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Updating the Electronics Cycle: Improving US E-Waste Management Practices

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Introduction

The last fifty years have seen an increase in the amount of electronic waste. Electronics have become more accessible for consumers as manufacturing costs have fallen. The market has also become more globally interconnected, with people in developing countries enjoying increased purchasing power and greater access to electronic devices (Balde et al., 2017). As a result, electronics have become a major part of the global economy, with production and consumption on the rise.

Mobile phone production is one such example of rapid growth. From 2015 to 2020, the number of smartphones produced has grown from 1.30 billion units to 1.50 billion (Statista, 2020). Meanwhile, electronics that are replaced by new products are either stored at home, sent to landfills, or recycled. The products that are discarded with no intention of reuse become electronic waste, or e-waste (Kumar et al., 2017).

The short lifespans of electronics also contribute to the growing purchase and disposal rates. Laptops last an average of five years before disposal, and flat panel TVs average seven years (Balde et al., 2017). Smartphones average four years due to wear from daily use and complicated hardware. In addition, software updates, online advancements, and yearly new versions (i.e. iPhone, Samsung Galaxy series) quickly render phones obsolete (Kumar et al., 2017). Consumers are then pressured to replace phones and other electronics for new ones, adding to the e-waste stream.

The pace of consumption has led to an e-waste stream that grows every year. The global amount of e-waste generated grew from 41.8 million metric tons in 2014 to 44.7 million metric tons in 2016 (Kumar et al., 2017; Arduin et al., 2019). In 2018, an estimated 50 million metric tons were produced (Kumar et al., 2017). This growth rate of 3-5% makes the e-waste stream one of the fastest growing sources of waste, triple that of other sources (Kumar et al., 2017).

While many national and local governments have devised ways for consumers to dispose of waste, e-waste disposal methods can be unclear, confusing, or inconvenient. People may then mix e-waste with other non-recyclable waste, whether deliberately or out of ignorance. Over 3.5 million metric tons of the US' e-waste have ended up in landfills as a result of this inconvenience (Powell and Chertow, 2018).

Instead of researching ways to recycle e-waste, consumers may also store their products at home, saving them in case their current equipment breaks (Nowakowski, 2019). Though not dangerous intact, stockpiled electronics increase the time between items falling out of use and being repurposed or refurbished. Due to the combination of landfilling and stockpiling, only 17% of e-waste in the US is recycled annually (Balde et al., 2017).

The generation of e-waste has multiple consequences. Electronics contain numerous chemicals with health and environmental risks. Although copper and aluminum are ubiquitous, other metals have become more or less common in electronics. Lead has become less common in products such as phones, owing to increased regulation and technological innovation (Chen et al., 2018). Meanwhile, ecotoxicity impacts from copper, nickel, and zinc in smartphones has risen (Singh et al., 2019). While the concentrations and exposure potential of hazardous substances in household electronics have a low risk during use, those risks become larger at e-waste processing and disposal sites. Toxin concentrations at these sites are high enough for adverse effects to be observed in people, and harmful byproducts can move off-site and harm residents and wildlife (Amankwaa et al., 2017).

The US needs a national policy to address the adverse impacts of the growing e-waste stream. The three main motivations for this policy are: pollution control, human health, and climate change mitigation. By identifying and implementing the most effective methods of e-waste collection and processing, we reduce the amount of e-waste sent to landfills. Adopting effective policy reduces the risk of chemicals leaching into the environment and increases the amount of reusable material recovered from e-waste.

Recovering more material can also decrease the amount of raw material that needs to be mined and manufactured for components. Reusing existing material is more economically and environmentally efficient, saving time, effort, money, and natural resources.

The equipment required for mining and creating parts for products come with their own carbon footprints. Increasing the longevity of resources will be an important part of reducing global greenhouse gas emissions. Avoiding the first source of emissions by not mining will prevent emissions further along the product's lifespan.

Lastly, improving e-waste management benefits people working in the processing and disposal sectors. These improvements include closer oversight of e-waste transport, to prevent e-waste from reaching facilities do not safely or sustainably handle e-waste. Preventing exports of hazardous material reduces the impact of illegal recycling facilities on the health of their workers and nearby residents. For example, limiting the amount of waste that reaches a facility that fails to capture acid used in metal recovery can improve the health of those who would otherwise be impacted by exposure to the chemicals via soil or water.

With ~6.3 million tons generated in 2016, The United States is one of the largest producers of e-waste (Balde et al., 2017). However, the amount of research on the US e-waste management sector is less than other world regions (Figure 1). This paper aims to help close that research gap.

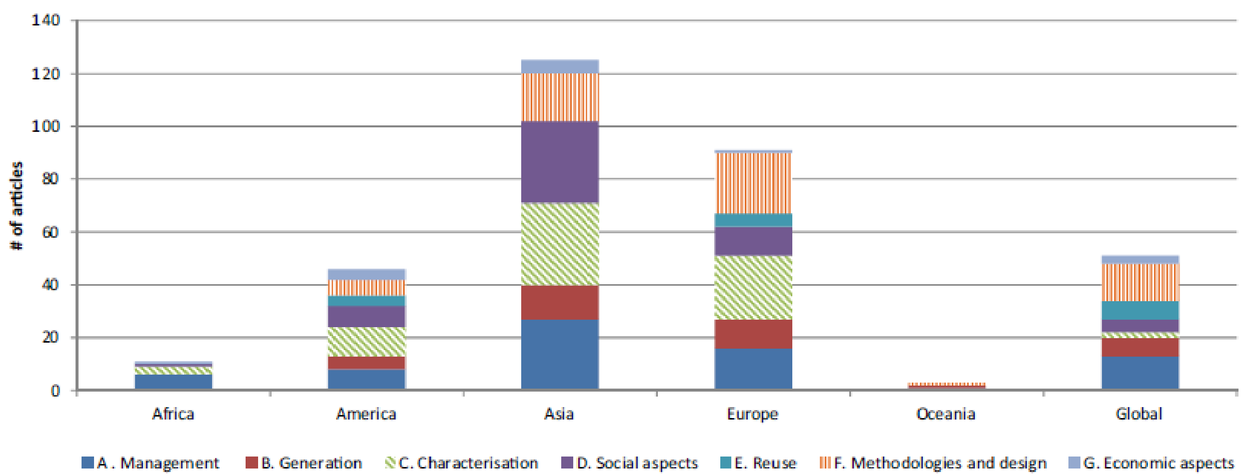

Figure 1. Number of research articles on e-waste by region through 2014. Regional bodies of research are further organized by the aspect of e-waste studied. (Pérez-Belis et al., 2015).

In this paper, I ask how e-waste management practices in the United States can be improved to reduce environmental impacts. I explore the toxicological and environmental risks from e-waste mismanagement. I then compare other e-waste management strategies and policies, and identify ways to integrate them in the US.

This paper makes comparisons of case studies of various e-waste management policies and practices. These cases range from instances in US states and municipalities to those in EU member nations. While successful programs (those that increase recycling or reuse) illustrate applicable practices, programs that fail or underperform can also serve as valuable lessons. Contributing factors to those failures may have parallels to current conditions in the US, signaling areas of management to approach differently.

A comparative analysis is done to show structural similarities and differences between countries and regions based on the principles, priorities, and goals they set when making decisions about e-waste. The behavioral, economic, geographical, and political drivers behind e-waste management vary between nations and states. However, similarities can be utilized to bridge those gaps and make integration feasible.

The case studies and comparative analyses can be synthesized using a common metric between them. This paper focuses on the amount of waste reduced for an e-waste management scheme, so the metric for analysis is total tonnage of e-waste reduced. This metric is used to quantify the effectiveness of policies. Where tonnage of e-waste is not used, public perception surveys of e-waste management programs are also used to predict the success of a program in a similar US region, using approval as a metric.

This paper begins by explaining the current state of e-waste management in developed countries, and the effects the e-waste industry has on workers and residents in developing countries. It then explores case studies of e-waste management programs designed to meet broader policy goals. Those programs are then discussed in the context of US e-waste policy, and recommendations for improving national e-waste management are made.

Analysis of studies from three EU nations and three US states finds positive impacts of extended producer responsibility principles on collection and recycling rates and local economies. While producer responsibility organizations increase recycling through competition and innovation, collection in rural areas is difficult. The analyzed e-waste programs did not significantly increase reuse of old electronics due to the cost of preparation and transportation. Survey data indicates a consumer's level of environmental involvement and other socioeconomic factors can be used to predict their willingness to fund recycling schemes.

Finally, this study recommends a national adoption of EPR for electronics producers, regional producer responsibility organizations, and increased contingent valuation surveys to determine appropriate financing for statewide and municipal programs. National expansions to education about recycling options, device trade-ins at retailers, and material flows analyses to predict e-waste component quantities are also recommended.

