Charging up Battery Recycling Policies

Extended Producer Responsibility for Single-Use Batteries in the European Union, Canada, and the United States

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Summary

Extended producer responsibility (EPR) policies have proven effective at raising consumer awareness, expanding waste collection infrastructure, and shifting costs of end-of-life (EOL) management from municipalities to stewardship organizations. Yet, such policies have been less successful in advancing waste management programs that ensure a net environmental benefit. This article analyzes how EPR policies for single-use batteries in the European Union (EU), Canada, and the United States address the environmental costs and benefits of EOL management. Considering these EPR policies is instructive, because single-use batteries have high collection costs and are of relatively low economic value for waste processors. Without deliberate planning, the environmental burdens of collecting and recycling such batteries may exceed the benefits. This article considers how EPR policies for single-use batteries integrate performance requirements such as collection rates, recycling efficiencies, and best available techniques. It argues that for such policies to be effective, they need to be extended to address waste collection practices, the life cycle consequences of EOL management, and the quality of recovered materials. Such strategies are relevant to EPR policies for other products with marginal secondary value, including some textiles, plastics, and other types of electronic waste.

Introduction

Extended producer responsibility (EPR) is gaining momentum as a policy strategy for addressing the challenges of waste management and recovery in the European Union (EU), Canada, the United States, and elsewhere. For instance, between 1991 and 2011, more than 70 EPR laws were enacted at the state level in the United States covering products such as paint, thermostats, mattresses, and batteries (Nash and Bosso 2013). EPR legislation is predicated on two concepts: first, that producers should bear responsibility for the postconsumer management of their products and packaging and, second, in doing so, producers gain incentives to consider environmental factors when designing products and packaging, such as upstream environmental impacts during sourcing and manufacturing and the potential for materials recovery downstream at end of life (EOL) (OECD 2001; PPI et al. 2012). EPR may offer potential benefits, such as shifting EOL management costs from municipalities to producers, encouraging producers to internalize life cycle costs, and promoting environmental benefits through improved product design, reverse logistics, closed-loop supply chains, and resource recovery (McKerlie et al. 2006; Kalimo et al. 2012; Austin 2013; Souza 2013; Blanco and Cottrill 2014).

Despite the potential of EPR to affect changes throughout the life cycle of products, scholars have observed that EPR policies have largely succeeded in shifting the costs for product collection and recycling from municipalities to producers and third-party stewardship organizations (Lindhqvist and Lifset 2003; Sachs 2006; Lifset and Lindhqvist 2008; Huisman 2013). It is often assumed that even if EPR policies do not

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promote changes in product design, such as those that might minimize environmental impacts during sourcing and production or promote materials recovery at EOL, such policies at least yield environmental benefits by increasing collection and recycling of waste products. Although scholars have undertaken comparative studies of EPR policies, examining differences such as individual versus collective responsibility, funding strategies, and issues of policy harmonization, few studies have focused specifically on how EPR policies are designed to address the environmental consequences of EOL management (Sachs 2006; McKerlie et al. 2006; Lepawsky 2012; Hickle 2013; Nash and Bosso 2013). Yet, how an EPR program is structured can determine whether the collection and recycling of discarded products yields a net environmental benefit or burden.

In this article, we argue that it is necessary to give greater consideration to the environmental consequences of EOL management, including how products are collected and recycled, in the design and assessment of EPR policies. To advance this analysis, this article examines existing and proposed EPR policies that apply to single-use consumer batteries in the EU, Canada, and the United States. Single-use batteries, such as 9-volts or AA batteries, are a useful class of products to consider, given that they are a marginal waste stream of relatively low economic value for waste processors, involve high volumes of dispersed waste that entail collection challenges, and incorporate multiple materials (including refined metals and chemicals) that pose challenges for recycling and resource recovery (Linden and Reddy 2002; Olivetti et al. 2011). In these respects, single-use batteries are an important subset of other forms of consumer waste that may have marginal value as secondary sources of recovered materials, such as textiles, glass, light bulbs, some plastics, and some types of e-waste, such as liquid crystal display screens (MacBride 2012; Kasper et al. 2015).

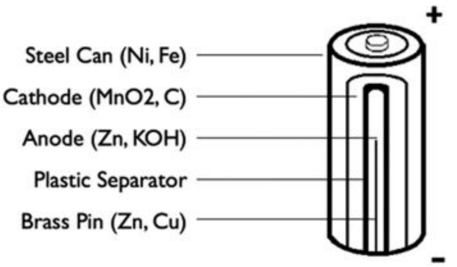
This analysis makes two contributions to analyses of EPR policies, one conceptual and one applied. The conceptual contribution is to expand the discussion of the environmental outcomes of EPR policy beyond the potential for changes in the sourcing and design of products to specifically include the consequences of collecting, sorting, and recycling waste products at EOL. Both policy makers and scholars have focused more on the allocation of responsibility, funding mechanisms, and the potential for eco-design in assessing EPR policy than how policies are structured to encourage efficient modes of collection and high levels of materials recovery. The applied contribution is to suggest policy strategies for existing and future EPR programs for batteries that will ensure that the collection and recycling of single-use batteries yields a net environmental benefit. This is particularly important given that EPR legislation that applies to single-use batteries is under consideration in several U.S. states, including California, Minnesota, and Connecticut.

