Quantifying barriers to recycling plastic packaging in the United States

by

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Submitted to the Department of Materials Science and Engineering
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Abstract

Staggering statistics about plastic waste have driven multi-stakeholder engagement from entrepreneurs searching for innovative technological solutions, companies pledging circular measures, and nations setting ambitious recycling targets. Yet, recycling rates for plastics remain abysmally low (<8%). In this dissertation, I use a supply-demand approach to investigate why we don't recycle more plastics and quantify barriers to better recovery and recycling outcomes. Recovery system architectures depend on geographic context. I examine plastic waste recovery in the United States (US), the world's largest producer of both total and per capita plastic waste. Of the 38 million tonnes of post-consumer plastic waste generated in the US in 2021, approximately 42% can attributed to single-use plastic packaging. I probe two large-volume plastic packaging contexts: PET (polyethylene terephthalate) bottles and flexible packaging made of PE (polyethylene) and PP (polypropylene). Demand for recycled PET has risen sharply due to policy and consumer pressures that target circularity. However, in the US, PET bottle collection rates have not increased in a decade, and the supply of recycled PET is unable to meet rising demand. Therefore, in the PET bottle case study, I quantify the cost of a supply-side policy push to increase recovery. I find that deposit return systems invite incentivized consumers into the recycled PET value chain, and recycled content mandates for PET bottles can reduce the net cost burden of a nation-wide deposit return system, improve PET bottle recycling rates to >80% (from 24%), and save 7.6 MtCO$_2$eq per year. Unlike PET bottles, demand for mechanically recycled PE from flexible plastic packaging waste is low, but researchers are exploring advanced recycling pathways to divert hard-to-recycle packaging formats from disposal. I survey advanced recycling methods proposed in the literature and investigate the techno-economic potential for their market scalability. I find that large-volume products such as fuels or feedstock chemicals are limited by the availability of plastic waste at low costs. On the other hand, high-value end-uses such as fine chemicals are constrained by process yields and unmatched scale of plastic waste generation. I generalize these findings to examine the role of policy in improving the plastics recycling landscape in the US. Lastly, I discuss the petrochemicals value chain and the significance of displacement in realizing the expected environmental benefits of plastics recycling.

Thesis Supervisor: Elsa A. Olivetti
Title: Jerry McAfee (1940) Professor of Materials Science and Engineering, Associate Dean of Engineering

Acknowledgements

Curiosity steered many of my choices at MIT. In my first semester, it led me down the Building 56 corridors to an orientation for the Program for Polymers and Soft Matter, following which, almost on a whim, I decided to join it. At the same time, I was exploring the grocery stores and supermarkets around me. Beyond the hundreds of choices of products always available to me, I was fascinated by the packaging. So much packaging. And plastics (which are polymers). In my classes, I learned about the wondrous ways in which polymer strands stretch and slide, curl and constrict, and make some of the most brilliant materials invented in the last century. Outside of my learning, I confronted curiously confusing instructions for recycling, marveled at the ubiquity of plastics, and shopped around MIT for an advisor. A quirky coincidence of fascinations and funding, and a few discussions with Professor Elsa Olivetti, produced a project about plastics recycling. I suddenly gained a PhD advisor and the ability to translate my curiosity into research hours.

Elsa is amazing. But I must expand. Sometime late in 2022, I decided to rethink this dissertation as an experiment in writing. I roped Elsa into this experiment, and she (to my astonishment) willingly, even enthusiastically, participated. She took on many roles as this experiment advanced – a supportive observer, a thoughtful thought partner, an insightful reviewer, and the kindest advisor. This co-creation experiment is just one example out of many academic endeavors and experiences with Elsa that I will fondly remember. To not take too much space, I must gloss over: the welcoming spaces she created for all the various academic tangents conceived by my curiosity; the wells of warmth and understanding she showed me when I relayed some of my struggles; and the many meetings where her contagious enthusiasm for research and collaborative instincts added joyous new dimensions to my work. As my PhD progressed, Elsa's responsibilities to the institute also compounded. Even so, she continued to carve out just enough time for me, whenever I needed it. I say this because as I look back, I realize that I never felt alone in this strange journey. And for this, I cannot thank her enough.