Background

1. E-Waste and Health

Electronics contain metals, plastics, glass, and other compounds that are necessary for the devices to work. Materials in e-waste can be recovered for use in new products, reducing the amount of new resources needed. Recovery reduces the amount of waste generated, and saves money and energy by reducing the amount of new material that must be mined or produced. For example, waste printed circuit boards (PCBs) contain 20% copper, a concentration 10 times higher than metallic ore (Ilankoon et al., 2018). The raw material value of the global e-waste supply is an estimated $55 billion yearly (Ilankoon et al., 2018). However, the effort required and the toxicity of many chemical components renders full recovery unfeasible. As a result, the value of the global e-waste stream is about $20.5-25 billion, at about $500 per metric ton (Ilankoon et al., 2018).

Multiple hazardous substances are present in e-waste, with each kind of device containing different substances. Among the substances in household e-waste are: lead, cadmium, mercury, hexavalent chromium, brominated flame retardants, and polyvinyl chloride (PVC) (Ilankoon et al., 2018). Table 2 shows hazardous substances found in e-waste, their sources in the e-waste stream, and the health and environmental effects of each substance.

Lead is one of the most extensively researched hazardous metals found in e-waste. Though its use has been restricted in recent years, lead remains one of the primary toxicants in the e-waste stream due to its presence in CRT glass (Chen et al., 2011). CRT TVs contain about 1.5-3 kg lead, and monitors contain about 0.5 kg (Chen et al., 2011; Ilankoon et al., 2018). Flat screen displays with lower lead content (<1 kg) have replaced CRTs in recent years, but CRTs continue to enter the e-waste stream as they break down and are discarded (Ilankoon, 2018). In 2014, lead glass from CRTs made up about 2.2 million metric tons of the 41.8 million metric tons of total e-waste generated (Ilankoon et al., 2018).

Table 2. Hazardous elements and compounds in e-waste, along with sources within e-waste stream and associated health effects (Ilankoon et al., 2018).

Metal	Sources	Effects
Lead (Pb)	CRTs (4–22% of Pb), television sets, PC monitors, batteries, PCBs, light bulbs, lamps	Can cause intellectual impairment in children, can damage the nervous, blood and reproductive systems in adults, neurobehavioral development of children, anemia, kidney damage, chronic neurotoxicity
Cadmium (Cd)	Rechargeable computer batteries, contacts and switches, older CRTs, PCBs, Ni-Cd batteries, infrared detectors, semi-conductor chips, ink or toner photocopying machines, CRTs, mobile phones	Highly toxic and bioaccumulation in the environment, affecting the kidneys and bones, possibly reproductive damage and lung emphysema
Mercury (Hg)	Lighting devices for flat screen displays, CRTs, PCBs, thermostats, sensors, monitors, cold cathode fluorescent lamps (1–2 g per device)	Can damage the brain and central nervous system, neurobehavioral development of children (methylmercury), anaemia, kidney damage, chronic neurotoxicity
Chromium or hexavalent Chromium (Cr) compounds	Production of metal housings (anti-corrosion coatings), data tapes, floppy disks	Highly toxic, human carcinogens, impacts on neonates, reproductive and endocrine functions
Nickel (Ni)	Ni-Cd batteries, Electron gun in CRTs	Increased risk of lung cancer, cardiovascular disease, neurological deficits, developmental deficits in childhood, and high blood pressure
POPs including brominated flame retardants (Penta-, Octa-, Deca-BDE)	Used in circuit boards (fire retardants for electronic equipment), plastic casings of computers, cables, as dielectric fluids in capacitors and transformers, lubricants and coolants in generators, fluorescent lighting, ceiling fans, dishwashers, electric motors, components such as connectors, mobile phones	Bioaccumulation in the environment (very resistant to break down), neurotoxicity, long-term exposure can lead to impaired learning and memory functions, interfere with thyroid and oestrogen hormone systems, exposure in the womb has been linked to behavioural problems.
Lithium (Li)	Li-batteries	Can cause nausea, diarrhea, dizziness, muscle weakness, fatigue, and a dazed feeling
Barium (Ba)	CRTs (2–9% Ba), fluorescent lamps	Low blood potassium, cardiac arrhythmias, respiratory failure, gastrointestinal dysfunction, paralysis, muscle twitching, and elevated blood pressure
Zinc (Zn)	CRTs, metal coatings, batteries	Increased risk of copper deficiency (anemia, neurological abnormalities)
PVC	For insulation on wires and cables	Incineration of PVC produces chlorinated dioxins and furans, which are highly persistent in the environment and toxic even in very low concentrations.
Beryllium (Be)	Power supply boxes, computers, x-ray machines, ceramic components of electronics	Affect organs such as the liver, kidneys, heart, nervous system, and the lymphatic system, may develop beryllium sensitization or chronic beryllium disease

Lead affects neurodevelopment in children, causing behavioral disturbances, attention deficits, and decreased cognitive function (Chen et al., 2011). Blood lead concentrations have been associated with IQ deficits (Lanphear et al., 2005). While consumers are unlikely to develop health effects from lead in e-waste, industry workers and children exposed to e-waste are at risk of exposure, increasing the possibility of developmental effects (Chen et al., 2011).

Mercury is found in low concentrations (<1-2 g per device) in flat screens, cold cathode fluorescent lamps, PCBs, and cell phones (Chen et al., 2011). While individual concentrations are low, the processing of millions of devices at informal recycling sites releases mercury vapor. This inorganic mercury may enter water bodies, where bacteria react with it to form organic methylmercury (MeHg) (Chen et al., 2011). MeHg bioaccumulates in fish, exposing people who eat those fish to MeHg (Chen et al., 2011; Ilankoon, 2018). Inorganic mercury has been associated with brain and nervous system damage, and MeHg has been found to hinder prenatal neurodevelopment (Chen et al., 2011; Ilankoon et al., 2018).

Hexavalent chromium (chromium (VI)) is found in metal housings of electronics to protect against corrosion (Chen et al., 2011). In addition to being a known carcinogen when inhaled, it can also cause DNA damage in fetuses and newborns (Chen et al.,2011). Proximity to e-waste recycling can raise blood chromium (VI) levels; one study in Guiyu, China found mean blood chromium (VI) levels of 99.90 μg/L in the fetuses of women who worked at e-waste recycling sites (Li et al., 2008). By comparison, workers in Italy with occupational exposure to chromium (VI) had blood levels of 6.9 μg/L (Chen et al., 2011).

E-waste processing produces toxicants besides those originally in the devices. Burning cables to recover copper wires produces polychlorinated dibenzo-p-dioxins (PCDDs) and polycyclic aromatic hydrocarbons (PAHs), which cause nerve damage and lead to cancers (Ilankoon et al., 2018). Prenatal PAH exposure has also been associated with IQ deficits (Chen et al., 2011).

In addition to human toxicity, contaminants in e-waste impact plants and other wildlife. Metals such as cadmium, chromium, copper, iron, and lead can bioaccumulate in plants; excess concentrations of these metals can lead to reductions in growth, seed germination, nutrient content, and biomass (Hira et al., 2018). Areas near e-waste processing and disposal sites risk soil degradation and ecosystem damage (Hira et al., 2018).

While e-waste processing and material recovery facilities in developed countries have regulatory oversight to limit pollution, noncompliant facilities in developing countries may operate with little risk of enforcement. The informal status of these facilities gives them more opportunities to release pollutants from e-waste into the surrounding environment.

The consequences of unmitigated e-waste pollution can be seen in Agbogbloshie, an e-waste dumping site covering 31.3 hectares in the Ghanaian capital of Accra (Amankwaa et al., 2017). It is one of the largest e-waste scrapyards in Africa, but also contains vegetable markets, residences, recreational areas, and a mosque. The site serves as a source of income, technological experience, and access to new technology for around 40,000 people (Sovacool, 2019). However, it also exposes workers, residents, and the environment to hazardous chemicals. The volume of waste processed at the site has made it the focus of toxicity studies, NGO action campaigns, and restoration efforts for e-waste (Amankwaa et al., 2017; Moeckel et al., 2020; Sovacool, 2019).

E-waste at Agbogbloshie is imported from developed nations and delivered from within Ghana. Workers there scavenge the waste, dismantle it to separate its components, and recover materials. Metals such as gold, copper, palladium, tin, and aluminum are recovered via acid washing and burning in open air pits (Sovacool, 2019). Waste byproducts enter the environment through multiple pathways. Acids, lead, copper, and mercury leach from e-waste components into soil and water (Kumar et al., 2017). The burning of plastic cables and circuit boards releases dioxins into the air and deposit on topsoil (Kumar et al., 2017). Workers involved in recycling have the highest rates of ingestion and inhalation of pollutants due to their close and constant proximity to waste. However, the release of pollutants also leads to uptake by fish, wildlife, livestock, plants, and crops, all of which are sources of ingestion exposure to the general population (Sovacool, 2019).