The Challenges of Recycling Single-Use Batteries

Portable batteries are divided into two categories: single-use batteries, such as alkaline-manganese and zinc-carbon batteries (which are the AAs and AAA batteries used in flashlights, toys, and remote controls) and rechargeable batteries, such as nickel-cadmium and lithium-ion batteries (which are used in portable electronics, cell phones, and power tools). Although sales of single-use batteries are declining in the United States, largely owing to the growth of rechargeable battery sales, single-use batteries still accounted for 80% of U.S. battery sales in 2010 by unit. Of those single-use batteries, 76% were alkaline-manganese batteries (CBR 2011a; Ng and McCarthy 2014). Recently, questions have arisen regarding the environmental impact of such batteries (PSI 2014). But since the late 1980s, the environmental profile of single-use batteries has been improved in two important ways. First, between the 1960s and 2000, specific energy output for alkaline batteries increased roughly 60% (Linden and Reddy 2002; VARTA 2011). Second, the industry phased out mercury from single-use batteries in the 1990s (Telzrow 1989; Telzrow 1995). Despite these advances, reducing the environmental consequences of single-use batteries across the life cycle remains a challenge for several reasons.

First, producing single-use batteries (including raw material extraction and manufacturing) is resource and energy intensive. Considered over its life cycle, approximately 88% of the environmental impact of a single-use battery is incurred during the sourcing and processing of raw materials and subsequent manufacturing (Olivetti et al. 2011). Measured on a per watt-hour basis, those impacts are substantial. It takes more than 100 times the energy (measured as cumulative energy demand [CED]) to manufacture an alkaline battery than is available during its use phase. The greenhouse gas (GHG) emissions per watt are approximately 30 times that of the average coal-fired power plant.[1] Thus, if the goals of EPR are going to be realized in the case of single-use batteries, it is necessary to reduce environmental impacts throughout the battery life cycle, including the sourcing of raw materials. Despite the impacts of sourcing and manufacturing single-use batteries, public discussions and policy initiatives with respect to alkaline batteries have focused almost entirely on EOL management.

Second, alkaline battery manufacturers have exacting standards for battery-grade raw materials, such as the electrolytic manganese dioxide and synthetic graphite used in the cathode, the powdered zinc and potassium hydroxide used in the anode, and the nickel-plated steel can that gives the battery its shape (see figure 1). Battery-grade electrolytic manganese dioxide and powdered zinc must have low levels of impurities, precise structural properties (such as hardness and surface area), and, in the case of powdered zinc, alloying additives (such as indium, lead, bismuth, and aluminum). Indeed, improvements in batteries, such as eliminating mercury and extending battery life, were made possible largely by improvements in the purity and quality of these raw materials (Telzrow 1989; Nardi and Brodd 2010). These quality standards pose hurdles for recovering materials from spent batteries for use in new batteries. Only one company markets an alkaline-manganese battery that incorporates materials recycled from spent batteries in its active ingredients. The Energizer EcoAdvantage currently contains



Constituent Materials	Symbol/ Formula	Weight	Composition by Percentage
Electrolytic Manganese Dioxide	MnO_2	13g	39.4%
Nickel-Plated Steel	Ni, Fe	6.0g	18.2%
Zinc	Zn	5.8g	17.6%
Potassium Hydroxide	KOH	3.7g	11.2%
Graphite	C	1.2g	3.6%
Brass	Zn, Cu	1.0g	0.03%

Figure 1 Structure of a typical single-use alkaline-manganese battery. Key battery chemistries are noted (*Source*: Olivetti et al. 2011, 17).

4% recycled zinc or manganese (which material is not publicly disclosed) and went on the market in February 2015. Energizer has signaled its intent to increase the proportion of recycled material in batteries to 40% by 2025 (Energizer 2015).

Third, it is difficult to recover high-quality secondary materials, such as materials suitable for reuse in new batteries, from spent single-use batteries. In part, this is a product of the electrochemistry of single-use batteries. The current produced by an alkaline-manganese battery is a result of a chemical reaction that results in a used battery containing an array of materials, including manganese oxide, zinc oxide, and potassium superoxide (De Michelis et al. 2007; Cabral et al. 2013) that are chemically distinct from the raw materials needed for new batteries. Historically, most materials recovered from spent batteries have gone to other end uses, rather than being reused for battery manufacturing. For example, an Ontario-based recycler recovers zinc and manganese for use in agricultural fertilizers and steel and nickel for the metals market whereas paper and plastic are burned to produce electricity (RMC 2014). A U.S.-based recycler recovers approximately 35% of the manganese, zinc, and iron and other metals for reuse in the metals market, but a portion of the metals are diverted to slag and the paper and plastic are burned to produce electricity (Call2Recycle 2014a). In Germany, one system recovers almost all of the zinc and iron for reuse in the metals industry, but diverts almost all manganese to slag (which can be used in road construction or other applications) (Rombach et al. 2006). Although it is possible to close the loop on alkaline batteries recycling and recover battery-grade materials from spent batteries, as demonstrated by Energizer's EcoAdvantage batteries, there is no information available regarding the environmental burdens and benefits of such recycling techniques (Energizer 2015).