Joyous new dimensions introduced by Elsa invited many exciting people into my PhD journey. Some, like Karan Bhuwalka, made a lasting impression that is captured in this dissertation. Karan is a dear friend and colleague, who, beyond just being a big part of what is now Chapter 2 in this document, has been an intellectually inspiring and steadfastly supportive presence in my research life. I could not have asked for a better research teammate. Speaking of Chapter 2, it is a product of numerous, rich discussions that Karan and I had with Dr. Richard Roth from the Materials Systems Lab (MSL). I am grateful for Rich who carefully and constructively cross-examined our arguments, saving us from modeling mishaps, and more importantly, motivating new modeling adventures. I also had the pleasure of working with several other wonderful people from MSL over the years. Notably, I am grateful to Dr. Randolph Kirchain, Dr. Jeremy Gregory, and Dr. Elizabeth Moore who helped me develop my own little niche in materials systems research when I was just getting started.

I must acknowledge the Olivetti group and its many members over the years for making research enjoyable in its own right. I'd like to thank Jacqueline Baidoo and Oksana Makarova more specifically, who, beyond being awesome, also contributed critical research support to what is now Chapter 3 in this document. I am also grateful for my collaboration with Dr. Poushali Maji from MIT's Climate and Sustainability Consortium (MCSC) as our many conversations about policy instruments and contexts enriched this thesis. Lastly, I'd like to thank my committee members, Professor Desiree Plata, and Professor Robert Macfarlane, for their continued support and guidance as my thesis evolved windingly over the years, with several detours – some exciting, some tedious, and some admittedly a tad harebrained - along the way.

Behind the scenes of my academic life at MIT, there's a constellation of family and friends. They help me find my way. My partner, Rishi, is a constant source of compassion and consideration that I have drawn from more often than I can admit. I cannot imagine this journey without him by my side. I'd also like to fondly record the gratitude I feel for the comfortable, long-distance friendships of some of my dearest friends – Aakriti, Tejas S, Yashaswini, Tejas K, Vishwa, Raunaq, Roshail, and more. They're my antidote to the anxious vicissitudes of school and life.

Finally, I want to thank my family. My mom and dad enthusiastically supported me in all my choices – including one that whisked me halfway across the globe. Both worked hard to create spaces where I could not only access but also indulge my curiosity unabashedly from a young age. The words in these paragraphs cannot reasonably contain my appreciation for their unconditional love and kindness. I'd also like to thank my sister, my favorite person on this planet, for winning this title consecutively for the last 20 years. She has been inextricably linked with my PhD journey in so many ways that have helped me grow into the person I am happy to be today.

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Chapter 1

Introduction

Polymers are macromolecules comprising many repeat units and exhibiting high molecular weights. Natural polymers make up most of plant and animal biomass on the planet – from proteins and nucleic acids to sugars and cellulose. Naturally, polymeric materials have been used for textiles, construction, and packaging for thousands of years – but the macromolecular structure underpinning their unique properties was not known. In 1920, Herman Staudinger, offending prominent organic chemists of the time, posited the novel concept of covalently bonded long-chain molecules in his 1920 paper "Uber Polymerisation" (Frey & Johann, 2020). Soon after, Staudinger and Fritschi's experimental work, "On Isoprene and Rubber", added empirical evidence to this new concept (Mülhaupt, 2004), and coined the term "macromolecule", marking the beginning of the flourishing field of polymer science and technology (Furukawa, 2016). The field would go on to rapidly create millions of remarkable materials that form the cornerstone of modern life in the twenty-first century.

While the macromolecular hypothesis was slowly garnering support in the chemical sciences through the early half of the twentieth century, large-scale industrial development of oil fields and advances in refining and petrochemistry led to the innovation and proliferation of new synthetic polymers (Spitz, 1988). These fossil fuel-derived macromolecules gave rise to materials whose diversity and versatility captured the imagination of an expeditiously developing twentieth-century society. Between 1950 and 2020, global production of synthetic polymers grew from 2 Mt to 380 Mt(Geyer et al., 2017). In less than 70 years, humanity had cumulatively produced 9.2 billion tonnes of synthetic polymeric materials (Geyer et al., 2017). Today, polymers find use in every economic sector– from packaging to healthcare to transportation (IEA, 2018) – and their indispensability is near absolute.