The contamination around Agbogbloshie has resulted in elevated blood lead levels in the workers and residents near the site (Amankwaa et al., 2017). Compared to the CDC's reference level of 5 µg/dL, workers had blood lead levels up to 18.80 µg/dL (Amankwaa et al., 2017). Residents and commuting merchants also showed blood lead levels up to 8.20 µg/dL (Amankwaa et al., 2017). These results are especially concerning because pregnant women and children are also at risk of lead exposure, which hinders development and causes neurological damage. In the absence of safety or disposal procedures, lead poisoning of workers, residents, and children will continue, damaging health and limiting economic advancement (Amankwaa et al., 2017).

2. E-Waste Legislation in the United States

The Resource Conservation and Recovery Act (1976) and the Hazardous and Solid Waste Amendment (1984) make the definitive law for solid waste management in the United States, mandating the reduction or elimination of hazardous waste and requiring treatment before disposal. While otherwise comprehensive, it does not fully cover e-waste as a hazardous substance, nor are special provisions included for its disposal. Electronics are not wholly ignored—products containing hazardous material are still bound by RCRA rules, federal agencies solely purchase from electronics manufacturers that allow take-back programs, and can only use certain certified recycling companies (Balde et al., 2017). However, household hazardous waste such as e-waste is not federally regulated (Wagner, 2009). Furthermore, the US currently has no federal legislation regarding e-waste, instead allowing states to write their own. (Schumacher and Agbemabtese, 2019).

Using EU definitions, 1.4 million tons of e-waste were collected in the US in 2016, or 22% of the total generated (Balde et al., 2017). However, the low collection rate may be due to a difference in scope between the European Union and the US; the EPA classifies e-waste as waste audio and video equipment, computers, phones, screens, printers, and other computer peripherals. By contrast, the EU's definition under the WEEE Directive (2002) includes air conditioning equipment, small vehicles, medical devices, lighting equipment, and sporting equipment. Using the EPA definition, ~70% of e-waste was collected in the US in 2016 (Balde et al., 2017). The discrepancy in e-waste collection figures between the two systems shows how the definition of scope impacts the interpretation of data.

In the absence of federal guidance, 25 states have passed some legislation to regulate the generation and disposal of e-waste. These laws range from bans on sending CRT TVs to landfills to recycling mandates (Balde et al., 2017). Standards for collection are also state-dependent. Responsibility for collection site placement can be assigned to manufacturers or municipalities; collection sites can be mandatory or optional with financial incentives; collection sites can be prescribed based on city size or not detailed at all (Schumacher and Agbemabtese, 2019).

About 84% of the US population live in states with e-waste recycling programs and legislation (Balde et al., 2017). Figure 2 shows states that have passed e-waste laws. While 25 states have passed laws addressing e-waste, the remaining states have passed no legislation for e-waste management. The lack of regulatory cohesion leaves holes in enforcement of standards. Businesses wishing to avoid scrutiny in one state may move operations to avoid compliance.

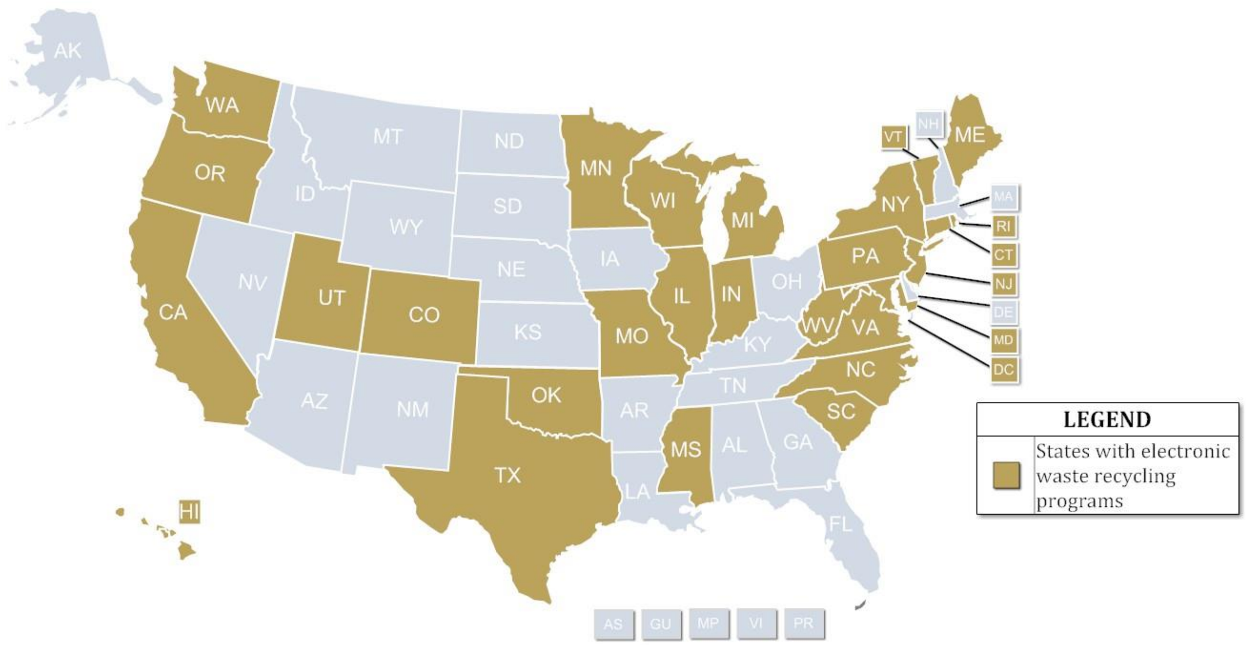

Figure 2. States that have passed e-waste recycling laws (NCSL, 2018).

The Basel Convention (1992) controls the transport of hazardous waste, including e-waste, across national borders. (Li et al., 2015). Though the US signed the treaty, inability to amend the RCRA and HSWA have resulted in it being the only country to have not ratified the Basel Convention. This federal abandonment of the treaty hinders the prevention of export of e-waste to informal recyclers in China, India, Nigeria, Ghana, and other developing countries with lax regulation or weak oversight.

Unregulated export is not an issue unique to the US, with businesses in other developed countries also regularly circumventing Basel Convention rules. An estimated 23% of all domestically generated e-waste in OECD countries is sent to China, India, and Western Africa (Petridis et al., 2020). However, due to unaudited self-reporting and mislabeling of e-waste quantities and destinations, the exact scale of illegal transport is difficult to quantify. (Lee et al., 2017).

3. E-Waste Legislation in the European Union

The EU has two major pieces of legislation governing e-waste management: the Restriction of Hazardous Substances (RoHS) Directive and the Waste Electrical and Electronic Equipment (WEEE) Directive. These two directives aim to improve the safety, efficiency, and scope of e-waste recycling in the EU. Since their passage in 2003, the two directives have undergone further amendments increasing their scope (Balde et al., 2017).

The RoHS Directive regulates the composition of products that are sold within its borders, prohibiting goods containing hazardous materials from entering the market. The Directive ensures that electronics do not contain: lead (excluding lead used in solder), cadmium, mercury, chromium (VI), and polybronimated biphenyl (PBB) and polybrominated diphenyl ether (PBDE) flame retardants. This restriction reduces the environmental impact of electronics before they enter the waste stream (Ongondo et al., 2011). Evidence of manufacturer compliance with the RoHS Directive can be found in toxicity assessments of waste printed circuit boards following its passage. Lead concentrations in circuit boards fell from 26,000 mg/kg in 2004 to 483 mg/kg in 2005 (Chen et al., 2016). Similarly, PAH concentrations fell from 2,000 μg/kg to 361μg/kg, and PBB concentrations fell from 670 μg/kg to 40 μg/kg between 2004-2005 (Chen et al., 2016).

The WEEE Directive is a regulatory framework that aims to reduce the generation and disposal of e-waste within the EU. It utilizes the extended producer responsibility (EPR) principle, requiring electronics manufacturers and importers to implement take-back schemes for consumers' e-waste. The initial version of the WEEE Directive in 2002 covered the collection, transport, storage, disassembly, and material recovery of e-waste. It was amended in 2012 to include new products entering the market, and further promoted reuse and recycling (Cole et al., 2019). Under the new amendment, member nations have a goal of recycling 85% of generated e-waste (Petridis et al., 2020). The amendment also streamlined the EU's categorization of e-waste (Arduin et al., 2019) (Table 3).

Table 3. E-waste categories under WEEE Directive prior to and after 2018. (Arduin et al., 2019).

