Metals

Closing the Loop on Cadmium

An Assessment of the Material Cycle of Cadmium in the U.S.

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Abstract

Goal, Scope and Background. In this study, the major flows of cadmium in the U.S. economy are quantified and the primary sinks are identified to gauge the need for additional policy to minimize the potential human health and ecosystem risks associated with these flows. Because of the concurrent occurrence of cadmium and zinc in ore, we also consider the relevant portions of the material cycle of zinc.

Methods. We estimated the flows of cadmium through U.S. manufacturing using a mass balance approach with data provided by the U.S. Geological Survey's Minerals Yearbook. Cadmium emissions factors were created using facility specific information found in the U.S. Toxics Release Inventory and were used to model future losses. Data gaps were filled through review of relevant literature. We modeled the import and sales of nickel-cadmium batteries with rechargeable battery usage trends and estimates of market share by battery chemistry.

Results and Conclusion. Primary cadmium in the U.S. is almost exclusively produced as a co-product of zinc. Almost all zinc and cadmium mined in the U.S. is exported to foreign smelters as ore concentrate. We estimate that the bulk of cadmium consumed in the U.S. economy (~90%) is imported in the form of nickel-cadmium rechargeable batteries. These batteries can be divided into the larger wet-cells and portable rechargeable batteries (PRB). The collection rate for the recycling of large wet cells was found to be high (80%) while the collection rate for PRBs is low (5–20%). The Rechargeable Battery Recycling Corporation (RBRC) is responsible for the collection of these batteries which are recycled exclusively by the International Materials Reclamation Company (INMETCO). The remaining PRBs are generally disposed of in municipal solid waste (MSW) landfills. This study provides a detailed substance flow analysis of U.S. stocks and flows of cadmium in products, however additional research is needed to better quantify the associated exposures and risks.

Recommendation and Perspective. Based on our analysis, we make four recommendations. First we suggest that if cadmium is to be used, it should be used in long-lived products that can be easily collected and recycled with minimal losses. Second, continued cadmium use should be coupled with renewed efforts on the part of policy-makers to encourage the collection and recycling of cadmium-bearing products. At present, consumers do not see the environmental cost associated with the proper disposal of the cadmium content of NiCd batteries. Policy options for improving recycling rates include collecting deposits and providing rewards for the return of spent batteries, taxing or otherwise discouraging discarding PRBs in municipal solid waste, and providing incentives for extended producer responsibility. Third, we highlight the importance of the connection between zinc mining and the supply of cadmium in designing an effective policy to manage the risks associated with cadmium. Fourth, we recommend that policy measures be taken to provide the necessary data required to improve our understanding of the flow of cadmium into the U.S. in the form of product imports and the amount of cadmium lost or disposed of by recycling processes.

Keywords: Battery recycling; cadmium; material cycle of cadmium in the U.S.; material flow analysis (MFA); nickel-cadmium batteries; substance flow analysis (SFA)

1 Background, Aim and Scope

1.1 Regulatory context

As federal environmental regulation has increased over the past thirty years, cadmium, along with other heavy metals, has received attention from policy-makers due to increasing concern about its environmental and health effects (Smith 1995). Exposure to elevated levels of cadmium is believed to cause lung, kidney and liver malfunction as well as prostate cancer (ATSDR 2004, Ikeda 2000, US EPA 1995). In the most extreme cases, kidney malfunction due to cadmium intake has led to bone decalcification as was the case with the outbreak of Itai-Itai Disease in Japan's industrial Jinzu River Basin.[1] This outbreak resulted in Japan's first water quality and industrial discharge control legislation (Ui 1992).

In the U.S., industrial facilities have been required under the Emergency Planning and Community Right to Know Act (EPCRA) to report their releases of cadmium since 1987. In 1998 the reporting threshold for cadmium was lowered due to its classification as persistent, bioaccumulative and toxic (PBT). The Clean Air Act of 1990 listed cadmium under its Title III hazardous air pollutants subject to maximum achievable control technology (US EPA 1993). More recently cad-

[1] Chaney (2004) and Reeves (2004) show that human absorption and retention of cadmium in the Jinzu River region may have been increased by deficiencies in zinc, iron and calcium caused by a subsistence rice diet.

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mium has been identified as one of the 31 Priority Chemicals for action under the U.S. Environmental Protection Agency's (EPA) National Partnership for Environmental Priorities. Together with lead and mercury, it is one of only three metals selected for this designation (US EPA 2005). Reference human exposure levels for cadmium promulgated by the EPA, Occupational Safety and Health Administration (OSHA) and the Agency for Toxic Substances and Disease Registry (ATSDR) are on the order of 10^{-4} mg/kg/day for food and water (ATSDR 2004, US EPA 1992) and an eight hour time weighted average of 5 µg/m^3 for occupational exposure (OSHA 1992).

Regulatory efforts to control the release of cadmium contained in post consumer products have been more modest. The Federal Trade Commission has issued guidelines for the proper labeling of recyclable batteries containing cadmium and other toxic materials. In 1996 congress passed the Mercury Containing and Rechargeable Battery Act (MCRBA) facilitating the recycling of nickel-cadmium NiCd and other rechargeable batteries by standardizing the labeling, collection and disposal requirements previously enforced by state agencies (US EPA 1997). Under the Universal Waste Rule (UWR), 40 *CFR* 273 (1995), the MCRBA also streamlines the process of returning of NiCd batteries to a recycling facility by relaxing their hazardous waste designation during shipment to a recycling facility.

In the European Union, officials have issued the Waste Electrical and Electronic Equipment (WEEE) Directive and Restriction on the use of Hazardous Substances (RoHS) in an effort to address concerns about toxic substances in products. These directives restrict the import and manufacture of products containing a variety of dangerous substances including cadmium, lead, hexavalent chromium, and mercury above threshold values (2002/96/EC, 2002/95/EC). Prior to these directives, policies to control the release of heavy metals in the EU focused on limiting their presence in industrial discharges and requiring facilities to use the *best technical means available* for the prevention of cadmium discharges (83/513/EC).

