suspect. Further, they use zinc price as an instrument for the price of lead, but zinc is frequently produced as a co-product with lead, so its price is unlikely to be exogenous.

consumption. Studies of other markets with recycling include related work by Anderson and Speigelman (1977) on steel, Slade (1980) on copper, and estimates of metal recovery from municipal waste by Bingham, et al (1983) (aluminum and ferrous metals, based on simulated data from engineering models). In addition, the extensive literature on Alcoa's aluminum monopoly contains estimates of secondary aluminum supply elasticity in the early twentieth century (e.g. Suslow, 1986); however, recycling behavior (of households at least) seems likely to have changed substantially over this time.

3.1 Recovery of battery lead

This section describes my estimates of lead recovery elasticities. Supply of recycled lead depends on the stock of available scrap and on the price of refined lead. The basic supply equation is estimated in the form:

$$\log r_t = \beta_1 + \beta_2 \log(p_t) + \ldots \text{other terms} \ldots + \epsilon_t. \tag{17}$$

where r_t is the recovery rate and p_t is the price of refined lead.[28] Secondary lead constitutes about half of all lead production, so the price of lead in this equation is endogenous and instruments are used for this price.

Variables The dependent variable for these regressions is the recovery rate — the amount of lead recovered from motor vehicle batteries divided by the amount of scrap battery lead (shown in Figure 1). The numerator of this variable is the total lead recovered from batteries, from which an estimate of lead recovered from industrial and traction batteries has been subtracted. The denominator is based on the sum of replacement batteries shipments and

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[28] A maintained assumption in these regressions is that the number of defunct batteries, the denominator of the regression is not itself sensitive to the price of lead. This assumption will be violated if households replace batteries before the batteries cease to function and the extent of this precautionary behavior depends on battery prices. There is no direct evidence about when households replace batteries. However, running the instrumental variables regression in table 4 with the log of estimated scrap battery stock as the dependent variable does not yield significant coefficients on price when the number of cars in use is included as a right-hand-side variable. The coefficient and its standard error are .15 (.14) and .05 (.14) depending upon whether a trend is included as well.

motor vehicle de-registration in that year, because every defunct battery must either have been replaced or the car taken out of service. This value was multiplied by the estimated lead content of these batteries: the fourth and fifth lags of the ratio of the amount of lead used for auto batteries to total U.S. battery production in a given year.[29]

Instruments Instruments for the price of lead were chosen from policy variables that shift demand for lead. First, the EPA phase-out of lead additives in gasoline beginning in 1975 had a substantial impact on lead use (see Table 1). This program had two distinct regimes: from 1975 to late 1982, the standards for maximum lead content applied to all gasoline; but after November, 1982, the standards applied to leaded gasoline only and EPA permitted trading of rights to meet this standard. Two instruments represent the level of these standards, a different variable for each regime. Second, demand reflected the level of military ordnance purchases, because ammunition was a major use of lead. The number of active duty military personnel is used as instrument to reflect exogenous changes in the government's demand for ammunition. In addition to these policy variables, the equations also include lagged price and recovery rate variables.

Results Table 4 shows the estimation results. The first five equations use the price of refined lead, because this price variable is appropriate for the analysis in Section 2. All the equations are estimated with an AR(1) error structure because this structure could not be rejected in preliminary regressions. The basic regression in column (1) yields an point estimate for the recovery rate elasticity of .25 that is statistically significant. Based on this regression, a value of .3 is used for this elasticity in the policy simulations because it lies in the middle of the range of estimated values. For comparison, Anderson and Speigelman (1976) find supply elasticity for all secondary lead of 0.48 for the period 1954-1972. In a different non-ferrous metals market, copper, Fisher, Cootner, and Bailey, find a short-run supply elasticity of secondary

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[29] The average lifetime of batteries varied from 35 to 40 months during this period, rising until the early 1970s and declining since then (Salkind et al, 1984). Lags of 4 and 5 are chosen, because in addition to its useful life, a battery may sit on a shelf for over a year before it is sold (Consumer Reports, 1987) and for a few months before it is reprocessed (Abt Associates, 1985).

copper of 0.42–0.44 and a long-run elasticity of 0.31–0.33.

The equations in columns (3) and (4) explore possible dynamics for the supply equation. In both cases, negative coefficients on the lagged endogenous variables are observed. Such a negative relationship might be expected if the effect of high price periods is to draw down inventories of used batteries, but this effect does not appear to be strong in magnitude nor are the coefficients statistically significant. The last column illustrates the results when the price variable is the price of used batteries paid by dealers.[30] A lower elasticity estimate is suggested by this equation.