WEEE categories. From August 12, 2012 to August 14, 2018	WEEE categories. From August 15, 2018
Ia. Large household appliances cold	I. Temperature exchange equipment
IIIa. IT Equipment and telecommunications IVa. Consumer equipment	II. Screens, monitors, and equipment containing screens having a surface greater than 100 cm^2
V. Lighting equipment	III. Lamps
Ib. Large household appliances non-cold	IV. Large equipment (any external dimension more than 50 cm)
II. Small household appliances IVb. Consumer equipment VI. Electrical and electronic tools VII. Toys, leisure and sports VIII. Medical devices IX. Monitoring instruments and control X. Automatic dispensers	V. Small equipment (no external dimension more than 50 cm)
IIIb. IT and telecommunications equipmentVII. Toys, leisure and sports	VI. Small IT and telecommunication equipment (no external dimension more than 50 cm)

The WEEE Directive sets collection targets for the amount of e-waste generated that rise over time. In 2016, member states had to collect at least 45% of the electronic equipment placed on the market, then 65% by 2019 (Balde et al., 2017). Due to the difficulty of preventing illegal exports, however, only about 37% of the EU's e-waste has been captured (Balde et al., 2017).

The WEEE Directive's emphasis on minimizing disposal puts e-waste in line with the EU's waste hierarchy in the Waste Framework Directive (2008) The waste hierarchy places preventing waste production as the top goal of waste management, followed in order by reuse, recycling, and recovery. If none are possible, the Waste Framework Directive aims for disposal that does not impact humans or the environment (Cole et al., 2019) (Figure 3).

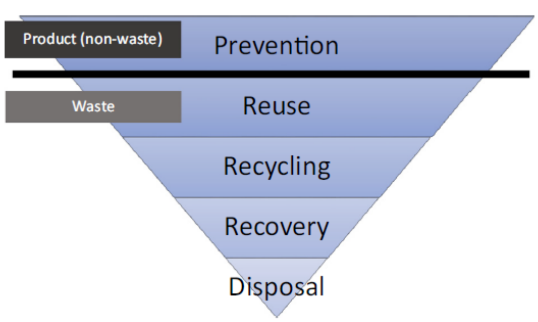

Figure 3. The waste hierarchy outlined in the Waste Framework Directive. (Cole et al., 2019).

Methods

This section reviews e-waste management programs in three EU countries and three US states. Case studies of EPR programs, takeback schemes, recycling fees, and collection methods are examined. The information from these case studies are used to evaluate their success in each region. A successful program is one that resulted in increased e-waste collection and recycling. In addition to case studies of regional programs, a material flows analysis approach to e-waste identifies consumer trends and predicts future compositions of the e-waste stream.

The cases in the UK and California measure success differently from the other case studies. While the other studies use changes to the amount of recycled material as a metric, the UK and California explore public perceptions around recycling programs and finance structures. In these cases, success is measured by the ability for the public to agree to a program or view one positively. The technologies analyzed in Althaf et al. (2019) demonstrate the utility of material flows analysisin e-waste management, which is used in the Recommendations section.

The data is later analyzed and their fit within the US' waste management system is assessed. Programs that can be widely implemented in states and municipalities without public opposition are considered feasible. Feasibility is determined by identifying similarities between structural and demographic aspects of EU nations and the US, and between US states examined and other states. Differences and information gaps are also acknowledged. If a country or state has circumstances that make a program less compatible with the US or other states, potential modifications are explored and suggested.

1. European Union

The WEEE Directive applies to all EU member states, but its application is not uniform. EU members have different legal, economic, cultural, and geographic circumstances that influence their policy infrastructures. Because of this diversity, nations have the opportunity to implement EPR, collection, and public involvement schemes in ways most appropriate for their respective systems. As a result, about 150 WEEE compliance schemes are active in the EU (Yla-Mella et al., 2014).

One example of this flexibility is the diversity of producer responsibility organizations (PROs) that organize the collection, transport, and processing of e-waste. EU states use a variety of PRO ownership systems to meet WEEE Directive goals. Some use one collective, government-run system to collect e-waste from producers (e.g. France, Spain, Sweden). Others use a combination of collective and individual, producer-dependent systems (e.g. Denmark, Finland, United Kingdom). Germany uniquely uses only individual collection systems, with a coordinating body to facilitate collection (Arduin et al., 2019). (Figure 4).

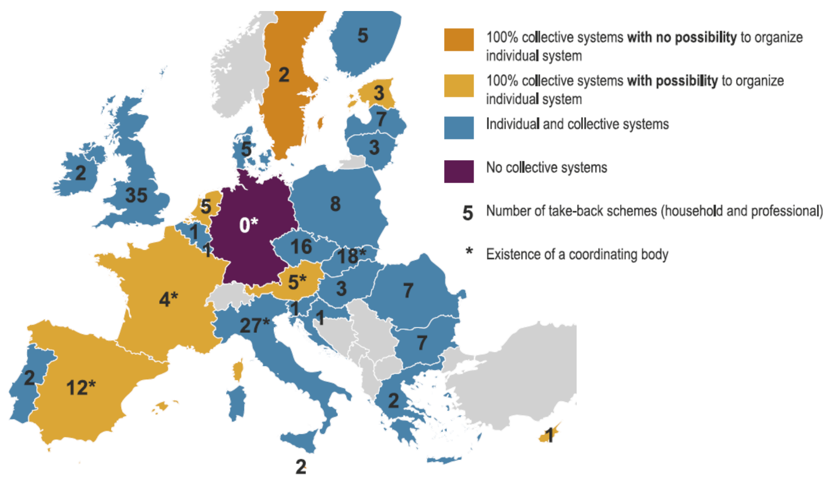

Figure 4. Producer responsibility organizations between EU nations. Composition varies between states that choose either entirely government-organized “collective” PROs, business-owned “individual” PROs, or some combination of both. (Arduin et al., 2019).

This example of individual nations autonomously assigning responsibility for e-waste collection resembles US policy structure in which states are able to choose how to meet or exceed federal standards. While the EU sets legislation at a multinational level, its federal structure makes it a suitable model for comparison with US e-waste management. EU standards (e.g., 85% recycling rate among all member states) can be adjusted to fit US law and implemented by states.

1.1 Finland

Finland is the most sparsely populated country in the EU, with a density of 18 inhabitants per sq. km and half of the country's 300 municipalities contain fewer than 6,000 residents (Yla-Mella et al., 2014). Although serving residents in its most remote areas has been challenging, its EPR scheme is one of the EU's most successful. With over 90% of electronics going through the recycling sector. To meet the obligations of the WEEE Directive, the Finnish Waste Act (2004) required that electronics producers organize the reuse, recovery, and disposal of their products. Collection facilities were established across the country, and retailers and consumers were given information on how to dispose of these products. Producers also reported on the amount of electronics entering the market, as well as e-waste management metrics such as amount collected, recycled, and exported.

To facilitate collection, Finland uses a combination of individual and centralized PROs. About 120 companies manage their own e-waste streams, primarily for business-to business purposes. In addition, five centralized PROs facilitate collection schemes for over 1,000 companies. Using these PROs, customers can dispose of e-waste via dropoff programs (designated collection points in public spaces or at retailers) or by mailing them to collection facilities. (Yla-Mella, 2014). Waste devices can be dropped off at retailers in exchange for a new, similar product, or in the case of small equipment (<25 cm), at no charge. 450 collection bins have been installed in 277 municipalities, exceeding the Finnish Waste Act's minimum of 340 bins in 235 municipalities (Yla-Mella et al., 2014).

Finland's EPR program has made it exceed every one of its target collection and recycling goals (Table 4). ~87% of generated e-waste was treated in the country, and ~13% was treated in another EU member state, with 0.2% reported to have been shipped outside the EU. 88.5% of the nation's e-waste was collected, and 3.1% was incinerated for energy (Yla-Mella et al., 2014).

Table 4. Finland's e-waste stream by metric tonnage and weight percentage as of 2010. Recovery and reuse percentages are compared to target goals. (Yla-Mella et al., 2014).

	Categories	Amount [tonnes]	Portion [w%]	Actual recovery/target [%]	Actual re-use and recycling/target [%]
1	Large household appliances	27,698	54.5	93/80	88/75
2	Small household appliances	1320	2.6	84/70	82/50
3	IT and telecom equipment	8034	15.8	92/75	92/65
4	Consumer electronics	12,117	23.8	90/75	88/65
5	Lightning equipment	961	1.9	91/70	86/50
6	Electrical and electronic tools	276	0.5	94/70	98/50
7	Toys, leisure and sports devices	99	0.2	84/70	82/50
8	Medical devices	53	0.1	75/–	75/–
9	Monitoring and control instruments	119	0.2	78/70	76/50
10	Automatic dispensers	189	0.4	98/80	78/75
	Total/average	50,867	100.0	91.5	88.5

However, issues with collection remain. While recycling in Finland is high, reuse is not prioritized. Reusable devices are not separated from unusable ones, and e-waste collectors handle both without regard for preserving the devices. Yla-Mella et al. (2014) propose that the reuse rate could be improved via separation of reusable devices at dropoff and establishment of a device testing and refurbishing system.