1.2 Analytical context

Mapping the life cycle material flow of cadmium in an economy allows us to identify opportunities for reducing environmental releases and other 'losses' of this useful, but toxic material. Traditionally, policies intended to reduce exposure to a toxic material focus on a specific exposure route such as industrial emissions. Quantifying the flows of a material through its life cycle can assist in revealing new means of reducing undesirable flows and provide insight into dependencies in the system. van der Voet et al. (2000) have developed models for the flows of heavy metals, including cadmium, through the economy and environment in the Netherlands and found that while environmental regulation has been effective in reducing their emissions from industry over the past few decades, heavy metals are accumulating in the Dutch economy as products and infrastructure. Unless collection patterns change, much of these stocks will enter landfills where their heavy metal contents pose a risk of contaminating leachate, requiring potentially costly management. Their analysis also revealed an important 'hidden' flow of cadmium as a trace element in fertilizers and sludges applied to agricultural soil. Several studies have shown that low level cadmium emissions from a variety of sources contributed to the buildup of cadmium in the Rhine River Basin over the period 1955 to the mid 1990s; these sources included phosphate fertilizers, sewage sludge, corrosion of galvanized zinc, tire wear, fossil fuel combustion, nonferrous smelting and various manufacturing processes (Stigliani et al. 1993, 1994a, 1994b, Anderberg 2000, Klepper 1995). Stigliani points out the role soil plays in buffering ecosystems against cadmium uptake and notes that long-term accumulation of cadmium in soils leads to increasing transport and plant uptake.

Today cadmium is used primarily in the manufacture of nickel cadmium (NiCd) rechargeable batteries. Lankey (1998, 2000) assessed the life cycle emissions, energy use and waste generation associated with the use and recycling of rechargeable NiCd batteries and found that the use of recycled materials reduces the energy requirements and environmental emissions associated with the manufacture of rechargeable batteries. She also found that a NiCd battery recycling facility requires a modest payment to make the process profitable. In light of these and other considerations, McMichael and Hendrickson (1998) suggest replacing cadmium and other toxic materials in batteries with efficient, cost-effective alternatives as an ideal while in the meantime regulations are needed to reduce the toxic effects of used portable rechargeable batteries (PRB). Rydh (1999, 2002a) has also performed a material flow and life cycle inventory of NiCd battery recycling in Sweden and concludes that the optimal recycling rate considering all of the environmental impacts is near 100%. In another study, Rydh (2002b) considered the global metal flows arising from PRB use and found that the mass of cadmium extracted for PRB manufacture is roughly four times that of natural cadmium flows (due to weathering and volcanic activity).

In this paper we examine the flow of cadmium through the U.S. economy and comment on policy options for reducing environmental emissions and the cadmium contents of municipal solid waste (MSW). Mine concentrates, refined cadmium, products and releases from manufacturing facilities are the primary focus of this analysis. Special attention is given to zinc ore concentrates as they are the only ore for which beneficiation of cadmium metal is currently economical. We included order of magnitude estimates of the flow of cadmium in fertilizers and fossil fuels to allow for comparison, however these flows occur independently from those that comprise the focus of our work and are not part of our dynamic model.

1.3 Cadmium use in the U.S.

Environmental regulation has played a role in the decline of the U.S. cadmium industry over the past decade (Plachy 2005). Higher costs associated with emissions control and waste management put economic strain on domestic industries. However, this increased strain has had the effect of

promoting the search for alternatives to some of the applications for cadmium (US EPA 2000). In 2004, 90% of the zinc mining in the U.S. was done in Alaska. The U.S. exported almost all of the zinc concentrates mined that year and then imported nearly as much refined zinc[2]. On average from 1993 to 1999, the U.S. exported 70% of its mine production of zinc in concentrates and then imported the same percentage of the refined zinc consumed (Plachy 2004).

The International Metals Reclamation Company (INMETCO) located in Ellwood City, Pennsylvania is currently the only facility engaged in the recovery of cadmium in the U.S. Recycling is available for NiCd batteries and electric arc furnace dust resulting from recycling of galvanized steels and alloys. The U.S. Geological Survey (Plachy 2000) estimates that about 15% of all post-consumer cadmium is recycled. Most of this material comes from industrial NiCd batteries which contain 20% of all cadmium used in batteries and are collected at a rate of 80% (Plachy 2003). The remaining cadmium comes from PRBs and electric arc furnace (EAF) dust which are recycled at lower rates.

In order to take extended responsibility for their products, manufacturers of NiCd batteries have contributed to formation of the Rechargeable Battery Recycling Corporation (RBRC) which coordinates efforts to collect NiCd PRBs in the U.S. and Canada. Over 30,000 drop off locations in retail stores and public institutions accept batteries which are shipped in boxes provided by RBRC to INMETCO for recycling.

INMETCO's cadmium recycling process is capable of recycling 3,000 tonnes of NiCd batteries (NRC 2005) or 500 tonnes cadmium metal[3]. Although this is enough cadmium to meet the U.S. manufacturing demand of 500 tonnes (USGS 2005a), additional cadmium continues to be extracted along with zinc ore. The increasing use of galvanized steel (Gordon 2003) continues to outstrip the supply of recycled zinc requiring continued extraction of ore. Recovery rates for zinc are low because most zinc is dispersed in products containing low concentrations making it difficult to achieve high recovery rates. In addition to batteries and EAF dust, recovery of cadmium from MSW incineration has been proposed by Brunner (2004).

Seventy eight percent of the cadmium used in U.S. manufacturing is used for nickel cadmium batteries while manufacture of pigments (12%), coatings and platings (8%), plastics (1.5%) and nonferrous alloys and other uses (0.5%) consume the remaining material. Over the past four years, consumption of cadmium by U.S. manufacturers has declined by 70% in response to environmental concerns (Plachy 2005). As consumption by U.S. manufacturing continues to decrease, imports of NiCd batteries are an increasingly important source of cadmium in the U.S. An important emerging use for cadmium is thin film photovoltaic (PV) cells which utilize the photoelectric properties of CdS or CdTe to capture a higher percentage of incident solar energy than traditional silicon PV cells. Currently the use of cadmium for PV technologies is very small (less than 0.5%).

1.4 Trace flows of cadmium

Significant flows of cadmium also occur due to its presence as a trace element in phosphates, fossil fuels and zinc compounds. The most important of these flows is the cadmium contained in phosphates, estimated to be between 250 and 3,200 tonnes. Most of this material is applied to agricultural soils as phosphate fertilizer. The cadmium content of coal is between 100 and 1,700 tonnes. Three tonnes of this cadmium become air emissions while the majority is concentrated in flu dusts and other combustion by products which are disposed of (70%), used in the manufacture of concrete (10%), used as a fill material (8%), or used for a variety of other purposes. The cadmium in the petroleum consumed in the U.S. is found to be roughly 2 to 200 tonnes. These flows will be discussed in more detail in the following section.