3.2 Supply of refined virgin lead

This section estimates the elasticity of primary metal supply, the other supply elasticity necessary for policy comparisons. Lead is mined in large underground chambers with thousand-foot shafts. Capacity can only be expanded slowly, and maintenance requirements make it difficult to shut down operations temporarily. For this reason, the supply of primary lead is modeled using a partial-adjustment framework. The desired supply in a given year is a function of the price of refined lead.

The dependent variable is the quantity of refined primary lead produced annually, a stage of production comparable to the recovered lead variable used above. The price is the U.S. producer price of refined lead as in the previous set of equations.

Instruments The instruments used for price in recycled lead supply could be used here, because they reflect shifts in the demand for refined lead. However, two of the instruments — the EPA's limits on lead in gasoline — are factors that were known in advance, whereas the third instrument describes demand shifts that were not entirely predictable at the time. It seems likely that the speed of adjustment to the two types of changes would differ. Suppose

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[30] The series, gathered from daily price reports in *American Metals Market*, has a gap between 1963 and 1974. A projected price based on refined lead price is used for these years

Table 4: Instrumental variables estimates for secondary battery lead supply

	Dependent variable: Log(Recovery rate)				
	(1)	(2)	(3)	(4)	(5)
Intercept	-.59	-.41	-.75	-.63	-.41
	(.68)	(.67)	(.20)	(.21)	(.07)
Log(Price of refined lead)	.25	.15	.52	.27	–
	(.12)	(.14)	(.13)	(.20)	
Log(Price of refined lead$_{t-1}$)	–	–	-.17	–	–
			(.13)		
Log(Price of used batteries)	–	–	–	–	.077
					(.054)
Log(Recovery rate$_{t-1}$)	–	–	–	-.056	–
				(.150)	
Trend	–	-.0010	–	–	–
		(.0013)			
Autocorrelation coefficient	.43	.34	.45	.47	.41
	(.15)	(.16)	(.16)	(.15)	(.16)
Implied supply elasticity	.25	.15	.44	.26	.08
	(.12)	(.14)	(.71)	(.73)	(.05)

Years: 1954-88.
AR(1) error structure for all equations.
Instruments for price: limits on lead content of gasoline; number of active duty military personnel; first lag of log(price) and log(recovery rate). Equations (3) and (4) also have first lags of all pre-determined variables.

supply is modeled as:

$$S_t - S_{t-1} = \theta_1(S_t^* - S_{t-1}) + \theta_2(E_{t-1}(S_t^*) - S_{t-1}) + \epsilon$$

where θ_1 reflects responsiveness to actual desired supply, and θ_2 reflects changes that are planned a period in advance. Let desired supply be defined as

$$S_t^* = Z_t\beta$$

with innovations to Z_t:

$$\nu_t = Z_t - E_{t-1}(Z_t).$$

Then,

$$S_t - S_{t-1} = (\theta_1 + \theta_2)(Z_t\beta - S_{t-1}) - \theta_2\nu_t\beta + \epsilon.$$

However, ν_t is unobserved and will bias the estimates of the coefficient on Z_t. The problem can be avoided with instruments that are uncorrelated with ν_t, which by construction will be true of variables used to forecast Z_t. Therefore, only variables whose values are known in advance are used; so the leaded gasoline phase-out instruments are included, but active duty military personnel is dropped. In addition, from 1959 through 1966, a quota was imposed on the imports of lead; a dummy for these years is included as an instrument.

Results Table 5 shows the results of these regressions, which appear to be sensitive to the specification. Long-run elasticity estimates range from .37 to .8. The basic regression in column (2) suggests a low supply elasticity but this increases when a trend is added in column (3). The estimate of .8 from column (3) is used in the regressions. For comparison, Anderson and Speigelman (1976) estimated a short-run supply elasticity of lead of .55 with a long-run elasticity of of 1.0, for 1949-67. Wise (1979) found the supply elasticity of .92, but this estimate excludes Missouri mines which have provided 90% of primary lead since the 1970s.

One concern in interpreting these equations is structural change in primary lead supply as a result of the switch in the late 1960s and early 1970s from small dispersed mines to the massive operations in the New Lead Belt. In column (4), this change is represented by a dummy for the period following 1969 the year of the first significant production from the New Lead Belt in Missouri, which has a significant positive effect on supply. A Chow test does not reject (at 5%) the restriction that the coefficients on price and lagged supply were stable across this period.

3.3 Demand for battery lead

As in section 2, the demand equation estimated here relies on an effective price of battery lead that depends on whether the battery is expected to be recycled or not. Total demand (in equation (1)) thus has two components:

$$TD(q, p_S) = rD_1(q - \delta p_S) + (1 - r)D_2(q). \tag{18}$$

where δ is a discount factor and p_S is the price of scrap battery lead. The appropriate scrap price for this equation is the expected price at the time when batteries wear out. These forecasts are assumed to equal current prices.