Polymers can be categorized along several different dimensions – by synthesis method, by end-use sector, by mechanical properties, and by crosslinking structure (Lodge & Hiemenz, 2020). The latter differentiates between thermosets (crosslinked, irreversibly cured) and thermoplastics (uncrosslinked, can be reversibly melted). ‘Plastics’ – a standard industrial term as well as common jargon for most man-made polymeric materials is not well-defined. Etymologically, the term is characteristic of moldability but may exclude elastomers (rubber) and/or synthetic fibers. However, textbooks and academic journals also use polymers and plastics interchangeably. In parts where the dissertation discusses matters of markets, policy, and environmental economics, ‘plastics’ broadly refers to thermoplastic polymers, not including synthetic fibers. In more technical sections, the term is avoided in lieu of specific polymer types such as polyethylene terephthalate (PET) or simply the word polymer to denote macromolecular chemistry.

1.1 Environmental concerns and recycling

The explosive growth of polymers in the economy has raised many environmental concerns across the material's life cycle. Plastic production inherits the environmental externalities posed by the fossil fuel industry as it uses fossil carbon for both feedstock and process energy (Cabernard et al., 2022). Estimates suggest that the rapid growth of plastics in the economy and reduced reliance on fossil fuels in other sectors such as transportation could increase the share of oil consumed by global plastics production from 6% in 2015 to 20% by 2050 (Ellen Macarthur Foundation, 2016). Greenhouse gas (GHG) emissions intensity of production and manufacturing of plastics varies by type but Zheng and Suh (Zheng & Suh, 2019) estimate that global plastics consumption caused 1.7 $GtCO_2e$ of emissions in 2015 (around 5% of total global emissions) and is posed to grow to 6.5 $GtCO_2e$ by 2050. At end-of-life, every year approximately 20 Mt of plastic waste enters terrestrial and aquatic ecosystems around the world (Borrelle et al., 2020). Litterbase (Bergmann et al., 2017) documents that almost 3700 aquatic species have been affected by marine pollution. In Figure 1-1, I visually summarize studies over the last decade that have estimated or projected the quantitative growth of environmental concerns.

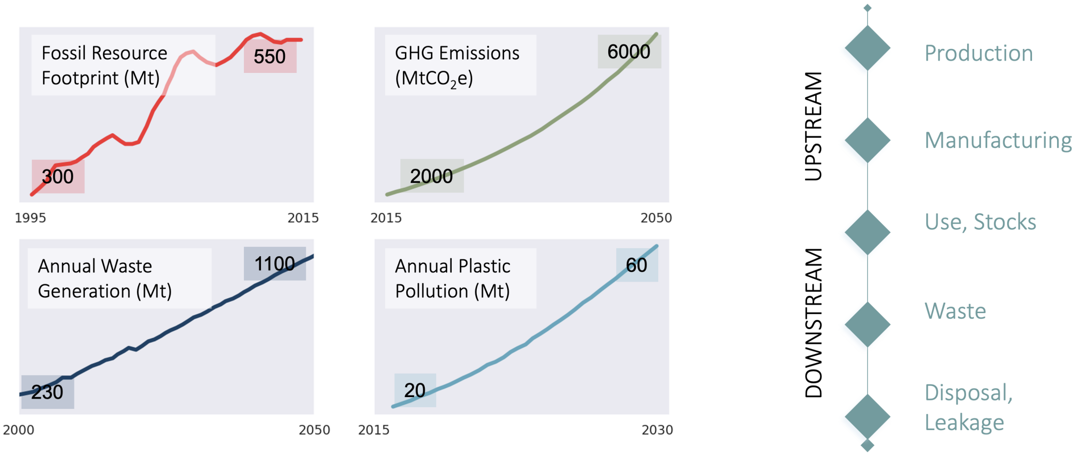

Figure 1-1: Summarizing literature studies of historical and projected growth of externalities associated with plastic production and disposal. Data from Borrelle et al., 2020; Cabernard et al., 2022; Stegmann et al., 2022a; Zheng and Suh, 2019,