Last, collecting and preparing single-use batteries for recycling is costly and energy intensive. Before single-use batteries can be recycled, they must be collected from consumers, sorted, and transported (often over long distances) to recycling facilities. In the case of single-use batteries, the economic cost of collecting, sorting, transporting, and recycling single-use batteries often outweighs the economic value of the materials recovered from these batteries. For instance, one recently proposed system costs US$1,286 per tonne (not including transportation costs) and yields raw materials worth US$382 per tonne (Bonhomme et al. 2013). As a result, producers, governments, or other entities generally must pay a fee for the recycling of single-use batteries. From an environmental perspective, equally challenging is that the environmental consequences of collecting, sorting, transporting, and recycling single-use batteries can exceed the environmental benefits of recovering secondary materials from batteries for other uses (Aumônier et al. 2000; Nelen et al. 2013). Currently, only four companies recycle alkaline batteries in North America. In the recycling scenarios modeled in Olivetti and colleagues (2011), the burden of collecting and transporting spent batteries to processors resulted in a net negative environmental impact for four of five recycling scenarios modeled.

Policy Approaches to Battery Collection and Recycling

Historical Background

Management of batteries in the waste stream has been a policy concern in Europe and North America since the 1980s. Initial policies regarding consumer battery disposal centered on toxics reduction. Owing to the fact that batteries were the largest source of heavy metals in municipal waste streams, the EU and the United States targeted both single-use and rechargeable batteries for collection. Out of those efforts emerged numerous policies, notably the 1991 European Battery Directive (Council Directive 91/157/EEC) and the 1996 U.S. Mercury-Containing and Rechargeable Battery Management Act (Public Law 104–142, 110 STAT. 1329). Though distinct, both policies established uniform labeling requirements and supported collection and recycling of rechargeable batteries containing heavy metals such as lead and cadmium. Although policy makers did consider recycling mandates for single-use batteries, such provisions were omitted from legislation after industry agreed to phase out mercury from most single-use batteries on the market by the mid-1990s.

Policy discussions regarding the fate of single-use batteries shifted from a focus on toxic reduction toward resource recovery for a variety of reasons in the early 2000s. Some jurisdictions continued to classify alkaline batteries, despite the mercury phase-out, as hazardous and unsuitable for municipal waste streams (California, State of 2001). In municipalities where single-use batteries are collected, either voluntarily or by law, the cost of diverting them from the waste stream is high (US$2,700 per tonne in some California municipalities), leading to calls to shift such costs to producers (Anonymous 2014). Recovering single-use batteries advances broader efforts to promote waste reduction and advance resource recovery goals. For instance, recovering single-use batteries may also advance efforts to collect rechargeable batteries—for which collection rates have lagged behind expectations and some of which still contain toxic materials—given that most consumers do not or cannot distinguish between various battery chemistries (PSI 2014, 31). Anecdotal evidence indicates that accepting single-use batteries has led to a 25% increase in the collection of rechargeable batteries in some instances (Smith 2014).

Thus, since 2000, battery policies have been expanded to include collection and recycling of single-use batteries. A revised EU battery directive took effect in 2006 that instituted targets for collection and recycling to include single-use batteries (Council Directive 2006/66/EC). Since 2008, several Canadian provinces, led by British Columbia and Ontario, have extended EPR programs to include single-use batteries. In the United States, states including Minnesota, California, and Connecticut have considered or, in the case of Vermont, adopted, mandatory recycling for single-use batteries (Minnesota 2013 SF 639; California 2014 Bill AB2284; Vermont 2014 Act H 695 No. 0139). The most sustained discussions have been in California, which has classified single-use batteries as hazardous waste, based on corrosivity and potential for leaching in landfills, since 1990. In 2006, the state designated batteries as a form of "universal waste" under hazardous waste law, and forbid disposal in household trash, which has resulted in efforts to develop state-level policies supporting EPR for single-use batteries (CEPA 2006).[2]

In this section, we consider existing and, in the case of the United States, proposed EPR legislation that applies to single-use batteries (see table 1). In each case, these policies shift financial responsibility for EOL management of batteries to producers; these policies allow for collective approaches, such as product stewardship organizations (as opposed to individual producer responsibility strategies, in which each producer takes responsibility only for the specific batteries they put on the market), and all take different approaches to policy implementation, including collection targets, recycling standards, reporting, and enforcement. In assessing these approaches, we give particular attention to how the policies are structured to ensure that the collection and recycling of single-use batteries yields a net environmental benefit.