The equation estimated assumes that demand has a constant elasticity form and that the demand behavior of recyclers and non-recyclers is the same, but the two groups experience different prices.[31] That is, both $D_1(p)$ and $D_2(p)$ are assumed to have the form:

$$\log D_t = \beta_1 + \beta_2 \log p_t + Z_t\gamma + e_t. \tag{19}$$

where Z_t is a matrix of other variables that may determine demand and e_t is a disturbance term. β_2 is the demand elasticity, ϵ_D, that needs to be estimated to calibrate the policy

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[31] It is possible that the demand behavior of recyclers and others may differ systematically; for example, those who do not recycle may be wealthier households with higher time costs and less elastic demand.

Table 5: Instrumental variables estimates for primary lead supply

	Dependent variable: Log(Refined primary lead) (1)	(2)	(3)	(4)
Intercept	13.06	2.87	3.73	6.19
	(.05)	(.95)	(1.31)	(2.17)
Log(Price of refined lead)	.053	.083	.25	.38
	(.098)	(.070)	(.15)	(.10)
Log(Refined primary lead_{t-1})	–	.78	.70	.37
		(.07)	(.11)	(.19)
Trend	–	–	.0048	–
			(.0039)	
Opening of Missouri mines	–	–	–	.26
(Dummy beginning 1969)				(.09)
Durbin-Watson	.46	–	–	–
LM test for AR(1)	–	.78	.16	.68
Significance level		(.37)	(.69)	(.40)
Implied supply elasticity	.053	.38	.84	.60
	(.98)	(.28)	(.34)	(.32)

Years: 1952–1988.
Instruments: limits on lead content of gasoline; quota on lead imports.
Serial-correlation robust standard errors in parentheses.

comparisons in section 2. Combining equations (18) and (19) yields

$$\log D_t = \beta_1 + \beta_2 \log(r_t(q_t - \delta p_t^s)^{\beta_2} + (1 - r_t)q_t^{\beta_2}) + Z_t\gamma + e_t. \tag{20}$$

The parameters of this equation are estimated using a nonlinear instrumental variables procedure.

The dependent variable employed in the first three equations is lead per auto battery rather than total battery lead. This variable is chosen because households are unlikely to alter measurably the number of batteries they purchase, but changes in the price of lead may be reflected in the lead content of batteries. In the last two columns, this restriction is released and total battery lead used as the dependent variable.

The composition of demand for lead has changed dramatically in over the time period covered over the years since the 1950s, as first section indicated. The parameters of demand for lead in batteries should be stable, however, despite these structural shifts. Instruments for lead price were chosen from policy variables that affected demand for lead in other uses. The same instrument set was used in the recovery rate equations in table 4: limits on the lead content of gasoline, and the number of active duty military personnel, reflecting demand for ammunition. In addition, the lagged price and lagged dependent variable are used as instruments.

Results The estimates in table 6 suggest a low demand elasticity for the lead content of batteries. In the first column, demand has a simple log-linear form that does not account for the effect of the scrap price on the effective price of lead for households that recycle. Including the price of whole used batteries in the next columns barely changes the results. The scrap price is only a small fraction of the price of refined lead (in 1988, \$.02/pound relative to \$.37/pound for refined lead, although early in the period the ratio is somewhat higher) so its limited effect is not surprising. Column (3) includes a time trend to account for technological changes; however, this equation does not produce a negative point estimate for the elasticity.

In the final two columns, the dependent variable is total lead used for batteries, rather than

Table 6: Demand for lead in motor vehicle batteries

	Dependent variable:				
	Lead content per motor vehicle battery			Total lead in batteries	
	$\delta = 0$	$\delta = 1$	$\delta = 1$	$\delta = 1$	$\delta = 1$
	(1)	(2)	(3)	(4)	(5)
Intercept (β_1)	3.95	3.94	3.97	4.87	3.65
	(.04)	(.05)	(.05)	(.21)	(.93)
Demand elasticity (β_2)	-.14	-.14	.082	-.12	-.14
	(.09)	(.11)	(.157)	(.06)	(.09)
Trend	–	–	.0029	–	-.067
			(.0013)		(.031)
Cars in use	–	–	–	6.1	.35
				(.7)	(.13)
Autocorrelation coefficient	.53	.58	.51	.54	.22
	(.20)	(.20)	(.18)	(.17)	(.17)

Years: 1954-1988
The basic equation is:

$$\log D_t = \beta_1 + \log(r_t(q_t - \delta p_t^s)^{\beta_2} + (1 - r_t)q_t^{\beta_2}) + Z_t\gamma + e_t,$$

which was estimated by nonlinear least squares, with an AR(1) error structure.
Instruments for price were: limits on lead content of gasoline; number of active duty military personnel; first lag of price and of dependent variable.

the lead per battery. This specification allows for the possibility that households may adjust the frequency with which they buy batteries in response to the lead price. The number of cars in use is included as a critical determinant of total demand for battery lead. The equations suggest estimates for the demand elasticity similar to those from the earlier equations.