1.5 Phosphates

Ross (1994) demonstrates the buildup of cadmium in agricultural soils associated with the application of phosphate fertilizers. Cadmium, present in the phosphate ore, remains throughout the fertilizer manufacturing processes and is deposited along with the fertilizer. The exact flow of cadmium in fertilizer is difficult to estimate in the U.S. because of the large range of cadmium concentrations present in different phosphate ore. Fthenakis (2004) estimates cadmium content of phosphate to range between 0.25 to 80 g cadmium per tonne phosphate while the GCA Corporation (1981) provided a slightly higher range of 3 to 100 g per tonne. According to Davister (1996), 60 g per tonne phosphate is the average cadmium content in European fertilizers. Zethner (2000) places the European average slightly higher at 140 g per tonne phosphate. Phosphates mined in Florida were found to contain 23 g/tonne and those mined in North Carolina contain 166 g/tonne (Davister 1996). U.S. phosphate consumption in 2004 was between 11[4] (Jasinski 2005) and 19 million tonnes (US DOC 2005). Assuming a cadmium concentration between 23 g/tonne corresponding to phosphates mined in Florida and 166 g/tonne associated with phosphates from North Carolina (Davister 1996), 250 to 3,200 tonnes of cadmium were applied to agricultural soils in 2004.

1.6 Fossil fuels

According to the U.S. Energy Information Administration U.S. coal consumption totaled roughly one billion tonnes in 2004 (EIA 2005). Estimates of the cadmium content of coal found in the literature include 0.1 (Pacyna 2001), 0.1 to 3 (Swaine 1995), 0.22 (Coles 1978), 0.5 (Andersson 2000),

[2] The zinc content of the zinc concentrates mined in the U.S. in 2004 were 770 tonnes. An additional 220 tonnes of zinc concentrates were imported and 850 tonnes of concentrates were exported. 785 tonnes of refined zinc were imported.

[3] Assuming NiCd batteries contain 16% Cd by mass (Rydh 2002). The Enquette Commission (1994) estimated the Cd in NiCd batteries as *up to 19%*. Gaines (1995) put the Cd content of a vehicle battery slightly higher at 24%.

[4] An estimate of the Cd content of marketable phosphate can be obtained by dividing the phosphate content of U.S. marketable phosphate rock production, 10.4 million tonnes, by its gross weight 35.8 million tonnes. Applying the calculated phosphate content of 29% to the apparent phosphate rock consumption of 39 million tonnes suggests a content of 11 tonnes.

0.5 to 0.7 (Klein 1975). 0.5 to 1 (Stigliani 1993), 1 (Azar 1996), 1.7 (Pavageau 2002) g/tonne coal. Assuming a concentration of 0.5 g/tonne, the cadmium content of coal consumed in the U.S. is roughly 500 tonnes. Applying the lowest and highest estimates yield a range of 100 to 1,700 tonnes. The EPA provides an emission factor for airborne releases of cadmium from coal combustion of 0.0026 g cadmium/tonne coal[5]. Although this emission factor is not valid for all coal combustion activities, it allows us to calculate a lower bound for coal combustion emissions at 2.6 tonnes. The remaining is contained in the 97 million tonnes of combustion by-products generated annually or trapped in emissions control device. Combustion by-products are comprised of fly ash (59%), bottom ash (16%), boiler slag (3%), and flue gas desulphurization material (23%). Seventy percenty of this material is disposed of while 10% is used in concrete, 8% for structural fill and road base material and the remainder for other uses including wallboard and snow/ice control. (Kalyoncu 1999)

In their analysis of cadmium in Swedish soils, Karlsson et al (2004) place the cadmium content of oil between 0.002 and 0.2 g cadmium per tonne oil and find the cadmium content of natural gas to be negligible. If we apply their estimate to the total U.S. oil consumption in 2003 of 20.5 million barrels per day (EIA 2004), assuming a density of 860 kg/m^3, this amounts to 8.5 billion tonnes of oil per year containing 2 to 200 tonnes of cadmium.

1.7 Zinc

Because the process by which zinc is separated from its ore concentrates cannot completely remove the cadmium, it remains as a trace element in products that contain zinc. The American Society for Testing and Materials' (ASTM) standards for the cadmium content of zinc range from 0.002 percent for special high grade to 0.2 percent for prime western grade (IZA 2001). If we assume the ASTM standard for high grade of 0.02 percent as a median we can calculate that the 1.5 million tonnes of zinc consumed in the U.S. in 2003 (USGS 2005b) contained approximately 300 tonnes of cadmium. Fifty five percent of zinc is used for galvanizing (USGS 2005b) implying 165 tonnes of cadmium content while 17% is used in zinc-based alloys, 13% in brass and bronze, and the remainder for other uses. Cadmium contained in galvanized steel or zinc alloys can be released or recovered from electric arc furnace (EAF) dusts when these metals are recycled. Councell (2005) discusses zinc liberated from tires while driving. Zinc is used in the vulcanization of rubber and may contain trace amounts of cadmium. This material accounts for the PM_{10} containing cadmium in the range of 0.1 and 10 µg/vehicle-km detected by Lough (2005). Stigliani (1993) attributes 2 to 3% of the cadmium emissions in Germany's Rhine River Basin between the mid-1960s and 1988 to the combustion of tires by cement manufacturers.

[5] Based on emissions data for 11 facilities firing bituminous coal, 15 facilities firing subbituminous coal, and 2 facilities firing lignite utilizing either venture scrubbers, spray dryer absorbers, or wet limestone scrubbers with an electrostatic precipitator or fabric filter.

2 Development of the Model

2.1 Cadmium content of products

We estimated the amount of cadmium used in the manufacture of cadmium containing products by multiplying the mass of cadmium used annually by U.S. manufacturers (M_{man}) by the estimate of the percentage consumed (f_{cons}) in each of the manufacturing processes involving cadmium provided by the International Cadmium Association (USGS 1994–2003, USBOM 1972–1993). Five categories representing the major manufacturing processes were identified: nickel-cadmium batteries, plastic stabilizers, pigments, coatings and platings, and nonferrous alloys and other uses. A factor for the amount of material 'lost' in processing (f_{loss}) was defined to calculate the cadmium content of products. Published estimates of generalized cadmium losses from U.S. manufacturing processes range from 0.3% (Plachy 2003) to 3% (Llewellyn 1994). We estimate the cadmium in imported products (f_{import}) to be 20% of the amount of cadmium in products manufactured domestically with the exception of NiCd batteries which are discussed in the next section. The cadmium content of products sold in the U.S. (M_{Cd}) is given by Eq. (1):

$$M_{Cd,i} = M_{man} f_{cons,i} (1\text{-}f_{loss}) \left[1 + \frac{f_{import,i}}{(1 - f_{import})} \right] \quad (1)$$

Where each 'i' corresponds to one of the five major cadmium product categories: pigments, coatings and platings, plastics and nonferrous alloys and other uses.