Case Study: The Battery Directive in the European Union

Although the Council of European Communities adopted the first Battery Directive in 1991 (Council Directive 91/157/EEC), that policy focused on toxics reduction. The 2006 Battery Directive, however, established EPR requirements for all batteries and addresses the importance of recycling and resource recovery.[3] The directive includes strict, prescriptive requirements that guided the development of battery regulations in European member states. For example, the directive requires countries to improve EOL waste infrastructure by setting collection targets—meaning the percentage of spent batteries collected—that increase over time—at 25% by 2012 and 45% by 2016. It also sets minimum standards for recycling efficiency—meaning the percentage of resources recovered from a spent battery during the recycling process—at 50% for single-use batteries (and higher for some rechargeable batteries). And it requires recyclers to use best available techniques (BATs) to treat waste batteries.

Collection rates are a basic component of most battery legislation in the EU and elsewhere. The 2006 Battery Directive defines the collection rate as the weight of collected waste batteries in the current year divided by the average weight of battery sales in the member state for the present and previous 2 years. Such calculations provide a reasonable estimate of collection rates. But such calculations do not account for numerous uncertainties, such as the life span of a single-use battery (which can range from weeks for high-drain applications, such as cameras, to years for low-drain applications, such as remote controls), consumer behavior (which includes hoarding batteries before disposal or recycling), or the number of batteries that are shipped inside products (such as toys, cameras, or remote controls) (PSI 2014). Most jurisdictions with rechargeable or single-use battery policies, including the United States and Canadian provinces, adopt a similar approach to estimating collection rates.

Although many EPR policies focus solely on collection rates, the 2006 Directive is notable for requiring the development of procedures to calculate and report recycling efficiency. Recycling efficiency assesses the performance of recycling processes by measuring the mass of nonwaste materials recovered from spent batteries against the mass of spent batteries sent for recycling. Making such calculations requires resolving questions, such as: Should batteries be weighed on a wet or dry basis? Should carbon used for energy recovery count as recycled? Should recovered materials diverted to slag count as recycled? After consultation with industry stakeholders, the commission released its regulations in 2012. It calculates the input mass on a dry weight basis, but includes fluids and acid. It does not count carbon used for energy recovery as recycled. And it does count materials used for slag, with some restrictions, as recycled. Because the recycling efficiency is based on mass, factors associated with the collection, transport, or sorting of batteries are excluded from the analysis. Following this methodology, recyclers must provide a detailed accounting of inputs and outputs from the recycling process and report their recycling efficiency for each battery chemistry. This regulation came into force in 2014 with initial reports due in 2015 (EC 2012).

The other major requirement the directive requires are BATs for treating waste batteries. BATs, as outlined by the EU, are intended to increase recycling and recovery rates, reduce waste and energy emissions, decrease use of raw materials and hazardous substances, and evaluate the net environmental impact

Table 1 Comparison of battery EPR legislation and implementation: North America and Europe

	European Union	*British Columbia*	*Ontario*	*United States (model bill)*
Broad EPR regulation (if applicable)	2006 EU Battery Directive (2006/66/EC)	British Columbia Recycling Regulation (BC Reg 449/2004)	2002 Waste Diversion Act (SO 2002, c 6)	Model Consumer Battery Stewardship Act
Specific plan or statute	Individual statutes adopted by each nation state	Call2Recycle, All-Battery and Mobile Phone Collection and Recycling Plan for BC	Consolidated Municipal Hazardous or Special Waste Program Plan (2010)	n/a
Scope of products	Single-use and rechargeable consumer batteries	Single-use and rechargeable batteries	Single-use batteries	Single-use and rechargeable consumer batteries
Recycling guidelines	Excludes energy recovery, incineration, and landfilling as recycling	None	Waste Diversion Ontario does not consider energy recovery or downcycling as recycling.	Defers to state definition of recycling
Collection targets	Statutory target: 25% by 2012, 45% by 2016	Statutory target: 75%; plan target: 25% by 2012 and 45% by 2016	Statutory target: None specified; plan target: Stewardship Ontario set a 25% collection target by 2012 and 45% by 2016.	Proposed target: 10% within 2 years; 20% within 5 years of implementation
Recycling efficiency targets for single-use batteries	Statutory target: 50%	Statutory target: None; plan target: 50%	Statutory target: none; plan target: 80%	Proposed target: None
Technology standards for recycling	Requires best-available techniques	None	None	None
Environmental reporting	Collection rates and recycling efficiency annually; additional information from member states every 3 years	Collection rates and recycling efficiency annually	No specific requirements	Collection rate, weight of batteries collected, and general account of recycling practices annually or biannually

Note: EPR = extended producer responsibility; n/a = not applicable.

of the BAT's waste emissions (Council Directive 96/61/EC). Theoretically, these guidelines encourage the use of recycling technologies that are more environmentally beneficial and energy efficient. Unfortunately, a lack of specific EU guidelines defining BATs for battery processors means that member states have not established or adopted BAT standards.[4] Therefore, the BAT requirement remains underimplemented and ineffective in practice.