The national e-waste collection network is also challenged by dropoff behavior and distances between collection points. Rural households are more likely to make larger, single deposits of e-waste at collection points than urban households that make multiple, smaller deposits over the same period. Collected volumes in rural areas may then be inconsistent (Yla-Mella et al., 2014).

Furthermore, collection in rural areas requires longer distances between bins, which may be less cost effective depending on the material in each bin. PROs can recoup more money by traveling to a bin filled with equipment containing valuable materials (e.g. usable computers, smartphones with recoverable metals) than to a bin containing low-value equipment (e.g. appliances, lamps). (Yla-Mella et al., 2014). The difference in revenue discourages travel to rural collection points, where the e-waste recovered may not be worth the travel costs. Yla-Mella et al. (2014) recommend decentralization of collection systems to increase collection efficiency.

Not all producers in PROs comply with the WEEE Directive, and free riders may join without taking any equipment from customers. Some have discarded waste via channels for consumers (retailers, dropoff events) at no charge (Yla-Mella et al., 2014). To counter free ridership, the e-waste inspection authority was given the power to impose financial penalties on producers that do not comply with the Directive (Yla-Mella et al., 2014).

1.2 United Kingdom

While the WEEE Directive has reduced the volume of e-waste sent to landfill by promoting processes further up the waste hierarchy, perceptions among those involved with the e-waste industry are mixed. Cole et al., (2019) interviewed thirty UK-based professionals in the e-waste sector to evaluate the effect of the WEEE Directive on the UK's waste management efforts. Organizations represented included academics, e-waste processors, manufacturers, compliance specialists, recycling companies, and governments. Interviewees noted successes within the UK's e-waste management strategy and identified areas for improvement.

Regarding the waste hierarchy, the interviewees felt that the UK was successfully promoting recycling and recovery. This sentiment is supported by a national recycling rate of 75.7% in 2015 (Clarke et al., 2019). Respondents noted that more e-waste would go to landfill without the WEEE directive in place. While reuse of electronics is low, e-waste collectors and processors are able to save money by focusing on recycling. When e-waste must be disposed of, some waste management companies focus on incineration instead of landfilling.

However, the interviewees noted multiple shortcomings with the UK's handling of e-waste. The primary complaint was the focus on recycling and material recovery over reuse. Instead of returning discarded electronics to manufacturers for repair, the components are shredded to recover materials such as iron and copper. Rare metals are not recovered due to the difficulty of recovering enough to justify the cost of separation.

Recycling is also prioritized over reuse for financial reasons; while large equipment such as refrigerators and washing machines have a secondhand market in the UK, smaller equipment is more complicated to prepare for reuse. Preparation for reuse requires securing individual devices to ensure they do not break during transport, and repair may require skills specific to a device beyond cleaning and safety testing. Reuse is more time and space intensive for recycling companies, so companies avoid the extra costs by recycling.

Consumer ignorance was also identified as a reason for continued landfilling of e-waste. Small devices such as phones continue to be mixed with non-recyclable waste out of convenience. Interviewees proposed adding incentives to sending devices back to producers, such as exchanges for new purchases, to increase collection.

1.3 Germany

In 2005, Germany implemented its version of EPR, ElektroG. Prior to the law, Germany's e-waste management was facilitated by regional waste management authorities that collected e-waste and sent it to contracted recycling companies. However, households were not obligated to separate e-waste from mixed municipal waste, and small devices were not collected (Walther et al., 2010). ElektroG improved e-waste collection by adding goals for collection, recycling, and recovery, and mandating that producers aid in takeback throughout the country. To ensure national reach of takeback schemes, regional authorities no longer awarded contracts to disassembly companies, but by business-run PROs. (Walther et al., 2010). The Elektro-Altgeraete-Register (EAR) serves as a coordinating body that organizes pickup. EAR assigns pickup of an e-waste category by a PRO in a region, and the PRO outsources pickup and processing to a recycling company (Walther et al., 2010).

While ElektroG increased the efficiency of e-waste recycling, reuse was impacted. Prior to ElektroG, transport of reusable e-waste was done in a "value-conserving" way; products were manually packaged and placed in smaller containers to avoid damage. Disassembly companies could then sell refurbished products back to producers (Walther et al., 2010). The centralization of e-waste collection led to e-waste being separated into categories and stored in 38 m^3 containers. Once full, the containers were sent to recycling facilities, with little regard for the usability of devices inside. As a result, electronics reuse fell from 10% to 3% under ElektroG (Walther et al., 2010).

Boldoczki et al. (2020) critique the claim that reuse is preferable to recycling. The authors performed life cycle assessments of four large appliance categories (washing machines, freezers, refrigerators, microwaves) and four small devices (printers, monitors, desktop computers, laptops), accounting for 68% of the EU's e-waste stream by mass. The environmental impacts of preparation for reuse were weighed against producing new equipment, based on six impact categories: global warming potential, ecotoxicity, carcinogenic toxicity, mineral resource scarcity, water consumption, and cumulative energy demand. If the impacts from reuse are lower than those from production, then reuse was recommended.

The LCAs showed that the large appliances had greater environmental impacts when prepared for reuse. Boldoczki et al. (2020) found that while impacts to toxicity and resource scarcity were lower under preparation for reuse, older appliances at the end of their life cycles were less energy and water efficient than new appliances. The energy and water demands from use of refurbished appliances exceeded the amount saved during production. For example, electric ranges saw impact increases of 23% for global warming potential and cumulative energy demand. Freezers were the most inefficient to reuse, with global warming potential rising by 115%, carcinogenic toxicity by 96%, and cumulative energy demand by 112%. However, the small devices showed environmental impact reductions for every impact category (except for a 2% increase in global warming potential for refurbishing laser printers) (Boldoczki et al., 2020).

2. United States

State and municipal governments have implemented their own policies to reduce e-waste disposal. Using examples from other nations and regions, those governments have adjusted policies to fit in the US framework.

2.1 Maine

Maine was the first state to pass legislation aimed at e-waste disposal in 2004 (Wagner, 2004). Given the state's rural nature, municipalities were unable to offer curbside pickup for e-waste separately from other solid waste. Faced with the options for municipalities to pay for e-waste pickup (i.e., property taxes, recovery fees, disposal fees), households tended to stockpile e-waste (Wagner, 2009).

Maine's e-waste law is a form of EPR with shared responsibility. Producers, consumers, and municipalities share in e-waste management costs (mainly transportation and recycling costs) (Wagner, 2009). Municipalities create and manage collection sites for households to dispose of e-waste. The municipalities then prepare the e-waste for transport to producers, who handle and recycle the equipment. Households pay for the program through property taxes and transportation to collection sites. While the initial law only covered monitors, televisions, and central processing units, the scope was expanded to include video players, computers, printers, and other devices (Maine.gov, 2019). While cell phones and accessories are not included in the law, collection sites often accept them (Maine.gov, 2019).

Prior to the law coming into effect in 2006, about 660 metric tons of e-waste were collected. Three years later, collections had nearly quadrupled; 1,745 metric tons of e-waste were collected in 2006, 2,126 metric tons in 2007, and 2,534 metric tons in 2008 (Wagner, 2009). Part of this was due to the increased convenience of disposal. Municipal end-of-life fees were lowered so that 74% of households paid under $10, with 29% paying nothing (Wagner, 2009).

2.2 Washington

The Seattle metropolitan area faces different challenges to their e-waste management, namely the collection and processing of e-waste for millions of people in one urban area. In 2006, the State of Washington passed legislation mandating the collection, sorting, transport, and recycling of e-waste by 2009. E-Cycle Washington, a statewide EPR program, was established to accomplish this goal. To gauge the effect of the law on e-waste recycling, Leigh et al. (2012) used a modified input/output model to track flows of electronics in the Seattle area's municipal waste sector. Recycled e-waste was assumed to be the primary product and recovered material a secondary product.

Leigh et al. (2012)'s found a 42% increase in the amount of e-waste collected in 2009, from 22 million pounds to 38 million pounds. (Table 5). The increase in e-waste recycling reduced landfilling, and also increased the supply of jobs in the e-waste sector. The Seattle area saw an increase of $13 million in output due to the e-waste sector expansion, and an addition of 118 jobs, 87 of which were in e-waste recycling (Leigh et al., 2012). The economic benefit of and EPR program can serve as an example for other urban areas considering an e-waste management strategy.

Table 5. E-waste processing companies in E-Cycle Washington, with lbs. e-waste processed. (Leigh et al., 2012).