2.2 Cadmium contents of portable rechargeable batteries sold in the U.S.

Estimating the amount of cadmium contained in PRBs sold in the U.S. is complicated due to the lack of data pertaining to the flow of imports. The most complete estimate of the cadmium content of batteries sold in the U.S. was reported by the New York State Department of Environmental Conservation in their *Report on Dry Cell Batteries in New York State* (1992). Following the trend of the PRB sales estimate provided in their report suggests that U.S. sales NiCd PRBs in 2000 would be 560 million batteries. Assuming a battery mass of 0.25 kg, 16% of which is cadmium by weight (Rydh 2002), this estimate is significantly higher than the U.S. Geological Survey (USGS) estimate of 11,300 tonnes of cadmium in batteries sold in the U.S. in 2000 (Plachy 2003). According to Plachy, 82% of the cadmium content of batteries sold into U.S. markets comes from imported batteries. We chose a more modest linear growth rate of 17 million units per year to project U.S. battery sales ($N_{NiCd\ PRB\ Sales}$) in the years following the NYDEC report (beginning in 1993). The cadmium content of batteries ($M_{Cd,batteries}$) is given by:

$$M_{Cd,batteries} = N_{NiCd\ PRB\ Sales}\ f_{market\ share,j}\ M_{NiCd,j} \quad (2a)$$

$$N_{NiCd\ PRB\ Sales}(t) = [17\ t - 33{,}900] \times 10^6 \quad (2b)$$

Where t is the calendar year.

The market share by battery application ($f_{market\ share,i}$) as well as the mean mass of the PRBs by application ($M_{NiCd,,j}$) were estimated using values from the literature (Rydh 2002, Fujimoto 1998, Hake 1998, Lankey 1998).

2.3 Cadmium releases

Our estimate of the emissions of cadmium from the processes of mining, smelting, and manufacturing was developed using data from the U.S. Toxics Release Inventory (US EPA 2004). The U.S. Toxics Release Inventory (TRI) contains data for over 14,000 facilities required to report their releases of 667 toxic materials. We aggregated the releases into eight release categories corresponding to zinc and lead mining, primary zinc smelting, INMETCO and the five manufacturing sectors previously described. The U.S. EPA classifies TRI releases using the four digit Standard Industrial Classification (SIC) System. We created a mapping system between our categories and the SIC system so that TRI values could be assigned to each of our eight product categories. Functional units were chosen based on data availability.

2.4 End of life cadmium products

Lifetimes were estimated for each product and battery application and used to determine when products would enter the waste stream. Values used as model inputs are as follows, dissipative uses: plastics 3 yrs, nonferrous alloys and other 6 yrs, coatings and platings 7 yrs, pigments 8 yrs; batteries by type: cell phones 3 yrs, cordless tools 6 yrs, industrial wet-cell batteries 17 yrs, and emergency lighting, portable computers, electronics and others 5 yrs. We assume that cadmium recycling only occurs for NiCd batteries. A 15% collection rate was used for PRBs and 80% for wet cell NiCd batteries (Plachy 2003). Exports of cadmium waste are excluded from this study despite concern over the inappropriate handling of toxic material in developing countries (Basel Action Network 2001). The export flow is assumed to be small when compared with the flow to MSW landfills in the U.S.

2.5 Total emissions and emissions factors

The total U.S. cadmium releases to air, surface water and other compartments (including land treatment, underground injection, POTW transfers and transfers off-site) as compiled in the 2002 TRI are shown in **Fig. 1**. Zinc mining and smelting contribute the most to industrial cadmium releases in the U.S. The large amount of surface water discharges by the plastics industry is surprising given the small amount of cadmium reportedly consumed as plastics stabilizers. This becomes apparent in **Fig. 2** where we find that plastic stabilizers have the largest emissions factors for both surface water and air. Battery manufacture seems to have the lowest emissions per unit cadmium input.

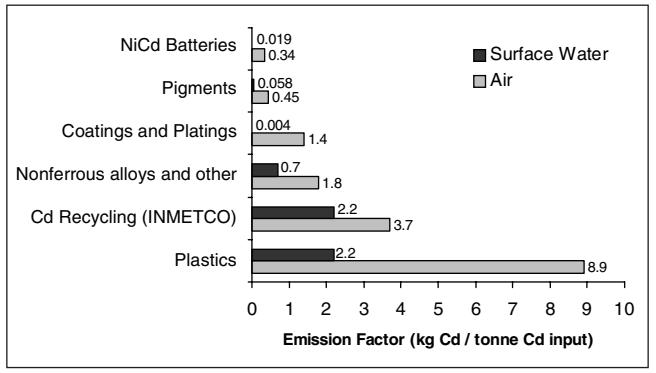

Fig. 2: Emissions factors for cadmium manufacturing processes and recycling, kg Cd per tonne Cd input

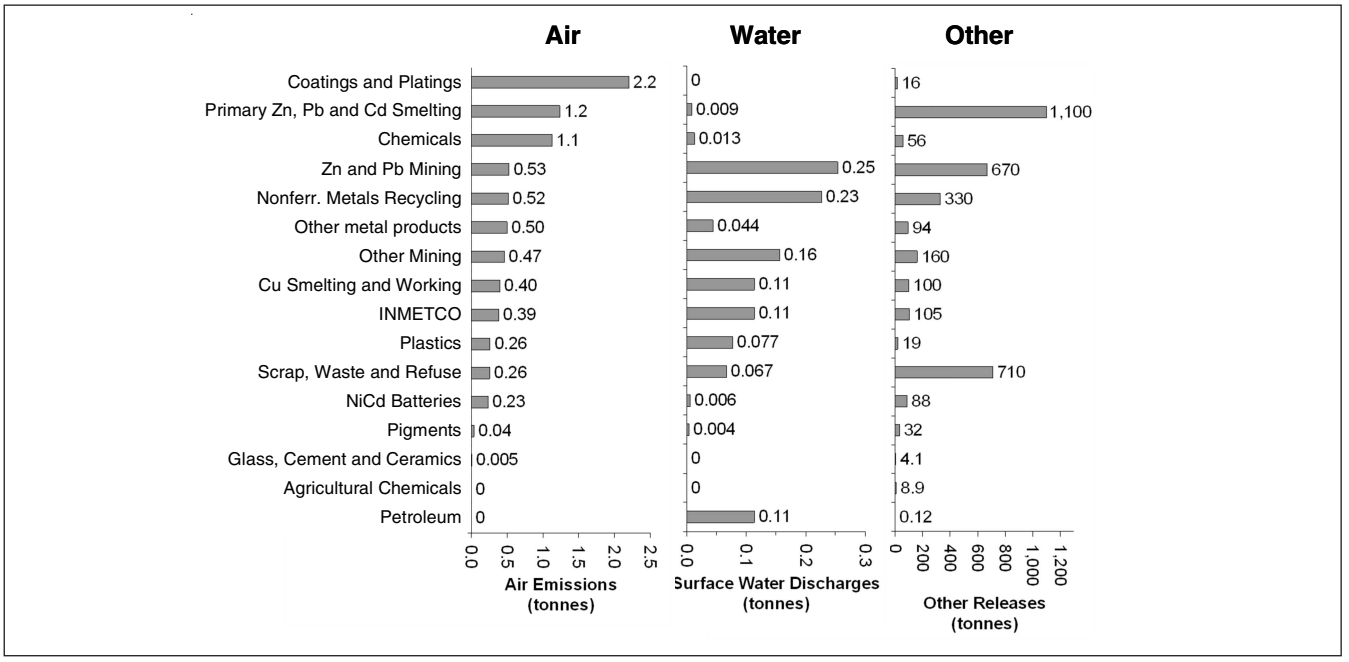

Fig. 1: 2003 Toxics release inventory (TRI) cadmium releases using our aggregation of industrial groups by SIC codes, tonnes