Previous studies have also found low demand elasticities for lead. Moroney and Trapani (1981) provide estimates of the Allen elasticities of substitution for inputs into storage battery manufacture, including lead. From their results, derived demand for lead appears to be somewhat more elastic than these estimates, $\epsilon_D = -0.20$. Anderson and Speigelman (1976) estimate a demand elasticity for all lead (not just lead in batteries) of $\epsilon_D = -0.21$ for 1949–72. Further, inelastic demand for lead in storage batteries is consistent with the lack of attention in battery engineering papers such as Salkind et al (1984) to the lead content of batteries, although battery weight is an important concern.

4 Empirical effects of battery recycling policies

The elasticities estimated in Tables 4, 5 and 6 can then be used compare the three alternate recycling policies. Table 7 provides a summary of the features of the three policies. It uses elasticities $\epsilon_R = 0.3$, $\epsilon_V = 0.8$, and $\epsilon_D = -0.1$, drawn from the estimated equations in section 3. The equilibrium before policy intervention is characterized by 1988 price ($.3714 per pound of refined lead) and recovery rate ($\rho = 0.65$).[32]

Several conclusions can be drawn from the results in table 7. First, the table indicates that the tax and subsidy rates necessary to achieve moderate reductions in lead disposal are consistent with those under consideration by policy-makers. For instance, the $5 per battery deposit/refund recently adopted by several states should result about a 20% reduction in battery lead disposal. On the other hand, a virgin-materials tax would have to be much higher than the current Superfund feedstock tax on primary lead-oxides (only .2 cents per

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[32] The model assumes a steady-state, so the recovery rate and the ratio of recycled to total lead are the same. For the simulations, the actual recovery rate is taken to represent the ratio of recycled lead and is applied to the 1988 total lead quantity to find V and R.

Table 7: Estimated effects of recycling policies

	Decrease in lead disposal		
	10% reduction	20% reduction	50% reduction
Virgin-materials tax and deposit/refund			
Tax level (per pound of lead)	\$0.10	\$0.20	\$0.51
Deposit per battery	\$2.20	\$4.40	\$9.39
Revenue (million 1988 dollars)	54	98	151
Private costs (million 1988 dollars)	6.0	24	151
Recycling subsidy			
Subsidy level (per pound of lead)	\$0.12	\$0.24	\$0.60
Revenue cost (million 1988 dollars)	150	320	990
Private costs (million 1988 dollars)	12	49	309
Recycled content standard			
Price of permits(per pound of lead)	\$0.07	\$0.14	\$0.35
Level of standard (r^*)	.69	.73	.84
Private costs (million 1988 dollars)	6.6	27	166

Simulations are based on $\epsilon_R = .3, \epsilon_V = .8$, and $\epsilon_D = -.1$ and 1988 prices and quantities. The necessary tax, subsidy, and permit prices for a given level of disposal reduction are calculated from the price changes in equations (5), (9) and (13), respectively. The other results follow from these rates or the private cost equations in section 2.

pound) to accomplish substantial disposal reduction.

Second, table 7 illustrates the magnitudes of the policy differences in section 2. For a reduction in disposal of 20%, a virgin-materials tax or deposit/refund would need to be $0.20 per pound of lead, whereas to achieve the same results, a direct subsidy would have to be $0.24 per pound. The price of permits can be even lower — $.14 for a permit to use a pound of primary lead — because they combine a tax and a subsidy. The variation in private costs is broad; they range from $24 million for the tax-deposit approach to $49 million for recycling subsidies. The recycled content standard does not appear to be much more costly than a tax or deposit program; for a 20% reduction, it costs $27 million, only 8% more than the tax-deposit approach.

Tables 8 and 9 contain sensitivity analyses for the results in table 7. Table 8 varies the supply elasticities. Note that the costs of the programs *decline* with increases in the elasticities, in contrast to the ordinary result for tax burdens. Higher elasticities make it possible to reduce lead disposal by any amount with a more modest policy intervention and therefore impose lower costs for this reduction. The percentage difference in cost between the programs also declines with the elasticities, because the tax or subsidy rates decline and costs are related to the square of these rates.

In table 9, varying the demand elasticity significantly affects only the subsidy program. It becomes more expensive as the elasticity grows because the perverse incentives to increase consumption of lead have more impact. The difference in costs between the tax-deposit approach and a recycled content standard also rises slightly for the same reason.