Processor	*E-waste recycled*	*Residuals (nonrecycled)*	*Reused*	*Total*	*County, State*
Ace Metal Company	1,010,066	138,000	0	1,148,066	Snohomish, WA
E Tech Recycling- Hillsboro	49,630	0	0	49,630	Washington, OR
E-Waste LLC-Lynnwood	1,118,965	70,300	1,000	1,190,265	Snohomish, WA
ECS Refining	394,372	0	0	394,372	Santa Clara, CA
EWC Group–Seattle	1,505,800	0	0	1,505,800	King, WA
Electronic Recyclers International–Auburn	1,789,505	0	0	1,789,505	King, WA
IMS Elect Recycling	5,738,389	381,155	0	6,119,544	Clark, WA
Total Reclaim–Seattle	26,313,381	0	38,111	26,351,492	King, WA
Sum	37,920,108	589,455	39,111	38,548,674	—

2.3 California

In 2003, California passed the Electronic Waste Recycling Act (EWRA) to help fund the collection and recycling of e-waste (CalRecycle, 2020). In addition to providing funding for electronics producers, EWRA aims to increase cost-free recycling for consumers, reduce stockpiling and illegal dumping, and limit the amount of hazardous material in devices (CalRecycle, 2020). Under EWRA, the state Department of Toxic Substances Control is required to adopt regulations limiting the allowable concentration of hazardous material in electronic devices, consistent with the EU's RoHS Directive. EWRA mandates an advanced recycling fee (ARF), added to the price of covered electronic devices (e.g. televisions, laptops, monitors) at retail sale, to help finance recycling schemes in the state (Nixon and Saphores, 2007). Currently, California is the only state to mandate an ARF (Schumacher and Agbemabtese, 2019). At implementation, ARFs were between $6-10 for new purchases. The fee peaked at $8-25 in 2009, but currently is between $4-6 (CA Department of Tax and Fee Administration, 2020).

Nixon and Saphores (2017) surveyed California residents to gauge their willingness to pay for ARFs. Mail surveys were randomly sent to 3000 homes between January and April 2004, before EWRA was implemented. Residents were asked about their willingness to pay ARFs of 1%, 5%, 10%, or not at all. Respondents were also asked about their level of involvement in environmental activities such as beach clean-ups, contributions to environmental organizations, and community meeting attendance. To better understand the motivations behind their choices, respondents shared socioeconomic data such as age, distance to drop-off recycling centers, education, and household income. Lastly, respondents answered questions about their environmental priorities, including their opinions on environmental quality in California, whether to prioritize environmental protection over economic growth, and the role of government and business in protecting the environment (Nixon and Saphores, 2007).

A principal components analysis was used to quantify the respondents' environmental behaviors and attitudes as factors with coefficients between 0 and 1. Levels of environmental activity were grouped as PC1, with higher values indicating less involvement. Attitudes on environmental quality and responsibility were grouped as PC2, with higher values indicating more confidence in environmental quality and higher belief that institutions should protect the environment (Nixon and Saphores, 2007).

Over half of respondents indicated a willingness to pay a 1% ARF, with the remainder evenly split between being willing to pay 5% or not willing to pay at all. Examining the factors in PC1 and PC2 further explains the willingness to pay ARFs. Respondents tended not to be environmentally active (PC1 = 0.77) and equally supported environmental protection and economic growth (PC2 = 0.44) (Nixon and Saphores, 2007). Respondents had high values for the belief that government and business should be involved in environmental protection (PC2 = 0.79 and 0.81), indicating high public support for these sectors to be involved in e-waste management (Nixon and Saphores, 2007).

Respondents were on average over 35 years old, college-educated, earned over $40,000 per year and lived within 5 miles of a drop-off recycling center. This kind of respondent was used as a baseline demographic. As PC1 rose, the predicted likelihood of respondents agreeing to a 5% ARF fell. Non-college respondents with low PC1 scores (<0.1) were most likely to support a 5% ARF, with ~55% in support. Adults age 18-35 were more likely than the baseline to support 5% ARFs. Other groups with consistent above-baseline support included those in rural areas and people who believed business had no role in protecting the environment. People who lived over 5 miles from a recycling center and people who believed the government has no role in environmental protection were consistently less likely to support a 5% ARF (Nixon and Saphores, 2007) (Figure 5).

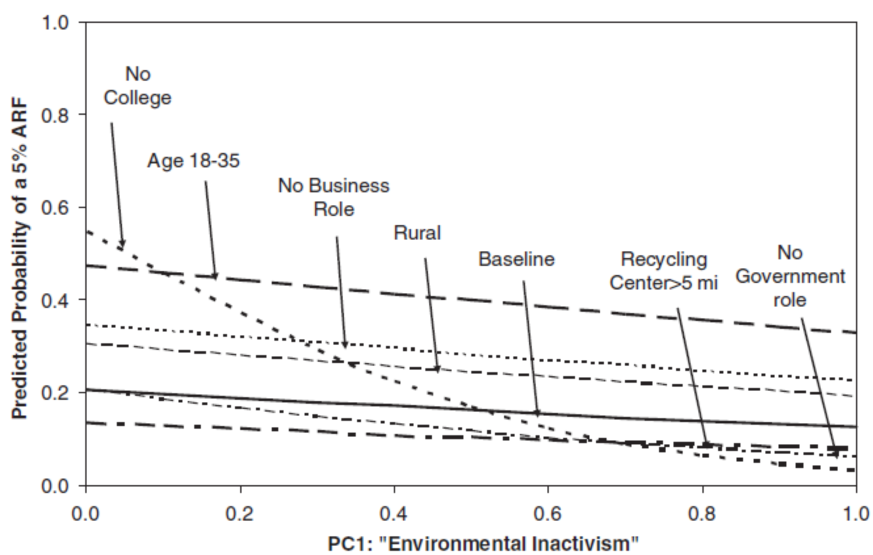

Figure 5. Predicted probability of California residents supporting a 5% ARF over PC1. Level of involvement with environmental activism and volunteering falls as PC1 increases. Other demographics are included and compared to the average respondent, reflected as the baseline. Nixon and Saphores, 2007).

In their analysis, Nixon and Saphores (2007) aim to explain the motivations behind the differences in support for ARFs. Young people were more likely to support higher ARFs due to their familiarity with electronics, and were more likely to understand the consequences of a mismanaged e-waste stream. Respondents in rural areas had fewer opportunities to recycle e-waste, with recycling centers farther away from homes. Because ARFs aim to increase opportunities for recycling, rural respondents were more likely to pay for efforts to increase that convenience and reduce illegal dumping in their communities. People who believed that business does not play a major role in environmental protection may have felt that the government was more appropriate to manage environmental issues, and saw taxes as effective ways to do so (Nixon and Saphores, 2007).

Conversely, people who did not see a major role for government in environmental protection would likely prefer that private entities and individuals fund recycling programs and supported ARFs less often than the baseline. People living over 5 miles from recycling centers were also less willing to support ARFs; they may have less experience with recycling facilities, and may be unwilling to pay for a program they do not expect to use. Households with incomes over $40,000 were less likely to support ARFs, possibly due to living in areas with better environmental quality and facing less pressure to improve their local environment (Nixon and Saphores, 2007)

The results of this survey may have been impacted by the response rate and demographic of the respondents. Of the 3000 surveys sent, 357 (12.4%) were returned. The survey totaled 12 pages, was only available in English, and was distributed in six counties: Alameda, Contra Costa, Kern, Mono, Orange, and San Diego. Two counties (Kern and Mono) were mainly rural, so one third of the respondents were rural. Respondents were on average over 35 years old, college educated, white, earned over $40,000/yr, and homeowners. Because the surveys were given to some of the state’s highest income areas, nearly all of the respondents were college-educated with yearly incomes over $40,000 (Nixon and Saphores, 2007). While the study does have sampling limitations, it does show trends in opinion among the represented groups, predicting environmental valuations with demographic data. The study provides evidence of the need for survey data, as well as lessons for improving data collection.

3. Material Flows Analysis

The material flow analysis (MFA) by Althaf et al. (2019) estimates waste flows of a product based on a function of the product's annual sales, probability of reaching end-of-life in a given year, and its mean and maximum lifespan. This analysis method aims to create an e-waste management system that looks at future flows, instead of focusing on dwindling concentrations of obsolete devices. MFA allows the estimation and forecasting of material flows in the waste stream based on unit sales and longevity. For example, if a brand of solar panel sold well and lasted an average of ten years, one could estimate the amount of panels in the waste stream for the next twenty years, expecting a peak ten years after peak sales (Althaf et al., 2019).

When applied to four emerging technologies—fitness trackers, smart thermostats, drones, and OLED TVs—the authors found that fitness trackers had likely hit peak sales in 2016, and were likely to become more common in e-waste within five years. Meanwhile, drones have failed to reach mainstream adoption in the household market, and their components were unlikely to be a concern in the e-waste stream. Another analysis showed that CRT and LCD TVs were projected to nearly disappear from the e-waste stream by 2030, replaced by LED and OLED (Figure 6) (Althaf et al., 2019).