Table 1: 2003 Cadmium releases from manufacturing facilities based on the toxics release inventory (TRI) releases of Cd and Cd compounds by our aggregation of industrial groups by primary SIC codes, tonnes

All values measured in metric tonnes	Coatings and Platings	Primary Nonferrous Metals, except Cu and Al	Chemicals	Lead and Zinc Ores	Secondary Nonferrous Metals (not incl INMETCO)	Other metal products	Other Ores	INMETCO	Copper Smelting and Working	Plastics	Scrap, Waste and Refuse	NiCd Batteries	Pigments	Glass, Cement and Ceramics	Petroleum	Agricultural Chemicals	Total
Facility Count	*20*	*5*	*14*	*7*	*9*	*28*	*9*	*1*	*21*	*20*	*26*	*6*	*7*	*6*	*3*	*1*	*183*
Fugitive Air Emissions	0.01	0.45	0.13	0.12	0.05	0.23	0.11	0.05	0.09	0.09	0.01	0.12	<0.01	<0.01	–	–	1.5
Stack Air Emissions	2.2	0.79	0.99	0.41	0.47	0.27	0.35	0.34	0.18	0.17	0.24	0.12	0.03	<0.01	–	–	6.6
Total Air	**2.2**	**1.2**	**1.1**	**0.53**	**0.52**	**0.50**	**0.47**	**0.39**	**0.27**	**0.26**	**0.26**	**0.23**	**0.04**	**<0.01**	**–**	**–**	**8.0**
Surface Water Discharge	**–**	**<0.01**	**0.01**	**0.25**	**0.23**	**0.04**	**0.16**	**0.11**	**0.08**	**0.08**	**0.07**	**0.01**	**<0.01**	**–**	**0.11**	**–**	**1.2**
Underground Injection	**–**	**–**	**–**	**–**	**0.01**	**–**	**–**	**–**	**–**	**–**	**66**	**–**	**1.1**	**–**	**–**	**–**	**67**
RCRA Subtitle C Landfills	–	510	–	–	–	–	–	–	–	–	490	0.11	–	–	–	–	1,000
Other On-Site Land Releases	–	7.9	<0.01	–	17	7	0.77	–	–	–	0.04	–	<0.01	<0.01	–	–	33
Land Treatment	–	–	–	–	–	–	0.04	–	–	–	–	–	–	–	<0.01	–	0.04
Surface Impoundments	–	–	–	670	–	–	127	–	–	–	–	–	–	–	–	–	800
Other Disposal	–	2.9	39	–	–	–	27	–	–	–	6.2	–	–	–	–	–	76
Total On-Site Land Releases	**–**	**520**	**39**	**670**	**17**	**6.9**	**160**	**–**	**–**	**–**	**500**	**0.11**	**<0.01**	**<0.01**	**<0.01**	**–**	**1,900**
Transfers to POTWs [a]	**0.08**	**0.09**	**0.02**	**–**	**0.04**	**<0.01**	**–**	**<0.01**	**0.04**	**0.04**	**<0.01**	**<0.01**	**<0.01**	**<0.01**	**–**	**–**	**0.35**
Solidification / Stabilization [a]	8.8	550	15	–	170	3.6	–	–	0.72	0.72	14	0.11	31	–	–	8.9	800
Wastewater Treatment [b]	–	–	0.01	–	–	–	–	–	–	–	0.62	–	–	–	–	–	0.63
Other Landfills	–	0.79	0.03	–	0.03	3.5	–	–	0.48	0.48	70	<0.01	0.13	0.58	–	–	76
RCRA Subtitle C Landfills	0.55	–	–	–	0.06	16	–	–	16	16	46	25	–	<0.01	–	–	120
Land Treatment	<0.01	7.2	–	–	–	–	–	–	–	–	–	–	–	3.5	–	–	11
Other Land Disposal	–	–	–	–	6.4	1.3	–	–	–	–	–	–	–	–	<0.01	–	7.7
Other Off-Site Mgt	–	<0.01	0.02	–	0.11	–	–	–	–	–	0.30	–	–	–	0.12	–	0.55
Transfers to Waste Broker for Disposal	0.34	–	0.92	–	–	0.01	0.01	–	<0.01	<0.01	1.5	–	–	–	–	–	2.7
Unknown	–	–	0.20	–	–	2.6	–	–	–	–	–	–	–	–	–	–	2.8
Total Off-Site Disposal	**10**	**550**	**16**	**–**	**180**	**27**	**0.01**	**–**	**17**	**17**	**140**	**25**	**31**	**4.1**	**0.12**	**8.9**	**1,020**
Total Off-Site Waste Mgt	**5.5**	**0.83**	**–**	**–**	**130**	**60**	**0.39**	**110**	**1.7**	**1.7**	**5.4**	**63**	**–**	**–**	**–**	**–**	**370**
Total	**18**	**1,080**	**57**	**670**	**320**	**94**	**160**	**110**	**19**	**19**	**710**	**88**	**32**	**4**	**0.23**	**9**	**3,400**

[a] Metals and metal compounds
[b] Excluding POTWs

Table 1 gives the average emissions factors for each of the processes between 1998 and 2002. Emissions factors (EF) for coatings and platings, nonferrous alloys and other, pigments, plastics and storage batteries were calculated by dividing the mass of cadmium released by the mass of cadmium input for each product category for each year. Values for the five manufacturing categories in Table 1 are the average of these values between 1998 and 2002. Although EFs varied across years, we did not see any trends over the five year time period. The lowest and highest annual values are given in parentheses. EFs for zinc and lead mining and zinc smelting operations were calculated on a facility by facility basis. TRI releases were divided by the mass of zinc in concentrates for mines and the mass of zinc produced for smelters (Roskill 2001).