The values in tables 7–9 represent an optimistic view of the outcome of the four policies. Actual policies must be designed correctly to achieve costs as low as those in the tables. First, a recycled content standard may be implemented without allowing trading of rights, like the current legislation under consideration in Congress. For this proposal, the private costs in table 7 represent a lower bound; true costs will probably be higher, reflecting differential possibilities for firms to substitute recycled for virgin lead.

Second, proposals for deposit/refunds usually rely on a single deposit for all batteries,

Table 8: Recycling policies with different supply elasticities

	Primary lead elasticity		
	$\epsilon_V = 0.4$	$\epsilon_V = 0.8$	$\epsilon_V = 1.2$
$\epsilon_R = 0.15$			
Virgin-materials tax and deposit/refund			
Tax level (dollars/pound)	.377	.285	.250
Private costs (million 1988 dollars)	45.9	33.9	30.2
Recycling subsidy			
Subsidy level (dollars/pound)	.509	.384	.340
Private costs (million 1988 dollars)	155	96.7	80.1
Recycled content standard			
Permit price (dollars/pound)	.272	.205	.183
Private cost (million 1988 dollars)	53.6	38.8	34.1
$\epsilon_R = 0.3$		Benchmark case	
Virgin-materials tax and deposit/refund			
Tax level (dollars/pound)	.296	.203	.172
Private costs (million 1988 dollars)	35.2	24.2	20.5
Recycling subsidy			
Subsidy level (dollars/pound)	.347	.238	.202
Private costs (million 1988 dollars)	88.8	49.4	38.6
Recycled content standard			
Permit price (dollars/pound)	.20	.140	.119
Private cost (million 1988 dollars)	40.3	26.6	22.2
$\epsilon_R = 0.45$			
Virgin-materials tax and deposit/refund			
Tax level (dollars/pound)	.263	.170	.139
Private costs (million 1988 dollars)	31.3	20.3	16.6
Recycling subsidy			
Subsidy level (dollars/pound)	.293	.190	.155
Private costs (million 1988 dollars)	70.2	36.5	27.4
Recycled content standard			
Permit price (dollars/pound)	.179	.116	.095
Private cost (million 1988 dollars)	35.2	21.9	17.7

Note: All results are for a 20% reduction in lead disposal. The central panel ($\epsilon_R = .3$, $\epsilon_R = .8$) is the benchmark level displayed in Table 7.

Table 9: Recycling policies with different demand elasticities

	Lead demand elasticity		
		Benchmark	
	$\epsilon_D = 0$	$\epsilon_D = -0.1$	$\epsilon_D = -0.2$
Virgin-materials tax and deposit/refund			
Tax level (dollars/pound)	.222	.203	.189
Private costs (million 1988 dollars)	26.5	24.2	22.5
Recycling subsidy			
Subsidy level (dollars/pound)	.222	.238	.254
Private costs (million 1988 dollars)	26.5	49.3	77.7
Recycled content standard			
Standard (r^*)	.726	.728	.731
Permit price (dollars/pound)	.146	.140	.136
Private cost (million 1988 dollars)	26.5	26.6	26.8

Note: Supply elasticities are those in table 7, as is the demand elasticity in the center column. All results in the table are for a 20% reduction in lead disposal.

rather than one that varies with the lead content of the battery. This traditional approach removes the deposit's incentive for substitution towards lower lead content in batteries. But the refund continues to have a marginal effect on battery recovery. As a result, this type of deposit/refund program will create similar marginal incentives to a recycling subsidy. The two programs differ primarily in that the deposit/refund continues to discourage the purchase of batteries, but auto battery demand is likely to be highly inelastic. The private costs of the program will therefore be similar to those for the subsidy.

Third, the full menu of policies is not available to states that want to reduce battery disposal on their own. A local virgin-materials tax or recycled content standard would be unlikely to have a substantial impact on the national price of primary lead and any increased recovery would not be concentrated in that state. The deposit/refund or direct subsidy offer mechanisms for increasing recycling within the state directly and may be the only sensible policies for states acting unilaterally.[33]

Finally, the analysis has assumed that all lead is used in batteries. Although batteries comprise a large and rising percentage of lead demand, there are also many other uses of lead, ranging from construction to medical uses, which make up 22% of total U.S. lead consumption. These other uses can be made explicit with an additional demand curve in the market-clearing equation,

$$D_b(q, p_S) + D_o(q) = V(p_V) + R(p_R),$$

where the *o* subscript refers to other uses. In the presence of alternative uses for lead, a number of features of the policy ranking change (results analogous to those in section 2 are outlined in an appendix). Table 10 illustrates the results of adding another sector that uses lead. The demand elasticities shown for this other sector should be compared to Wise's (1979) estimate of -.33 for lead demand in uses other than batteries, ammunition, and gasoline additives in

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[33] Another restriction on the results in table 7 is that they are based on the steady-state lead reductions for each policy intervention. In the short term for policies that increase the consumer price of lead, disposal falls by less than this amount because initially high lead stocks provide a large supply of cheap recycled lead. When these short-term effects on lead supply are taken into account, the subsidy and recycled content standard will look more favorable relative to taxes or deposits than they appear in these tables.