Despite the strengths of the 2006 Directive and subsequent regulations, its major shortfall is a lack of adequate enforcement measures, which is also a problem with battery policies in the United States and Canadian provinces. For EU collection targets in 2012, this was not a significant issue: only three EU members failed to achieve the 25% battery collection target, and many countries exceeded this goal (Perchards and SagrisEPR 2013). But it is anticipated that more EU nation states will struggle to meet the target 45% collection rate standard in 2016. It remains unclear how shortfalls in collection, recycling efficiency, or the implementation of BATs can be addressed. In all cases, the 2006 Directive delegates penalties and enforcement to member states, which will likely lead to uneven performance in the future (Mudgal et al. 2014).

Case Study: Product Stewardship in British Columbia and Ontario

Canadian battery policy has been implemented at the provincial level in British Columbia, Ontario, Manitoba, and Quebec. These programs have been developed under provincial product stewardship and waste management framework policies, which apply to a range of consumer products, instead of product-specific legislation. As a result, most Canadian EPR policies delegate category-specific standards to product stewardship plans (PSPs) developed by provincial governments or stewardship organizations. This regulatory model promises more flexibility for producers (Hickle 2013). Provinces have taken different approaches with respect to the implementation of such PSPs, as is evident in British Columbia and Ontario.

British Columbia's battery recycling program was developed as part of the Recycling Regulation under the Environmental Management Act, which was amended in 2008 to include batteries as a special waste.[5] This framework approach includes only general guidelines regarding the structure and implementation of PSPs, making it an "industry-managed, outcomes-based approach" (Hickle 2013, 251). In 2010, Call2Recycle, a private, nonprofit stewardship organization specializing in managing EOL cell phones and batteries, proposed a PSP for batteries that was accepted by British Columbia. Under the PSP, Call2Recycle, which represented major battery producers, assumed responsibility for developing a collection system, public outreach, handling transport and recycling, and meeting reporting standards. In the case of single-use batteries, the initial stewardship plan proposed collection targets for single-use batteries rising from 12% in 2010 to 40% in 2014 (Call2Recycle 2010). The program specified a recycling efficiency rate of 50% for single-use alkaline batteries based on existing recycling practices. Call2Recycle pledged that recycling would be handled by certified recyclers selected based on a competitive bidding process (Ibid).

Ontario's battery recycling program was developed under Ontario's Waste Diversion Act of 2002.[6] To date, a government-designated, private stewardship organization, Stewardship Ontario, has overseen the planning, implementation, and operation of stewardship programs for municipal hazardous and special wastes targeted by the provincial government, including batteries. Stewardship Ontario's Orange Drop Program is a unified service that accepts regulated wastes at collection points around Ontario. Stewardship Ontario established target collection rates for single-use batteries rising from 20% in 2011 to 45% in 2016 and it adopted a more ambitious recycling efficiency standard of 80%. Unlike British Columbia, where producers contract with Call2Recycle to participate in its stewardship program, producers have no such option in Ontario. Stewardship Ontario is currently designated by the Ontario provincial government and funded through fees levied on producers.

The implementation of these PSPs raises important questions about performance and reporting of programs developed under framework EPR programs. The PSPs in both provinces have lagged behind stated targets. For instance, in 2013, Call2Recycle collected only 16% and Stewardship Ontario collected only 17% of the available batteries, well below their respective targets of 32% and 30% for the year (Stewardship Ontario 2009, 2013; Call2Recycle 2010, 2014a). In neither case did these failures result in the stewardship organizations facing any penalties; instead, both organizations outlined plans for continuing to improve advertising, public outreach, and collection networks. Under the framework legislation authorizing these PSPs, it is unclear if penalties would ever be imposed for continued failure in the future.

There are also significant uncertainties in how recycling efficiencies are reported under these PSPs. Both plans report recycling efficiencies for single-use alkaline batteries exceeding 80% in 2013 (well above Call2Recycle's goal) (Call2Recycle 2014a; Stewardship Ontario 2013). In both cases, however, recycling efficiency figures were reported with no supporting methodology (Ibid). Recently, this has become a point of concern. In 2013, Call2Recycle proposed a PSP for Ontario, which, if accepted, could displace the existing Stewardship Ontario program. Some stakeholders have raised concerns regarding Call2Recycle's recycling performance, focusing on Call2Recycle's practice of shipping spent single-use batteries to the United States, where more of the recovered material is recovered as slag through a pyrometallurgical process. In contrast, Stewardship Ontario's current recycling partner, which recycles most materials for use in the metals and agricultural industries through mechanical processing, is described as "up-cycling" given that they claim it displaces virgin metal consumption (RMC 2014; CAPE et al. 2014). But standards for recycling vary across jurisdictions, and in some localities recovering metals for agricultural uses may not constitute recycling. Thus, without better reporting requirements and more carefully defined metrics, the net environmental benefits of these respective programs remains unclear.