Emissions to air and water resulting from consumption-related activities were also calculated using emissions factors developed by Ayres (1994). Values are calculated as the quotient of the mass of cadmium releases (as reported in the TRI) and a functional unit of material input or output depending

on data availability. For mining and smelting, data related to cadmium output are available, while for manufacturing processes the USGS provides data related to refined cadmium inputs. In calculating the emissions factors we assume that all cadmium released from a sector originates with a measured cadmium input. This assumption does not hold for the *plastics* and the *nonferrous alloys and other* categories as the releases nearly exceed the inputs (see Table 1). The most likely sources of the additional cadmium input to these sectors are cadmium contained in the fossil fuels consumed at these facilities, trace cadmium in other metals processed at the same facilities or other cadmium compounds not included in the USGS data for consumption of cadmium metal. Other possible sources of this material included cadmium as a trace element in zinc compounds and the cadmium contained in recycled plastics.

3 Results

3.1 Material flow for cadmium in the U.S.

The flow of cadmium through the U.S. economy (**Fig. 3**) is dominated by two major streams. The first is a flow of roughly 8,500 tonnes of cadmium contained in imported batteries. Almost all of this material (>83%) finds its way into MSW landfills at the end of its life. The second major flow is the cadmium contained in zinc ore concentrates mined in the U.S. and exported to foreign refineries. The scale of these flows of unrefined cadmium exports and imports of products containing cadmium suggests that there are economic or environmental barriers to cadmium smelting and manufacturing operations in the U.S. If these barriers are environmental or occupational health and safety regulations, then U.S. policy is shifting its problems to other nations. To reduce the future risk associated with the large stocks of cadmium in landfills, steps should also be taken to separate or recycle cadmium products at the end of their life.

3.2 Trends for cadmium in MSW

Using the assumptions and equations described above, we created a dynamic model to evaluate the impacts of three policy options on the cadmium content of the U.S. MSW stream (**Table 2**).

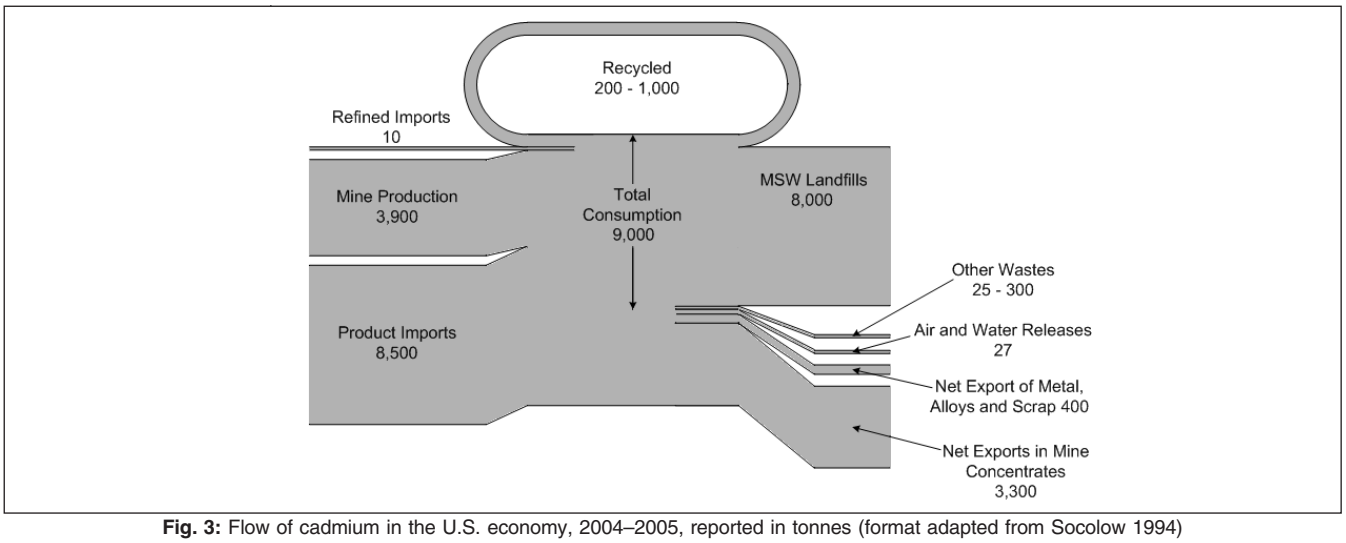

Fig. 3: Flow of cadmium in the U.S. economy, 2004–2005, reported in tonnes (format adapted from Socolow 1994)

Table 2: Cadmium emissions factors for extraction, smelting and manufacturing processes. mean values displayed with the range for the facilities sampled given in parentheses

Process Name (No. of facilities included)	Air	Surface Water Discharge	Other Releases [a]	Units
Zinc and Lead Mining (N = 15)	2.4×10^{-4} ($0–1 \times 10^{-3}$)	8.8×10^{-5} ($0–4 \times 10^{-4}$)	1.0 (7×10^{-3}–5)	kg Cd / tonne Zn in concentrates
Zinc Smelting Operations (N = 4)	9.5 (3–10)	1.9 (0–4)	1,300 (600–2,000)	kg Cd / tonne Zn produced
Secondary Cadmium (N = 1, INMETCO)	3.7	2.2	320	kg Cd / tonne Cd input
Coatings and Platings (N = 436)	1.4 (0.4–4)	4.0×10^{-3} (0–0.02)	220 (130–350)	kg Cd / tonne Cd input
Nonferrous alloys and other (N = 129)	1.8 (0–40)	0.7 (0.4–2)	950 (160–17,000)	kg Cd / tonne Cd input
Pigments (N = 186)	0.45 (0.2–0.7)	0.058 (0.002–0.2)	170 (5–430)	kg Cd / tonne Cd input
Plastics (N = 805)	8.9 (3 – 31)	2.2 (0.5–13)	470 (200–2,300)	kg Cd / tonne Cd input
Storage Batteries (N = 73)	0.34 (0.2–0.8)	0.019 (0.003–0.01)	22 (6–40)	kg Cd / tonne Cd input

[a] Includes data from the following TRI release categories: *underground injection, surface impoundments, other disposal, total on-site land releases, transfers off-site to unknown disposal, and other on-site waste management.*

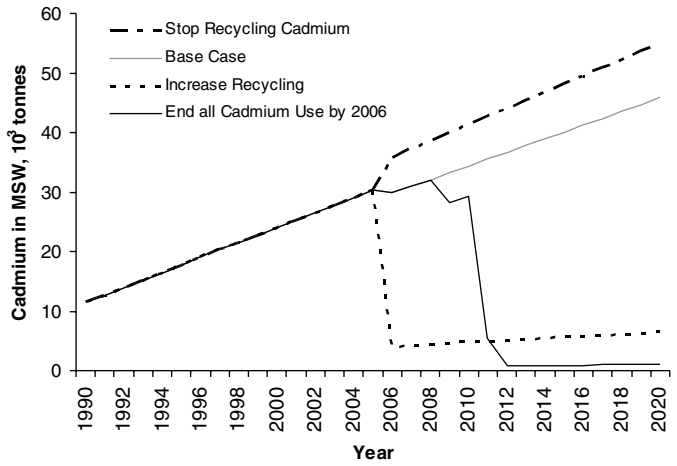

Fig. 4: Cadmium in MSW (10^3 tonnes) under four policy scenarios: base case of 17% and 80% recycling for NiCd PRBs and industrial cells respectively, continue cadmium use without any recycling, increase to 90% PRB recycling, and discontinue cadmium use in all applications in 2006