Table 10: The effect of a non-battery sector on recycling policies

	One sector benchmark	Two sector results Demand elasticity in other sector: $\epsilon_o = 0$	 $\epsilon_o = -.3$	 $\epsilon_o = -.6$
Deposit/refund				
Tax level (dollars/pound)	.203	.166	.159	.153
Private costs (million 1988 dollars)	24.2	19.8	18.9	18.3
Virgin-materials tax				
Tax level (dollars/pound)	.202	.166	.183	.199
Private costs (million 1988 dollars)	24.2	19.8	28.1	37.4
Recycling subsidy				
Subsidy level (dollars/pound)	.238	.195	.187	.180
Private costs (million 1988 dollars)	49.3	36.7	34.4	32.7
Recycled content standard (for batteries only)				
Standard (recovery rate)	.728	.737	.737	.736
Permit price (dollars/pound)	.140	.115	.110	.106
Private cost (million 1988 dollars)	26.6	21.4	20.4	19.6
Recycled content standard (for all uses of lead)				
Standard (recovery rate)	–	.597	.600	.603
Permit price (dollars/pound)	–	.096	.099	.102
Private cost (million 1988 dollars)	–	22.8	24.9	26.7

All results are for a 20% reduction in battery lead disposal.
The total recycled and disposed battery lead is the same for in all columns, but the amount of lead used for other goods in 1988 is added in the last three columns. For the recycled content standard on battery lead alone, it is assumed that all recycled lead is used in batteries from the outset. These values are based on expression for price changes and private cost shown in the appendix.

the United States. [34]

The most important change is that a deposit/refund now dominates a virgin-materials tax.[35] The virgin-materials tax creates production inefficiencies because it taxes not only the disposal of batteries (like the deposit/refund) but also the use of lead in the other sector. In table 10, the virgin-materials tax results in private costs of $28 million for a 20% reduction in battery lead disposal, while the deposit/refund accomplishes this reduction for only $19 million. In addition, a subsidy can be preferred to a virgin-materials tax, if amount of lead in alternative uses and the elasticity of demand in these uses were large enough. In the table, the tax dominates the subsidy for low elasticities in the other sector, but not for higher values.

Under this new regime, an ambiguity arises about the design of a recycled content standard. If the standard applies only to batteries, with perfect substitution of secondary for primary lead, there should just be a shift of any recycled lead used by other industries into battery manufacture. Thus, if much recycled lead is used in the manufacture of goods other than batteries, the standard should apply to all goods manufactured with lead to assure that it does increase total demand for recycled lead. However, if almost all recycled lead is used in batteries at the outset, a standard applied to batteries alone will be sufficient and preferable because it results in a lower private cost. Both alternatives are presented in the table.

As before, these standards are equivalent to revenue-neutral combinations of a tax and a subsidy. The standard for batteries alone is equivalent to a deposit/refund in which the revenue raised is redistributed as a recycling subsidy, while the standard on all lead is like a virgin-materials tax and recycling subsidy. Therefore, the standard on batteries is less costly than a straight subsidy, but the standard on all lead, like the virgin-materials tax, may be more or less costly than a subsidy. Table 10 suggests that even the most costly recycled content standard remains better than the subsidy for the estimated parameters.

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[34] An additional sector alters the costs of the policy even if demand in this sector exhibits zero elasticity, as a result of the assumption of constant elasticities. When the amount of lead in the market increases, a larger price change is necessary to achieve the same absolute reduction in lead disposal.

[35] The results in the table 10 and the text assume that the goal remains a reduction in battery lead disposal only, not disposal of lead in alternative uses. This target could be appropriate if lead in these other uses is disposed only in the very distant future, as for example lead used in construction, or if environmental damages from disposal of lead in these uses is already regulated, for example through industrial hazardous waste policy.

5 Conclusion

This paper considers recycling policies that can reduce environmental costs from waste disposal when more direct restrictions are too difficult to enforce. Successful recycling policies address disposal at two levels, encouraging recovery of lead and discouraging its consumption. The virgin-materials tax and deposit/refund are the best policies because they operate on both margins. By contrast, a subsidy for recycling is the worst policy because it fails on the second margin, decreasing the price of lead to users and thus encouraging lead consumption.