Case Study: The Model Consumer Battery Stewardship Act in the United States

The United States has adopted EPR policy on a state-by-state basis, and currently eight states have policies pertaining to rechargeable batteries, most of which were enacted in the early 1990s (Nash and Bosso 2013; Call2Recycle 2014b). Historically, the U.S.-based battery industry maintained that mercury-free single-use batteries could be safely disposed of in the municipal waste stream and that recycling offered no environmental benefits (NEMA 2002). But, as the Canadian provinces, California, and other states began to explore EPR legislation for single-use batteries in the 2000s, U.S. industry stakeholders changed their position. In 2010, the National Electrical Manufacturers Association (NEMA) commissioned a life cycle assessment (LCA) by Massachusetts Institute of Technology researchers on prospects for battery recovery and recycling (Olivetti et al. 2011). In 2011, the industry hosted a multistakeholder battery summit to discuss policy strategies for battery recycling. Later that year, leading battery manufacturers formed the Corporation for Battery Recycling (CBR) with the stated goal of developing a national, voluntary program for recycling single-use batteries (CBR 2011b, 2012).

These efforts are motivated by the potential for policy fragmentation in the United States. In the past two years, California, Minnesota, Connecticut, and Vermont have considered EPR legislation that applies to single-use batteries, with Vermont enacting the first such bill in May 2014.[7] The Vermont bill, however, includes only broad provisions for an industry-proposed stewardship plan and includes no statutory performance-based requirements regarding collection rates or recycling efficiency. Industry has actively lobbied at the state level for EPR for batteries, including for the Vermont legislation. Since 2014, CBR has worked with Call2Recycle and the Product Stewardship Institute (PSI) to propose a model all-battery EPR legislation, known as the Model Consumer Battery Stewardship Act, in hopes of harmonizing policies across states.[8] Legislation modeled on this proposal is currently being considered in California and Connecticut.

As is consistent with U.S. EPR legislation more generally, the model bill includes detailed requirements regarding stewardship plans, financing, and enforcement (Hickle 2013), but little attention to recycling performance or environmental benefits. For instance, producers must participate in a battery stewardship plan if they sell batteries in the state; stewardship plans must detail the collection points, educational efforts, recycling processes, and financial model; and stewardship organizations must report annually on plan performance. An important uncertainty is how battery-containing products (including medical devices) are handled; such products were exempted from the Vermont legislation. The bill does include novel reimbursement incentives and enforcement provisions, which distinguish it from the EU and Canadian bills. For example, if a stewardship organization exceeds the collection target, it can recover reimbursement costs for excess collection from other producers or plan operators.[9] If a producer fails to participate in a stewardship plan, the model bill also includes a "right of private action," which allows a producer or stewardship organization in compliance with the law to bring civil action against a producer who has failed to comply with the act.[10]

Although such penalties reward compliance with collection targets and discourage free riders, the bill does not include similarly detailed or stringent provisions to address collection and recycling practices. Whereas the bill requires a detailed financial audit of collection activities, stewardship organizations are only required to provide a general explanation of recycling processes in their stewardship plan and annual reports.[11] Such requirements have been omitted from draft legislation under consideration in some states, such as Connecticut. The bill does not even necessarily require battery recycling, instead requiring producers to "ensure that the components of the discarded covered batteries, to the extent economically and technically feasible, are recycled or otherwise managed responsibly."[12] Recycling is defined based upon existing state laws and includes no specific targets for recycling efficiency or provisions for ensuring recycling efficiencies are reported consistently.

This is troublesome in the context of the United States for two reasons. First, if the intent of the model bill is to ensure a consistent approach to battery management across the U.S. states, it is important that a standard methodology is adopted to assess recycling efficiency and other performance metrics. Second, the industry sponsored LCA of the management of single-use batteries demonstrated that for EPR programs to yield a net environmental benefit, they require deliberate planning (Olivetti et al. 2011). Without clearer standards for recycling efficiency and reporting, stewardship organizations have little incentive to undertake such deliberate planning. Thus, although the model bill is well positioned to ensure producer responsibility for the collection of single-use batteries, it is poorly positioned to ensure a harmonious approach to how waste batteries are recycled, how those activities are reported, and assessment of the environmental consequences of such activities.

Discussion and Recommendations

These approaches to regulating EOL batteries represent important steps toward diverting batteries from the municipal waste stream and promoting extended producer responsibility. Each policy also includes provisions that further the goal of achieving net environmental benefits (see table 1). The EU has progressive standards for collection rates, recycling efficiency, and best-available technology. Ontario excludes downcycling from its definition of recycling. The U.S. model bill has collection incentives and private action enforcement measures. But each policy also highlights the challenges that remain to ensure that extended producer responsibility policies yield a net environmental benefit. Even though each policy adopts collection targets, whether in statute (EU) or in practice (Ontario), they lack enforcement and accountability measures. This is especially true in the Canadian provinces, where battery legislation consists of broad statutes that are not conducive to enforcement. Further, the approaches do not contain clear provisions that ensure the *quality* of recycling. In fact, British Columbia's recycling regulation and the U.S. model bill do not directly define recycling at all. Other recycling efficiency standards, such as BATs in the EU legislation, theoretically guarantee higher recycling efficiency, but are ineffective in practice because they are not implemented. Thus, the metrics included in existing battery policies are insufficient to achieve a net environmental benefit, as is evident in the case of collection rates and recycling efficiency, which are discussed below.