Although the results are dependant on the assumptions made about future use of cadmium, the relative comparison between policies is consistent when we assume the status quo involves increasing use of cadmium in PRBs (increasing at the rate of 17 million units per year). Under the assumptions of the base case (see Table 2) the amount of cadmium in MSW continues to increase following the growth rate of the cadmium content of products sold in the U.S. but with a delay equal to the lifetime of the products. If the recycling of NiCd batteries were to stop, the amount of cadmium entering MSW would increase by roughly 20%. Conversely, if the recycling rate for NiCd PRBs were increased to 90% in 2006, the amount of cadmium entering MSW would decrease by 87%. We can compare to a scenario where cadmium use is discontinued in all products manufactured or imported for use in the U.S. Under the scenario where cadmium is no longer used, we assume that the cadmium content of zinc ores are stabilized and either landfilled or returned to the mine it was extracted from in accordance with current industry practice. As expected, reductions in the amount of cadmium in MSW would not be realized until the products already in use have been discarded. According to our calculations, we would expect a slight decrease in cadmium entering MSW occurring at the lifetime of the shorter lived cadmium products, around 3 years. The larger decrease, however, occurs after roughly 5 years when most of the NiCd PRBs produced in 2005 have been discarded. Results for our analysis of the cadmium content of MSW under these four scenarios are shown in **Fig. 4**.

3.3 Emissions from recycling

One major disadvantage to cadmium recycling is the emissions resulting from the recovery process. The difference in total emissions for the three scenarios over the time period 2005 to 2020 can be found in **Table 3**. Raising recycling rates causes an increase of almost 500 tonnes cadmium air emissions when compared to the status quo while 110 tonnes air emissions could be avoided if recycling were discontinued. Although recycling cadmium initially results in significant reduction in cadmium entering MSW landfills, our analysis shows that ending all cadmium use reduces all forms of emission over the time period of fifteen years. This is dependent on what happens to the cadmium contained in the zinc ore concentrates that would presumably continue to be mined in each of these scenarios. If cadmium is no longer consumed in the U.S., this material would either be disposed of by zinc smelters as hazardous waste or exported to consumers in nations that still allow its use. However, increasing the recycling rate could also have a detrimental effect on primary cadmium by increasing the supply and lowering prices thereby reducing the incentive for separating cadmium from zinc ore wastes.

Table 3: Policy options evaluated using the dynamic model

	Collection rates for NiCd Battery Recycling	
	PRB	Wet Cell
Base Case	17%	80%
End Recycling	0%	0%
Increase Recycling	90%	80%

The risk associated with cadmium in landfills is remains a source of debate. A research group at the University of Florida has performed experiments using the standard Toxicity Characteristic Leaching Procedure (Federal Register 1986, EPA 1996) for several types of electronic wastes and found that although leachates from computer printed wire boards and cathode ray tubes do not exceed the heavy metal concentrations specific for toxicity characteristic materials (2003), slightly altering the experimental setup can cause the levels of heavy metals from several electronic devices to exceed toxicity characteristic levels (2004). In a review of hazardous materials in landfills Slack (2005) reports cadmium levels of 0.0001 to 0.4 mg/L detected in landfill leachates. Ehrig places average cadmium concentrations in landfill leachates at 0.005 mg/L. At present heavy metal concentrations in MSW leachates are generally low due to attenuating processes such as sorption and precipitation occurring within the waste. Concentrates of heavy metals in

leachates may increase in the future due to lowering of leachate pH due to the oxidation of materials in the landfill, however experiments suggest the time frame for this occurrence to be several thousand years (Kjeldson 2002).

3.4 Energy

The energy benefit of recycling cadmium over the next 15 years was calculated and it was found that 16 MWh energy savings could be achieved when compared to refining primary cadmium from zinc ore concentrates. This calculation is based on 103 MJ energy needed for the primary production of the cadmium required to produce one kg of NiCd batteries compared to 5.6 MJ for recycled material. Energy recovery of 0.8 MJ/kg NiCd batteries from combustion of battery plastics and paper as well as the additional 6.9 MJ/kg NiCd battery required beyond the disposal energy for collection are also accounted for in this calculation (Lankey 1998).

Schmidt and Beyer (1999) performed an analysis of the environmental impact of recovering the plastic housing from automotive batteries and found that recycling outperformed landfilling in all relevant cases, however they were not able to quantify the differences in terms of toxic releases. The tradeoffs between recycling and energy recovery were less conclusive and were found to depend on the energy mix used to perform the analysis.

3.5 Importance of zinc recycling in cadmium flow

If we choose to continue using cadmium, coupling cadmium recycling with zinc recycling is important for reducing cadmium releases. The opportunities to recover zinc include separation from EAF dust collected from galvanized steel and zinc alloy recycling or recovery from tire incineration fly ash (**Fig. 5**). Tan (2005) performed an LCA for zinc recycling and using the Eco-indicator '99 method for impact assessment found the energy savings achieved through zinc recycling far outweigh the direct increases in air pollution and heavy metal releases associated with recycling. Nonetheless, currently only about 25% of U.S. zinc consumption is recycled material (USGS 2005b).

Fig. 5 depicts several opportunities that exist for cadmium and zinc recovery from products. In addition to recycling NiCd batteries, cadmium could be beneficiated from zinc coatings stripped from galvanized steel, EAF dust and fly ash from tire combustion. Cadmium is currently recovered in the U.S. from only a fraction of the EAF dusts generated. More of these dusts are exported for metals recovery or disposal.

Increased zinc recycling impacts the material flow of cadmium both by offsetting primary zinc production and its co-product flow of primary cadmium and by concentrating the trace cadmium content of zinc products for proper management as hazardous waste or re-use in products. Perverse incentives such as taxing the disposal of cadmium-bearing wastes could actually harm the environment by inducing secondary smelters to hide their flow of cadmium. Providing secure markets for recycled cadmium provides an incentive for improving recovery processes. Without recycling this material would likely end up in landfills where the cadmium could leach into the soil and groundwater over time.