The shift in e-waste flow composition means that priorities for hazard reduction and resource recovery will need to shift, as well. While lead from CRTs is expected to fall to under 5,000 metric tons in the national e-waste supply by 2025, indium, an element needed for flat panel displays, will become more common as flat panels are discarded. By 2025, indium stock in waste TVs is expected to exceed the demand for indium in new devices by 30%. Similarly, cobalt, necessary for lithium-ion batteries, is projected to become more common in waste than will be in demand for use in new batteries (Althaf et al., 2019). These findings indicate a closed loop on material flows, as materials in waste will be in high enough supply to meet new demand, reducing or eliminating the need to mine for raw material. The authors conclude that prevention of lead contamination from CRTs will become less of a concern than maximizing recovery of indium, cobalt, and other critical elements in the next decade.

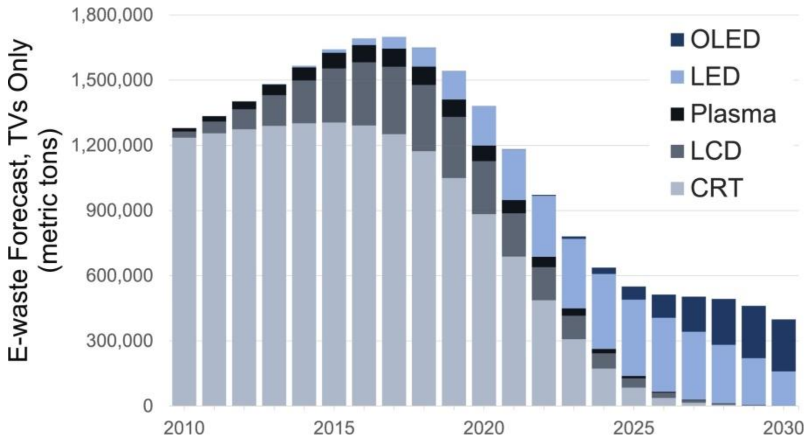

Figure 6. Metric tonnage of TV types in US waste stream over time. (Althaf et al., 2019).

Analysis

In place of federal guidelines, US states have adopted a patchwork of policies to address e-waste disposal. Similar to the EU allowing member states to use individual methods to meet Union collection and recycling standards, the US can use a federal baseline for collection while states aim to meet or exceed those goals. In this section, direct comparisons are made between the countries and states studied and the US. Factors such as population density, system centralization, and public acceptance are compared to determine how well a program can be implemented in the US.

Finland has the lowest population density in the EU, but its EPR law boasts one of the highest e-waste collection and recycling rates. One reason for this high success rate is the variety and availability of e-waste collection methods. Consumers choose between dropping off equipment at retailers or other points of sale, dropping off at public collection points dispersed nationwide, or mailing it to producers. Being able to conveniently return used devices is necessary for EPR schemes to be successful, and an array of options for how to do so ensures that the greatest number of people can take part.

Finland also uses a mixture of collective and individual PROs to organize collections among the 1,000 waste management companies. Consumers use collection schemes organized by collective PROs, while business are able to organize private PROs to process larger loads. The split between consumer and industrial collection is useful due to the ability for businesses to directly coordinate with each other and with recycling companies. In addition, having multiple collective PROs incentivizes competition for market share and allows specialization. For example, a PRO may focus more heavily on collecting from suburban areas or processing smaller devices such as tablets to fill a niche and gain more users.

Utilizing PROs also reduces the government's burden in funding and organizing collection. Because companies are setting up collection points, the government can spend less on maintaining collection points and transporting equipment. Producers, by reporting on metrics such as tonnage of e-waste recycled and exported, provide data that can be used for improving management programs.

State governments would benefit from having a decentralized, business-organized EPR program similar to Finland's. The diversity of drop-off options allows residents to participate regardless of location, and the government can save money on collection by working with competitive PROs that provide the highest quality services at the lowest cost.

However, while setting up collection points increases drop-offs, rural areas still face difficulty with pickup. The farthest collection points from recycling facilities are filled inconsistently, as residents tend to make larger deposits to minimize travel. Regular pickups may not be profitable due to the possibility of not collecting enough material to justify the trip. To address inconsistent filling, a decentralized collection system with more small-scale collection facilities would reduce the distance between a collection point and a facility.

Maine's minimal end-of-life collection fees and focus on property taxes could help fund collection efforts. Consumers in Maine avoided e-waste drop-offs prior to the EPR law due to the end-of-life disposal costs, a sentiment other states share. Residents in Florida, for example, resisted the idea of paying for e-waste dropoff, with 64% expressing a willingness-to-pay of $0 (Wagner, 2009). However, with the law, disposal costs were lowered to the point that 64% of residents paid between $0-5 for drop-off (Wagner, 2009).

The focus on shared responsibility helps Maine's EPR program. By spreading the cost of funding the program between all stakeholders—consumers, producers, and municipalities—the cost to each party is low enough to facilitate the movement of equipment. The lifted burden was felt most strongly for consumers, who determined that the diluted cost of recycling, spread between property taxes, end-of-life fees, and transport, was now lower than the mental cost of stockpiling e-waste.

While California approached funding for e-waste programs from a different stage of the economic cycle, the state faced similar achievements and obstacles to Maine. Maine residents paid recycling fees at the end of their products' lifespans, while California's ARFs shifted payment to the beginning, at sale. In both cases, consumer adoption of the fee was dependent on the rate. While a minority of surveyed Californians were unwilling to pay any fee (~15%), most were willing to pay a 1% fee (64.8%), indicating broad support for helping to finance e-waste recycling. Though Nixon and Saphores (2007) noted that the cost of recycling could not be fully covered by an ARF under 5%, they acknowledged that a low rate was preferable to no rate.

The full cost needed for an ARF seems to have deterred Maine residents from supporting it in their state. When added to Maine's 5% sales tax, product prices could have been high enough to drive consumers away from in-state retailers. Consumers would have avoided the added tax by shopping online or in New Hampshire, which had neither sales tax nor ARFs (Wagner, 2009). An ARF would have been counterproductive, damaging the state's economy and doing little to fund a working e-waste management program.

Similarly, Californians did not tolerate higher ARFs. The initial rate for devices was $6-10, but from 2009-2010, the rate was $8-25.The following year, the rate reverted to $6-10, and is now $4-6, lower than before. While this range is above the survey respondents' desired price, it is more tolerable than buying online to save $25.

States considering ARFs will need to also consider that customers may opt to avoid paying. People with the means and will to avoid paying a new tax will do so, depressing revenue and encouraging nearby states to keep low tax rates. While a federal rate would prevent circumvention of ARFs, it may be unpopular enough to be overturned similar to California's rate increase. It would be more effective for states to set their own rates according to the public's willingness to pay.

Leigh et al. (2012)'s economic analysis of Washington's EPR program serves as useful evidence for cities and states considering implementing e-waste legislation. E-Cycle Washington brought in manufacturing, retail, transportation, finance, labor, and waste management sectors to facilitate an end-of-life path for e-waste. The program had a significant impact, adding 16 million pounds to the e-waste stream in three years. The sector most impacted by the program was landfilling, as jobs and economic value shifted towards recycling. Of the $13 million in added economic output, $3.9 million were outside the waste management sectors (Leigh et al., 2012).

Whereas Maine's EPR law is designed to increase recycling in a rural state, E-Cycle Washington showed the greatest benefit in the Seattle metropolitan area. Nearly half of the state's population lives and works in the metro area, and ~69% of the state's e-waste was collected and processed there. For urban areas like Seattle, mandating the expansion of a sector that increases product life cycles and reduces the need for new purchases now has evidence of net benefits to the economy.

This study was limited in scope, so some factors may affect the observed success of the program. First, the study only investigated e-waste handling up to material recovery and processing. Facilities unable to recover mercury, glass, or metals may send e-waste to other specialized facilities, which were not included in the analysis. The study also focused on new additions to the recycling industry. Companies that started accepting e-waste after the law may be more concerned with compliance than proactive involvement with recycling. Proactive producers may operate on a wider scale than that used in the study, shipping material across state and national borders. The economic impact may be greater than this study suggests, which Leigh et al. (2012) acknowledge as a limitation.

Germany's implementation of ElektroG aimed to increase oversight of municipal e-waste management. By forming a single body to set collection and recovery targets, governments could set goals while giving producers the freedom to organize and partner with private recycling companies. However, for a nation as large as the US, implementation on a national scale may be cumbersome. That central body would need enough staff to assign tasks to every electronics producer in the country. On a national scale, a central body similar to Germany's EAR would be unfeasible for a country with over triple the population and 28 times the land area, even discounting unused wilderness.