Collection rates focus only on how many batteries are collected, not how they are collected. Stewardship organizations, however, have explored a variety of approaches to collecting batteries—including dropoffs at municipal facilities, retail collection points, return-by-mail programs, and co-collection with curbside recycling—which result in different environmental burdens (Olivetti et al. 2011; Masanet and Horvath 2012). In a California-based study, models indicated that the least preferable options included mail-in and municipal drop-off programs. Retail dropoff was better, given that consumers usually do not make dedicated trips to retail locations for recycling purposes (whereas they do make such trips to municipal drop-off sites). But, by far, the least impactful collection method, as measured by energy consumption or GHG emissions, was co-collection with curbside recycling (Masanet and Horvath 2012). For instance, in the two most favorable scenarios modeled in Olivetti and colleagues, the determining factor in whether the scenario yielded a net environmental benefit (measured in terms of energy savings or GHG reductions) was the mode of collection: municipal dropoff added 3 to 4 megajoules (MJ) and 0.2 to 0.25 kilograms carbon dioxide (kg CO_2) per kilogram of batteries processed to the environmental burden at EOL compared to retail collection. However, as is consistent with most EPR legislation, which is designed to allow stewardship organizations flexibility in meeting collection targets, current battery policies do not provide any specific guidance regarding modes of collection.

Similarly, recycling efficiency standards—which measure the materials recovered—also fail to address the environmental consequences of transporting batteries from collection points to recycling facilities. In North America, long-distance transport is often necessary, given that only four facilities currently recycle single-use alkaline batteries (PSI 2014, 12). For instance, in British Columbia, Call2Recycle ships batteries approximately 4,000 kilometers to Pennsylvania for processing, which adds 3 MJ and 0.2 kg CO_2 per kilogram batteries processed to the environmental burden at EOL. But, even if battery recyclers were more widespread, stewardship organizations might still choose to ship batteries long distances to recyclers with high recycling efficiency, when a nearby facility, operating at lower levels of recycling efficiency, might be preferable in terms of maximizing the environmental benefit. Careful assessment is necessary to evaluate these scenarios. Unlike battery policies in Canada or the United States, the 2006 EU Directive acknowledges this risk: "collection and recycling schemes should be optimized, in particular in order to minimize the cost and the negative environmental impact of transport."[13] However, the EU excluded transportation costs from the protocol for calculating recycling efficiency, nor is such assessment require elsewhere under the Battery Directive.

To address these challenges and ensure that EPR policies for batteries account for the environmental consequences of managing single-use batteries at EOL, we make two recommendations. First, EPR policies should adopt reporting requirements that address the net environmental benefits associated with all aspects of EOL management of spent batteries. Second, EPR policies should set recycling standards that encourage recovery of high-grade secondary materials, potentially suitable for reuse in new batteries or other products, if doing so yields a net environmental benefit.

Most existing battery legislation includes specific requirements for annual reporting and periodic audits. In this context, policies should require that reports include an explanation of how "collection and recycling schemes have been optimised in order to minimise the cost and negative environmental impact of transport," as specified under the Battery Directive. Such requirements would be consistent with other reporting requirements regarding the location of collection sites, the manner in which batteries are sorted, consolidated, and processed, and the financing of stewardship plans. We also recommend that more comprehensive audits of stewardship plans—which some policies already require on a periodic basis—include specific language requiring that the scope of such audits include the environmental consequences of the plan. As LCAs of battery recycling have demonstrated, such activities involve significant environmental burdens and benefits, which must be assessed in the context of particular programs (Olivetti et al. 2011; Nelen et al. 2013). Such reporting requirements will provide incentives for product stewardship organizations and recyclers to ensure that programs yield a net environmental benefit, rather than just achieving collection rates or recycling efficiencies.

EPR policies also need to require recycling using BATs. Currently, the recycling efficiency standards included in European and Canadian policies do not distinguish between whether recovered zinc is used as an agricultural fertilizer or used as a feedstock for metal production. As a result, most battery materials are recycled for lower-quality metal-based products, agricultural nutrients, or slag. It is, however, possible to recover higher-quality materials from spent alkaline batteries, which could reduce the environmental consequences of sourcing raw materials for batteries or other products. In the case of alkaline batteries, raw material extraction and processing accounts for the bulk of the environmental burden of single-use batteries across the life cycle (Olivetti et al. 2011; Smith et al. 2014). Energizer has recently begun selling the first alkaline battery containing recycled electrolytic materials, demonstrating the technical feasibility of closing the loop on alkaline batteries (Energizer 2015). But if recyclers are going to invest the financial capital necessary to develop and build facilities capable of recovering high-quality manganese dioxide, zinc, and steel from spent batteries at volumes appropriate for being qualified for use by battery manufacturers or other end uses, EPR policies must include standards that require or reward the recovery of high-quality secondary materials. And such BATs also must be informed by LCAs that demonstrate that such recycling systems achieve a net environmental benefit (Nelen et al. 2013).