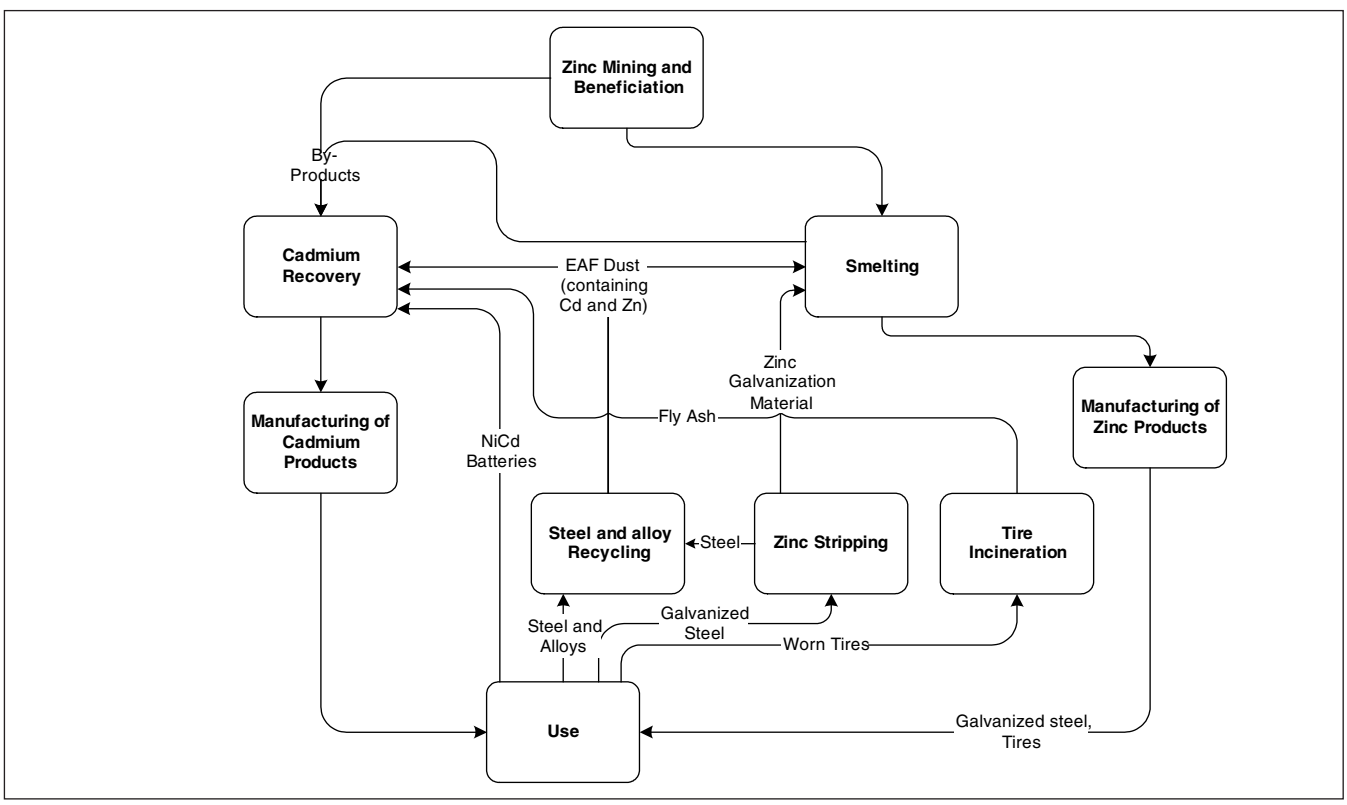

Fig. 5: Intersections of the Zn and Cd cycles and opportunities for the recovery of Cd from used products

4 Conclusion

We find that imported NiCd batteries represent the largest flow of cadmium contained in products sold in the U.S. Although cadmium recovery is possible for NiCd batteries as well as the electric arc furnace dust resulting from steel recycling, less than 20% of the cadmium in the product waste stream is currently collected for recycling. We believe that the fate of most cadmium in the U.S. is municipal solid waste landfills.

We propose three steps for reducing the risk associated with the cadmium flows considered: 1) Alter use patterns toward low exposure, long-lived products with high recycling rates while enforcing low releases during recycling. 2) Improve the collection rates of cadmium-bearing products to keep them out of MSW landfills. Ensure that recycling occurs with minimal emissions and proper management of residuals. 3) Discourage the dissipative use of cadmium in products with short lifetimes or for which no feasible recycling route exists coupled with proper stabilization and disposal of the cadmium content of zinc concentrates that there is no demand for. In addition to these three, we recommend that policy measures be taken to provide the necessary data required to improve our understanding of the flow of cadmium into the U.S. in the form of products imports and the amount of cadmium lost or disposed of by recycling processes.

5 Recommendation and Perspective

This analysis has demonstrated the large flow of cadmium imports through the U.S. economy and into MSW landfills. It was shown that while increasing collection of cadmium for recycling has a more immediate impact on the amount of cadmium entering landfills, ending cadmium use would end this flow altogether once all the products currently in the economy had been discarded, the greatest decrease occurring after a 5 year time lag. However, as noted by Fthenakis (2004), ending the use of cadmium would cause primary zinc smelters to dispose of cadmium as a waste product at additional cost rather than selling its cadmium content. While this additional cost might provide a stimulus for the secondary zinc production, demand for zinc exceeds the amount of secondary material available. This highlights the importance of considering both the cadmium and zinc cycles in formulating an effective policy for the control of cadmium. In the present study only the stocks and flows of cadmium are investigated. Continued research is needed to improve our understanding of the exposures and risks associated with these flows.

Undesired flow of cadmium might be reduced to levels considered more acceptable by allowing its use in products with long lifetimes, low risk of exposure during use and high recycling rates with minimal losses. Industrial NiCd batteries and thin film photovoltaics are both reasonable candidate technologies. Thin film photovoltaics are manufactured by depositing thin layers of cadmium sulfide or cadmium telluride and are interesting because of their potential improvements in efficiency when compared with traditional silicon cells. Cadmium telluride PV cells are expected to last 25 to 30 years (Fthenakis 2000), while industrial NiCd batteries have lifetimes in the range of 15 to 20 years. Thin film solar cells have found niche markets in the southwest U.S. and in Germany as a result of public incentives for solar power. Processes have been developed for the recovery of cadmium from thin film PV cells with minimal losses (Fthenakis 2000, Rich 2004).

Finally, we request that environmental policy-makers consider requiring the public release of several additional data to allow for the informed management of cadmium. Manufacturing facilities and importers should be required to report the cadmium content of their products while smelting facilities should report the cadmium content of their inputs and marketable outputs. These reports could be handled under the system already in place for the handling of the TRI. Making this data available on an annual basis would allow analysts to more accurately calculate the toxic content of products currently in use and the fraction of material lost in recycling.

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