From these concerns, one can distill five problems that can use recycling as a (partial) solution: (1) fossil resource consumption, (2) climate impacts of production and manufacturing, (3) waste management/disposal, (4) plastic pollution at land and sea, and (5) waste accumulation. The extent to which recycling can practically tackle these problems depends on plastic types, product/application sources and sinks, regional contexts, and historical developments. My study, and this dissertation, is about plastic packaging in the United States (US). Historically, for plastics, recycling motives can be traced to problems associated with rapidly growing post- consumer waste. This contrasts with valuable metals and alloys such as copper or steel where recycling was driven primarily by economics (Söderholm & Ekvall, 2020). As lightweight, short-lived plastic products and packaging flooded consumer markets in the latter half of the twentieth century, the US was phasing out unsanitary landfills, the waste industry was consolidating beyond local and state-managed landfills, and curbside recycling was developing across US cities as a benign alternative to waste-to-energy where landfill space was scarce (Pollans, 2021). Within this shifting waste management landscape, policy objectives and public action on local, state, and national levels were set to tackle plastic litter and reduce the municipal waste burden (Hadjilambrinos, 1996). Between 1960 and 2000, the share of plastics in the municipal waste stream grew from 0.5% to 10% (US EPA, 2020); as plastic waste volumes grew visibly, the focus on recycling intensified . Situated in this context, I untangle policy, technology, and market factors that have informed the US plastics recycling system architecture and analyze the limits to which recycling can solve the many problems simultaneously.

1.2 Circularity for plastics

Circularity is defined in numerous ways (Saidani et al., 2019) and has been varyingly described as a materials efficiency lever, a zero-waste strategy, a sustainability principle, and more. For plastics, 'The New Plastics Economy' report presents some of the earliest circularity calls to action, persuading institutions to combat plastic pollution challenges (Ellen Macarthur Foundation, 2016). The report envisions a 'circular' plastics economy as one operationalizing materials efficiency strategies towards minimizing plastic waste and pollution, and its circularity framing has become central to the discourse on recycling plastics. Many policy and business perspectives have opined what this vision means (Baldassarre et al., 2022; Gao et al., 2023) while academic articles have collated definitions and metrics for circularity (Corona et al., 2019), identified dimensions of sustainability (Leipold et al., 2022) and mapped them to circularity objectives (Saidani & Kim, 2022) or criticized equating the two (Blum et al., 2020).

Some authors consider recycling to be a circularity strategy (Allwood, 2014) or inherently part of a circular business model (Morseletto, 2020), while others argue that only closed-loop recycling activities must be considered circular (Harris et al., 2021). Vague definitions continue to generate subjective and inconsistent interpretations (Corvellec et al., 2022), but for the purpose of this dissertation, I choose the latter definition because it is most common in plastics recycling literature. In this view, for recycling to be circular, the material must be able to be recycled again and again. Because different applications have different material quality needs, for a material to be recycled again and again, it must go into applications that are recycled. In plastics, the diversity of applications necessitates restricting recycled material use in the original application. For example, a PET bottle being recycled back into a PET bottle allows circularity. In contrast, if PET from bottles is used in polyester textiles, it is unlikely to be recycled again.

The circularity concept is still fuzzy: the PET bottle that uses recycled PET may not actually be recycled again – and end up in a landfill, or the textile might be recycled into carpets that is recycled one or two more times. The concept also does not broadcast additional information on sustainability. Geyer et al. demonstrate that recycling more than once or closed-loop recycling does not necessarily imply superior environmental benefits (Geyer et al., 2016). However, the circularity concept remains firmly embedded in plastics recycling literature. It confounds many of the trade-offs that I explore in this dissertation. Trade-offs are commonly considered in materials' chemistry (optimizing across multiple properties) and sustainability science (optimizing for lower environmental impacts). For plastics, circularity as a sustainable strategy provides interesting case studies where these considerations intersect.

1.3 Polymer chemistry and quality challenges

Many thermoplastic applications, including packaging, rely on properties of the material in the solid state at room temperature. In the solid state, thermoplastic polymers are amorphous or semicrystalline materials. In an amorphous polymer, below the glass transition temperature, long chain molecules are kinetically 'frozen' in place in non-equilibrium arrangements with no long-range order (Lodge & Hiemenz, 2020). On the other hand, in a semi-crystalline polymer, below crystallization temperatures, chains are also frozen albeit in ordered crystal lamellae suspended between amorphous regions (Lodge & Hiemenz, 2020). A polymer's crystallization ability and the degree of crystallinity achieved depends on repeat unit structure and microstructural composition (copolymers, stereoregularity, etc) but all such polymers can be melted[1] by raising the temperature above glass transition and/or crystallization temperatures. Melting relaxes stresses frozen in from previous manufacturing and molding processes: a polymer melt is an unordered, random collection of polymer chains that can flow – and by cooling again, it can be reformed, and recycled.