A statewide implementation of an ElektroG analogue may be more viable, as coordinating with waste collection companies and electronics producers would be on a smaller scale. However, profitability would be another issue. Walther et al. (2010)'s cost-revenue analyses of ElektroG found consistent negative net incomes between € 8,600-20,600 per year.

None of the EU case studies show a significant increase in electronic device reuse. Germany's e-waste transport system favors bulk transport of devices to optimize vehicle space, exposing otherwise reparable equipment to damage. Value-conserving transport would ensure that more devices can be refurbished and resold, but manually packaging and cushioning each item slows down collection, affecting revenue. To save money, transport companies prioritized bulk transport, and reuse fell to a third its original rate, to 3%.

Finland's EPR law also saw low equipment reuse, with 0.4% of e-waste repurposed into a working device. Like Germany's bulk transport method, Finnish collection points lacked protection methods for reusable devices (Yla-Mella et al., 2014). Separate bins may be able to store reusable devices.

The surveys with British professionals revealed doubt about the UK's reuse rate. Interviewees felt that recycling and material recovery was more profitable than reuse, which was why it was prioritized. Compared to the time, skill, and money required to secure, clean, and test reusable equipment, recovering metals from shredded devices was simpler and cheaper, ensuring that companies could stay in business.

Cole et al. (2019) also found a shared sentiment among professionals that customer ignorance remained a factor in the continued landfilling of handheld e-waste. Due to the ease and lack of personal cost to not recycling, consumers could mix phones and other devices with landfill waste with no personal loss. The interviewees' suggestion that incentives be added to dropping off e-waste would boost collection and recycling, such as exchanges toward new purchases. Finland's retailer drop-offs employ this strategy. Customers are able to exchange an old device while buying a new one, more closely associating new purchases with trades. (Yla-Mella et al., 2014).

US businesses and consumers currently benefit from used device trade-ins. Major cell service providers (e.g. Verizon, T-Mobile, Sprint) offer cash or store credit for returning a used device, which can go towards new products. Physical retailers such as Best Buy and online marketplaces such as Amazon also have trade-in promotions (Wired, 2019). Adding financial incentives for customers to let go of old devices reduces stockpiling and helps close the loop on e-waste more quickly.

Planning ahead for new e-waste flows will increase the efficiency of the waste management system. A circular economy, in which product materials are sent back to create new products, helps limit the amount of money, energy, and resources spent mining for new materials. Integrating electronic products and e-waste streams into a circular economy conserves the value of those materials over time and minimizes waste (Bridgens et al., 2019). Because the e-waste stream's composition changes with the addition and loss of new technologies, wanted materials may not always be available. Althaf et al. (2019) propose closer examinations of market trends in the electronics industry to anticipate material flows and guide policy. The authors note that current e-waste legislation in the US is reactive, responding to flows of products that are either obsolete or headed towards obsolescence. For example, CRTs are obsolete compared to flat screen displays, and rarely sold. However, they are one of the only categories of e-waste that is nationally banned from landfilling due to their toxicity (Balde et al., 2017).

Recommendations

Having analyzed various e-waste management strategies, we can now make recommendations on how to further guide US e-waste policy. Federal programs that can feasibly be implemented are discussed, followed by programs appropriate for the state level.

EPR is not a new principle in the US; states have included it in waste legislation since the early 1990s (Nash and Bosso, 2013). Since then, 32 states have enacted EPR legislation to address e-waste, batteries, paint, and products containing mercury (Nash and Bosso, 2013). The level of familiarity state governments have with EPR make it easier to recommend as a national guideline for increasing e-waste collection. EPR has successfully reduced landfilling and increased e-waste recycling in each of the study areas it was implemented in. In Europe, consumers had more options and opportunities to leave e-waste with producers, and companies were able to coordinate to meet collection targets. In the US, EPR has shown more economic advantages, such as being more affordable for states than municipal pickup and spurring job growth in urban areas. Every state can be free to go about meeting federal collection and recycling targets in ways that work best for them. I also recommend shared responsibility as a useful cost structure for funding EPR programs. Splitting the cost between groups of stakeholders makes those costs low enough to be publicly acceptable.

At the national level, education programs about managing one's end-of-life electronics will increase the rate of recycling behavior. People unaware of the environmental consequences of e-waste mismanagement will be more conscious of how to dispose of waste. Clear messaging of available recycling options will allow consumers to more easily seek out recycling methods instead of mixing e-waste with other disposables due to a lack of information.

The rate of reuse was surprisingly low in the European case studies, considering the placement of reuse in the waste hierarchy chart. However, the logistical reality of securing, transporting, repairing, and testing electronics before reuse impedes focusing on reuse. In addition, Boldoczki et al. (2020) showed that large home appliances tend to be not worth repurposing, due to the environmental impacts of reuse outweighing the material saved from extending an appliance's lifespan. As a result, a focus on reuse over recycling in accordance with the EU's waste hierarchy is not currently recommended. However, continuing to provide incentives for consumers to trade in covered devices (monitors, phones, laptops) benefits consumers, producers, and retailers, due to the added income and savings in manufacturing costs. Trade-ins should be expanded where possible to maximize these outcomes.

MFAs have proven useful in predicting future material trends in the e-waste stream. While Althaf et al. (2019) demonstrated the method with small devices and televisions, more analyses for other equipment categories can inform future material recovery decisions. For example, as solar panel technology matures and old panels are discarded, an MFA of their components could show the optimal time to begin closing the materials loop by shifting towards recovering those components. The renewable energy sector could be made more sustainable by using sales and product data to extend the longevity of existing resource stocks.

At the state level, adoption of PROs would be useful for facilitating collection and processing of e-waste between cities and counties. Similar to Finland, tech companies and other businesses can use business-to-business PROs to quickly move large quantities of equipment, and consumers can utilize collective PROs that organize pickups, drop-offs, and transport. In addition to retail and curbside pickup, collection points in rural areas would help reduce stockpiling, landfilling, and dumping. Lastly, allowing multiple PROs is recommended because of increased competition and wider coverage. To serve the most remote communities, PROs need to be allowed to specialize and fill niches.

Funding seems to be one of the most prominent sources of disagreement among the American public. While Californians were comfortable with ARFs under $6, Maine residents preferred low to no cost end-of-life drop-offs. Meanwhile, most Floridians were unwilling to pay directly for drop-offs at all. Applying one funding method across all fifty states will leave many dissatisfied. While one rate or funding scheme may be the most efficient based on cost to consumers and benefits gained, public perceptions may still influence acceptance.

The federal government can help bridge state gaps in funding. By subsidizing waste management programs, pressure for funding can be taken off states, which can in turn offer lower pricing rates to residents. To meet the full cost of financing e-waste programs, comprehensive demographic, behavioral, and contingent valuation data from surveys similar to Nixon and Saphores (2007) will be necessary to determine what funding schemes residents in each state will find most acceptable. Funding schemes can then be modified at the state level until a balance is found between resident satisfaction and economic sustainability.

Conclusion

This study outlines methods for the US to increase the efficiency of its e-waste management strategy at a national level, while giving states the flexibility needed to meet national goals. Recommendations are made using successful policies from EU nations and US states, and potential barriers are addressed by modifying policies to fit in a US framework. Regardless of whether the US as a whole adopts these suggestions, they can be used to inform state and municipal e-waste management decisions.

A few factors limit the scope of this study. The lack of recent willingness-to-pay and other survey data for e-waste management in the US affect the results. The most recent survey found was Nixon and Saphores (2007). While the survey generally informed attitudes about the environment and the ARF, it was taken before the ARF was implemented and adjusted. More recent data would improve the accuracy of this case study.

Second, this study draws parallels between e-waste management schemes in the EU and waste management and political structures in the US. Due to the time frame of this paper, a limited sample of case studies was chosen, so three WEEE Directive compliance schemes out of over 150 were chosen. While adding more examples from countries such as France, Spain, or Sweden would have provided different perspectives, the countries used in this study had a diverse set of successes, shortcomings, and lessons to inform US policy.

Greenhouse gas reduction was initially a more prominent motivation behind this paper. Maintenance and reuse are ranked high on the waste hierarchy due to their prevention of raw material mining, which requires carbon-intensive machinery. By evaluating a program based on the amount of carbon prevented from release, the program's impact could be more solidly quantified. However, such a result would require a carbon footprint analysis of multiple e-waste categories, compared with a similarly detailed assessment of a program's waste flow. Few studies were found that attempted carbon footprint analyses on e-waste programs, and none were found that analyzed carbon reduction from programs.

In March 2019, China announced that it would halt all waste imports by 2020 (Reuters, 2019). This move was part of a larger plan for China to address waste pollution and illegal waste treatment within its borders. The ban has also interrupted e-waste exports in developing countries worldwide. The full impact of this decision on e-waste industries in the US and other countries is still unclear. Whether it inspires more domestic e-waste processing and disposal or shifts exports to another developing country remains to be seen.

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