The policy analysis was applied to programs aimed at the recovery of lead from automobile batteries, using supply and demand parameters estimated from the U.S. lead market. This empirical analysis suggests two conclusions. First, price-based recycling policies can effectively increase lead recycling, a conclusion that contrasts with earlier studies of recycling policy. This difference from previous work results in part from the moderate price sensitivity of lead recovery that is observed. In addition, these simulations explicitly consider the complete market for lead, so the effects of the policies on virgin lead production and overall demand are taken into account, in contrast to earlier studies. Second, this empirical analysis reveals substantial differences between the policies in the costs of accomplishing the same disposal reduction. A recycling subsidy may entail roughly twice the private costs of the most efficient tax-deposit approach.

The policy rankings in this paper are relevant for a number of environmental policy issues other than those raised by automobile batteries. Households dispose of several kinds of hazardous wastes besides auto batteries; automobile engine oil and other household batteries contribute substantially to the toxicity of municipal solid waste and are likely targets of state and federal recycling programs in the next few years. Chlorofluorocarbons in household refrigrators and air conditioners would be good candidates for these policies because prohibitions against releasing them into the air pose particularly thorny enforcement problems. Further, the analysis applies to recyclable industrial hazardous wastes, like organic solvents, for which deposit/refunds and subsidies have been suggested.[36]

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[36] For example, by Russell (1988) and Baumol and Mills (1985).

A number of empirical dimensions of recycling remain to be explored for auto batteries and other applications of these policies. In particular, a better understanding of households' waste disposal and recycling behavior will clarify issues that have been raised in this discussion. For example, this analysis has assumed that disposal levies will be ineffective because they are difficult to monitor, but in practice they may be successful in reducing disposal. Further research on the extent of illegal disposal, and its sensitivity to disposal prices and the threat of sanctions would be important for choosing policies. In addition, any distributional analysis of recycling programs will require better evidence about the characteristics of households that dispose or recycle.

A Policy comparisons with an additional sector

Suppose there are other uses for lead is addition to batteries, but lead is not recyclable in these other uses.[37] The market-clearing condition in terms of residual demand for virgin lead (analgous to expression (3) in the text) is now

$$(1 - r(p_R))D_b(q_b) + D_o(q_o) = V(p_V),$$

where the o subscript refers to this other sector.

A.1 Simple analytics

Virgin materials tax A virgin materials tax raises the consumer price in both sectors, as well as the price of recycled lead: $q_b = q_o = p_R = p_V(1+t)$. If the demand elasticity for lead in other uses is ϵ_O, then the incidence of the virgin materials tax is:

$$\frac{dp_V}{p_V} = \frac{(\epsilon_R R - \epsilon_D(1-r)D_b(q) - \epsilon_O D_o)}{\epsilon_D(1-r)D_b(q) + \epsilon_O D_o - \epsilon_R R - \epsilon_V V}\frac{dT}{T}. \tag{21}$$

where $(1-r)D_b(q)$ is the demand for battery lead from households that do not recycle.[38] The private cost of the virgin materials tax is the same as in section 2:

$$dPC^{tax} = -tp_V dV. \tag{22}$$

Deposit/refund A deposit/refund now differs from a virgin materials tax, because it only raises the consumer price of lead used in batteries: $q_b = p_V(1+f)$ but $q_o = p_V$, where f is the level of the deposit/refund normalized by p_V. ($F = 1 + f$). As before, the effective price to recycling households does not change: $q - p_S = \alpha$, which implies $p_V(1+f) = p_R$. The change

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[37] Further uses in which the lead is recyclable will not alter the nature of the results in this section. Because reprocessing costs are assumed to be constant, the amount recycled from other goods will not affect the cost of recycling batteries.

[38] In $D_b(q)$, the q is explicit because this is demand at the effective price of lead for households that do not recycle, rather than $q - p_S$, the effective price for those who do.

in price is now

$$\frac{dp_V}{p_V} = \frac{\epsilon_R R - \epsilon_D(1-r)D_b(q)}{\epsilon_D(1-r)D_b(q) + \epsilon_O D_o - \epsilon_R R - \epsilon_V V}\frac{dF}{F}. \tag{23}$$