In the discussion that follows, I present a macromolecular perspective on recycling – tying together polymer chemistry and plastics recycling concepts to describe material quality

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[1] Melting is a terminology reserved for a first-order thermodynamic phase transition. Glass transition is not such a transition and researchers debate whether it can be accurately described as a second-order transition instead (Nemilov, 2018). Either way, melting would be technically incorrect for amorphous polymers like atactic polystyrene – however, for the purpose of this dissertation, the difference is immaterial. Moreover, both PE and PET are semi-crystalline.

challenges. Material quality is a relative concept – it depends on function, and utility of that material towards an application or product. In recycling literature, high quality is associated with circularity, i.e., a high-quality recycled material can be recycled back into original use (Helbig et al., 2022).

1.3.1 Mechanical recycling

Mechanical recycling refers to the process of melting and cooling to reform plastics. In theory, all thermoplastics are infinitely recyclable by melting and reforming. In practice, mechanical recycling is complicated by conditions of use and recovery. Mixing of different plastic types, polymer heterogeneity within a plastic type, and chain degradation during mechanical recycling can all compromise quality of recycled plastics (Vilaplana et al., 2007).

'Plastic' is a vast umbrella term for millions of synthetic thermoplastic polymers, each with their own distinctive chemistry, that must be separately recovered for mechanical recycling. In this dissertation, I discuss PET and PE extensively, and most examples trace this focus. Though the lines along which plastic types are differentiated is ambiguous in practitioner literature, IUPAC recommends nomenclature based on repeat unit or monomer chemistry (IUPAC, 2008). Polyethylene (PE) is made by polymerizing ethylene monomer; polypropylene from propylene; polyvinylchloride (PVC) similarly uses the vinyl chloride monomer; polyethylene terephthalate (PET) is made by esterification[2] of ethylene glycol and terephthalic acid (or its derivatives) (Odian, 2004). Repeat unit chemistry affects thermal properties such as glass transition and melting temperatures, as well as packing and crystallization (Lodge & Hiemenz, 2020). For instance, consider PE recycling: PP melts at a higher temperature (approx. 230 °C) than PE (approx. 160-200 °C). Therefore, during mechanical recycling of PE, PP will remain suspended in a PE melt matrix, disrupting crystallization, and compromising desired properties (Karaagac et al., 2021; Kazemi et al., 2021). In PET recycling, PVC, even in small quantities, degrades producing hydrochloric acid (HCl) that reacts with PET to form unintended side products during recycling,

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[2] Esters usually have '-ate' or '-oate' suffixes derived from the carboxylic acid and prefixes modeled after the alcohol root. Because PET was one of the first polyesters commercially synthesized for fiber applications, PET used in textiles is still called polyester.

compromising molecular structure and properties (Awaja & Pavel, 2005). Therefore, once collected, plastics must be at least sorted by polymer type for mechanical recycling – this step adds costs but can be practical for some large-volume thermoplastics. Material recovery facilities (MRFs) today sort large-volume plastic types such as PE, PET, PP using near-infrared technologies that identify polymer spectroscopic signatures (Scheirs, 1998), and PE or PP and PET can be separated based on density. Besides non-target polymers, contaminants can include other materials such as metals or paper both due to convenient product design (paper labels), and economic collection modes (curbside recycling of many household recyclables together). Setting aside collection and sorting challenges, even if we assume all PE or PET can be isolated and decontaminated at cost, two problems remain.

First, recycled plastics blend numerous plastic sub-types. The versatility of commodity thermoplastics is derived from the ability to modify microstructure and morphology to achieve the desired properties. For instance, the PE plastic type consists of numerous polymers – all synthesized with ethylene, but with different chain structures, and comonomers (Peacock, 2000). I call these variations subtypes[3] i.e., polymers having the same major repeat unit but different chain shapes and sizes. PE subtypes exemplify the role of structure-property relationships in optimizing polymer and process design towards an application (Mills et al., 2020). Differences in chain structure also hold implications for recycling – in the most visible example, the difference between high-density polyethylene (HDPE) and low-density polyethylene (LDPE) was significant enough to earn them distinct resin identification codes[4] – 2 and 4 respectively. HDPE polymers are almost linear with few short chain branches (4-8 carbon long) while LDPE polymers are highly branched with short and long chain branches (>100 carbons).