Without such changes in existing and proposed EPR policy, it is unclear that product stewardship programs that include single-use batteries in North America will result in a net reduction in environmental impact for categories such as CED, global warming potential, and resources damage. In short, the energy necessary to collect and process batteries may exceed the environmental benefits of recovering secondary materials, even when those materials are reused in batteries or other high-quality applications. It is possible that these costs will diminish as more recycling facilities are constructed, the volumes of waste batteries increase, technology improves, or these benefits are considered in the broader context of increasing the recovery of all types of batteries (including rechargeable batteries). But the best way to advance that goal is to ensure that the life cycle consequences of managing batteries at EOL are considered, EPR policies are structured to minimize the burdens of collecting and processing single-use batteries, and recyclers are incentivized to promote high-quality and high-efficiency materials recovery. Such analyses represents a significant burden for product stewardship organizations and the recycling companies with whom they partner, but the burden is not dissimilar to the financial auditing and reporting requirements already incorporated into or proposed for some extended producer programs for single-use batteries.

Conclusion

Spent single-use alkaline batteries comprise a marginal waste stream with high collection costs and low-value secondary materials. Given these challenges, EOL battery management for single-use batteries can easily produce net negative environmental impacts (Olivetti et al. 2011; Nelen et al. 2013). To ensure that EPR legislation is achieving its environmental goals, high performance standards for each stage of EOL waste management are necessary. Such standards, however, are underdeveloped in most existing EPR policy. But for batteries and similar product waste streams, such as some textiles, glass, and e-waste, which have high collection costs, complex material compositions, and contain materials of varying secondary value, the economic and environmental challenges are similar. For such waste streams, EPR policies need to give more attention to the net environmental benefits of EOL management.

In order to be effective, such performance requirements must also be enforced. Thus far, EPR programs in the EU, Canada, and United States have not adequately implemented their environmental performance provisions. The EU, for example, has not implemented BATs in practice. British Columbia and Ontario cannot enforce the collection and recycling efficiency targets in their PSPs. And the U.S. model bill has enforcement structures for achieving collection rates, yet it does not possess similar measures for other performance requirements. Some of these excluded performance requirements include collection methods, transport costs, and BATs. Thus, although EPR programs for single-use batteries have succeeded in creating collection and recycling infrastructure, more work needs to be done to ensure the quality of this infrastructure.

If EPR programs do not adopt and implement sufficient environmental performance standards, they run the risk of becoming a form of what Samantha MacBride has described as “busy-ness” (MacBride 2012). MacBride argues that certain initiatives, such as curbside glass recycling, generate a great amount of activity and thus appear to achieve real environmental gains, but these recycling programs fail to address the founding concerns of the programs. At the same time, they divert attention away from more beneficial programs (glass bottle refill programs). Similarly, without substantive performance requirements that examine the entire scope of EOL environmental impacts, EPR programs for single-use batteries may not produce environmental benefits, as originally intended. Instead, the programs may serve to distract consumers, producers, and policy makers from other policy alternatives, such as banning single-use batteries or promoting rechargeable batteries. With proper standards for EOL management, however, EPR for single-use batteries could become an effective tool to reduce the environmental impact of batteries across the entire life cycle, which is the ultimate goal of all EPR policies.

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Notes

1. These figures are based on the authors’ calculations using data reported in Olivetti and colleagues (2011).
2. “Universal wastes” are common hazardous wastes that can be transported, handled, and recycled by individuals and businesses, instead of under stricter hazardous waste policies. The goal of such policies is to keep universal wastes out of landfills, and to make it easier to direct them into safe disposal streams.
3. Council Directive 2006/66/EC on batteries and accumulators and waste batteries and accumulators [2006] OJ L 266, 26/09/2006 (Battery Directive).
4. The EU’s Best Available Techniques Reference Documents (BREFS), which outline BATs for a large number of EU industries and waste products, do not exist for batteries.
5. 2008 amendment to the 2004 Recycling Regulation, BC Reg 449/2004. The Recycling Regulation is under the authority of the Environmental Management Act, SBC 2003, c 53.
6. The 2006 Municipal Hazardous or Special Waste (MHSW) Amendment 542/06. The amendment is under the authority of the Ontario’s 2002 Waste Diversion Act, SO 2002, c 6.
7. State of Vermont, An act relating to establishing a product stewardship program for primary batteries. Vt. Stat. Ann. tit. 10, § 168.
8. Model Consumer Battery Stewardship Act, proposed 11 June 2014. Introduced by the Corporation for Battery Recycling (CBR), the National Electrical Manufacturers Association (NEMA), the Rechargeable Battery Association (PRBA), and Call2Recycle. www.call2recycle.org/wp-content/uploads/Model_All_Battery_Bill.pdf. Accessed 15 April 2015.
9. Model Consumer Battery Stewardship Act, Section 15.
10. Model Consumer Battery Stewardship Act, Section 16.
11. Model Consumer Battery Stewardship Act, Section 11.
12. Model Consumer Battery Stewardship Act, Section 4.
13. Council Directive 2006/66/EC on batteries and accumulators and waste batteries and accumulators [2006] OJ L 266, 26/09/2006 (Battery Directive).

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