The private cost from the deposit/refund is now smaller than the virgin-materials tax at the same intervention level:

$$dPC^{D/R} = -p_V f d((1-r)D_b(q)). \tag{24}$$

Subsidy The subsidy remains very similar to its one-sector form: $q_b = q_o = p_V$ and $p_V(1+a) = p_R$. The resulting price change is

$$\frac{dp_V}{p_V} = \frac{\epsilon_R R}{\epsilon_D(1-r)D_b(q) + \epsilon_O D_o - \epsilon_R R - \epsilon_V V}\frac{dA}{A}, \tag{25}$$

with private cost

$$dPC^{subs} = ap_V dR. \tag{26}$$

Recycled content standard (RCS) As mentioned in the text an ambiguity now arises with the recycled content standard. In what follows RCS_1 (with permit price π_1) is a standard that applies only to batteries, creating the same implicit tax as a deposit/refund. For this first RCS policy, assume that all the recovered lead was used in battery manufacture before the policy intervention. The price change is

$$\frac{dp_V}{p_V} = \frac{\epsilon_R \frac{R}{r} - \epsilon_D(1-r)D_b(q)}{\epsilon_D(1-r)D_b(q) + \epsilon_O D_o - \epsilon_R R - \epsilon_V V}\frac{d\pi_2}{\pi_2}, \tag{27}$$

with private cost

$$dPC^{RCS_1} = p_V \pi_1((\frac{1}{r} - 1)dR - d(1-r)D_b(q)). \tag{28}$$

RCS_2 (with associated permit price π_2) refers to a recycled content standard that applies to all lead; it creates an implicit tax in both demand sectors. Now,

$$\frac{dp_V}{p_V} = \frac{\epsilon_R \frac{R}{\rho} - \epsilon_D(1-r)D_b(q) - \epsilon_O D_o}{\epsilon_D(1-r)D_b(q) + \epsilon_O D_o - \epsilon_R R - \epsilon_V V}\frac{d\pi_2}{\pi_2}. \tag{29}$$

where ρ is the fraction of all lead that is recycled (as opposed to r, the fraction of batteries). It has private cost

$$dPC^{RCS_2} = p_V \pi_2((\frac{1}{\rho} - 1)dR - dV). \tag{30}$$

A.2 Policy comparisons

When the objective is to reduce battery lead disposal (that is, $(1-r)D_b(q)$) but not disposal of lead in other uses, the policy comparsions in section 2.4 of the paper are altered as follows. The level of intervention rankings are now

$$\pi_1 < f,\ f < t, \text{ and } f < a,$$

where f is the deposit/refund, t is the tax, a is the subsidy, and π_1 is the recycled content standard that applies only to lead. The resulting private cost ranking is:

$$dPC^{D/R} < dPC^{RCS_1} < dPC^{subs}; \tag{31}$$

$$dPC^{D/R} < dPC^{tax}; \tag{32}$$

$$dPC^{D/R} < dPC^{RCS_2}. \tag{33}$$

(31) holds for the simple reason behind the results in the one sector model: a subsidy encourages consumption of lead. The RCS on batteries only is essentially a revenue-neutral combination of a deposit/refund and a subsidy, so it lies between the two. In (32), the deposit/refund is preferable to the virgin materials tax because it does not distort the non-battery sector. For the same reason, the RCS on all lead (which is a revenue-neutral combination of a virgin materials tax and subsidy) is more costly than the deposit/refund. Interestingly, it is not possible to sign the difference between the virgin materials tax and and RCS_2: the subsidy may reduce distortions in the other sector.[39]

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[39] If the policy objective is to reduce lead disposal in all uses, the virgin-materials tax and deposit/refund should be switched in all of the comparison as should the two recycled content standards. The virgin-materials

A Data sources

Batteries: data on the number of replacement and original equipment batteries, exports and imports, were taken variously from Putnam, Hayes and Bartlett (1987), Franklin Associates (1989) and the *Statistical Abstract of the United States.* Number of original equipment batteries previous to 1960 was inferred from motor vehicle sales during this time.

Lead quantities: data for the amount of lead used for batteries recovered from batteries, mine production and refined primary lead production are from U.S. Bureau of Mines *Minerals Yearbook*, and Metallgeschellschaft, A.G. *Metall Statistik.* Lead for storage batteries in industrial uses is reported by Franklin Associates and *Minerals Yearbook* for recent years, but previous to 1967, it is assumed to be 10% of total lead used for batteries.

Motor vehicles: motor vehicle de-registrations, cars in use, and motor vehicle sales are from Motor Vehicle Manufactures Association, *Motor Vehicle Facts and Figures.*

Prices: refined lead prices are the annual average U.S. producer prices for soft lead reported in *Metall Statistik.* The price series for scrap batteries is the price per pound of whole, drained used batteries paid by dealers in New York. It was gathered from daily price reports in *American Metals Market.*

Other data: active duty military personnel is from *Statistical Abstract of the United States.* Information on phase-out of lead in gasoline is from Anderson, et al (1989).

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tax now address the correct margins, while the deposit/refund is too limited in scope.

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