Chain structure affects both polymer rheology and morphology. How polymer melt flows

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[3] In industry jargon, these subtypes are called resins or differentiated by tradenames. 'Resin' is a confusing terminology as the word is also used to refer to crosslinked thermoset polymers used as a synthetic alternative to natural plant resins for applications like lacquer.

[4] The US Society of Plastics Industry (SPI) rolled out the "Voluntary Plastic Container Coding System" in 1988 as curbside recycling was maturing to help facilitate post-consumer plastics recycling – i.e., help consumers identify which plastic product/packaging to place in the recycling bin. However, the chasing arrows symbol used in the code design has since been co-opted as a recycling symbol, and now the coding system is considered outdated at best, and intentionally misleading, at worst. ASTM maintains that Resin Identification Codes are not 'recycle codes' but meant to aid recycling (ASTM, 2021).

(rheology) is critical for process design, and how polymers pack and crystallize (morphology) impacts mechanical and performance properties. Long linear chains of HDPE crystallize well (>75% degree of crystallization), are denser due to close packing, and have higher yield strength and flexural modulus than LDPE (Patel, 2017). Consequently, some HDPE subtypes are used to make drum containers for industrial chemical storage. On the other hand, lower crystallinity in LDPE subtypes allows lower melting points, and higher melt strength that can maintain bubble stability in blown-film processes (Cantor, 2006). Producers and converters optimize polymer and process design to obtain desirable processing and in-use properties. Consider an application such as trash bags - LDPE films are not very strong or have much gas barrier protection, so HDPE is copolymerized with branched comonomers to improve processability while maintaining higher crystallinity that provides better mechanical and barrier properties (Jenkins & Osborn, 1992).

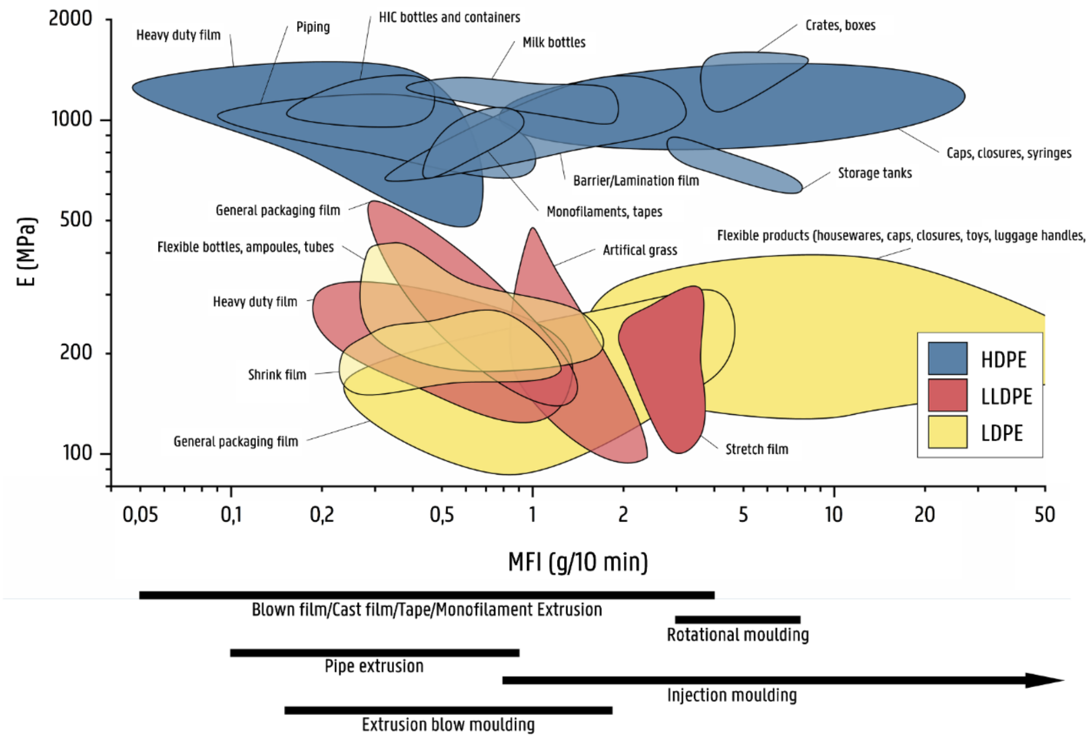

Figure 1-2: PE types, applications, and properties. Figure reproduced from Demets, R., Van Kets, K., Huysveld, S., Dewulf, J., De Meester, S., & Ragaert, K. (2021). Addressing the complex challenge of understanding and quantifying substitutability for recycled plastics. Resources, Conservation and Recycling.

Figure 1-2 shows the diversity of PE properties, and visualizes the practical applications derived from microstructural variations. Polymers are stochastic molecules, and polymer chain structure and shape can be varied by industrial synthesis methods along several dimensions – branching, molecular weight, and its distribution, comonomer ratios and microstructures, etc (Odian, 2004). This vast combinatorial space allows endless customization of virgin PE subtypes for applications – but creates quality problems at end of life (Demets et al., 2022). Even if PE can be removed, flexible LDPE containers in HDPE bottle stream can lower crystallinity of recycled polymer and decrease mechanical properties (Thoden van Velzen et al., 2021). Films made from two different sub-types for different applications – such as stretch films and shrink films – cannot be easily recycled together into either application (Eriksen et al., 2019), even if they are mono-material (i.e. only LDPE, no multi-material layers) products. Moreover, composition of polymer used in a product, and composition of products collected and sorted together, are highly variable and unknown. As a result, only few end-use applications can tolerate mechanically recycled polymer made from mixed polymers sub-types (Demets et al., 2021).

The second problem beyond collection and sorting is the loss of properties during mechanical recycling. Even if a single large product type uses relatively similar polymer subtypes, the recycling process itself leads to structural changes in the polymer (Badia et al., 2012; Cruz & Zanin, 2003). High temperatures and shear profiles in the mechanical recycling equipment – an extruder – create physical stresses and facilitate chemical reactions that can cleave polymer chains, create reactive intermediates, and alter molecular weight distribution of polymers (Schyns & Shaver, 2021). Subtle changes in molecular weight can affect rheological (flow) properties, crystallinity, as well as mechanical properties (Robledo et al., 2009). For example, LLDPE (or linear low-density polyethylene), presents properties between HDPE and LDPE, and has significantly grown flexible film applications over the last decade (Patel, 2017). LLDPE subtypes, synthesized by copolymerizing ethylene with alpha olefins, are designed by precisely controlling branch points using comonomer content (Odian, 2004). When reprocessed, chemical changes in the extruder can alter branching density and molecular weight distribution, and compromise properties for the original application (Demets et al., 2021). However, this does not necessarily prevent recycling: degradative pathways can be curbed to some extent using antioxidant additives, and clean films can be recycled into products such as injection molded articles that do not have as strict processing requirements (Horodytska et al., 2018). Nevertheless, since recycling back to original application is preferred (Helbig et al., 2022), mechanical recycling is often considered inferior (Li et al., 2022; Rahimi & García, 2017).

1.3.2 Beyond mechanical recycling: advanced recycling

Beyond collection and sorting by plastic types, quality issues evidently stem from the macromolecular nature and complexity of polymeric materials. Advanced recycling methods attempt to solve these problems by deconstructing the macromolecular chains; I cluster all recycling methods (including what scholars define as chemical recycling, molecular recycling, upcycling and more) that intentionally cleave the polymer chains to make new products under this definition. While mechanical recycling retains the material's plastic type and polymeric identity (recycled PET is still PET, a polymer), advanced recycling[5] - defined this way - does not necessarily recreate the polymer.

Few polymers can be chemically deconstructed into constituent monomers at high yields under economic conditions to permit polymer-to-polymer recycling and circularity (Rahimi & García, 2017). The chemistry and practicality of depolymerization can be better understood by reviewing basic polymerization chemistry (Odian, 2004). Odian distinguishes between two broad classification archetypes. Step and chain growth polymerizations describe synthesis reaction mechanisms, while condensation and addition polymerizations differentiate based on the presence of functional group linkages in the polymer backbone. PET is a condensation polymer synthesized by step-growth polymerization, and PE is a chain polymerized addition polymer. While both polymerization processes are exothermic, PET is synthesized through stepwise polyesterification with removal of water driving the

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[5] Since advanced recycling of plastics allows greater fungibility of material identity (i.e., molecules can react and form new materials), such processes stray from conventional recycling definitions that are fashioned to describe metals recycling (for instance, lead remains lead, and copper remains copper). Alloys can be diluted or converted to other alloys, but the identity of individual metals in the alloy material is preserved. Due to inconclusive definitions, advanced recycling is not well-recognized in legislative fine print, and several US states have expressed differing opinions on considering it recycling.

equilibrium towards longer chain formation, while PE grows by chain propagation to high molecular weights.

Different polymerization pathways point to different depolymerization possibilities. PET, owing to ester linkages, is amenable to relatively facile solvolysis (including hydrolysis or adding back water) to its repeat unit constituents (terephthalic acid and ethylene glycol), and can be reconstructed back into virgin-like PET of desired chain structure and properties (Barnard et al., 2021). From a thermodynamic perspective, depolymerization to monomers is usually entropically unfavorable ($\Delta S < 0$) and requires high temperatures to overcome enthalpic barriers (Ellis et al., 2021). Kinetically, thermal reactivity of polymers is complex, and products from degradative reactions with lower activation barriers than depolymerization can dominate the yield. For PET, researchers show that suitable catalysts and enzymes can improve yields, energetics, and economics (Barnard et al., 2021). On the other hand, polymers such as PE that only have C-C linkages in the polymer backbone cannot be depolymerized to their monomers directly. For chain polymers, a ceiling temperature can be defined; this is the temperature at which the rate of propagation and de-propagation of the chain are equal (Odian, 2004). Above the ceiling temperature, de-propagation should be favored – but de-propagation is one of many reaction pathways available to the polymer chain, some of which are favorable far below the ceiling temperature. As a result, depolymerization of PE yields a mixture of hydrocarbons that can be tuned by varying reaction conditions and catalytic measures (Scheirs & Kaminsky, 2006). If ethylene is desired, PE must first be deconstructed into short chain hydrocarbons (typically in the fuel-range), distilled to produce naphtha for steam-cracking, following the usual petrochemical pathway for ethylene production.

Producing ethylene from PE in this manner is highly uneconomical (Yadav et al., 2023), and advanced recycling studies on PE (and other polyolefins) do not focus on recovering the monomer or repolymerizing to PE. Instead, the research community innovatively uses waste polyolefin feedstock to synthesize various organic chemicals and fuels (Zhao et al., 2020). As a result, advanced recycling of PE is not compared against mechanical recycling on a quality basis but instead on a 'value' basis. Dubbed 'upcycling', polymer researchers optimize reaction conditions to optimize for products priced higher than PE itself – arguing not for product or polymer circularity, but instead for a demand-side market pull for collection of waste plastics (Jehanno et al., 2022). This is a notable shift in perspective – one that motivates higher waste recovery but may not improve life cycle impacts of plastics consumption. By posing it as a 'value' maximizing problem, researchers narrow their solution space to either (a) niche expensive chemicals that do not have the scale to match the PE/PP waste problem or (b) produce fuels to subsidize the costs of cheaper commodity chemicals production. Fuels production violates material cycling – decoupling this perspective from any of the benefits promised by recycling, while niche chemicals do not make a sizeable dent in decreasing waste disposal. Numerous authors propose upcycling processes to consume waste plastics and link it to the circular plastics economy (Jehanno et al., 2022) through vague ambitions (Ellen Macarthur Foundation, 2016), but based on our discussion on circularity, upcycling is not circular. Practical scalability and net system benefits of such processes are also uncertain (Davidson et al., 2021; Solis & Silveira, 2020).

Coates and Getzler instead advocate for 'an ideal, circular polymer economy' by designing new monomer/polymer systems that are more amenable to chemical recycling to monomer using ring-opening polymerization (Coates & Getzler, 2020). Ring-opening polymerizations usually follow chain kinetics but unlike polyolefins, have heteroatom linkages amenable to depolymerization like condensation polymers (such as PET). Building on chemically recyclable polymers, Shi et al. (Shi et al., 2021) present design principles that can guide monomer design for 'intrinsically circular polymers' (their term) – polymers that can depolymerize and repolymerize. Coates and Getzler note that new monomer chemistries must displace applications using recalcitrant PE polymers. But the sheer scale and low-cost economics of ethylene - the backbone of the petrochemical industry (Nichols, 2013) - presents a steep barrier. Moreover, the diversity of PE subtypes and applications mandates new monomers for each application – further diversifying plastic types and diluting recovery potential. From a practical perspective, alternative chemistries present interesting challenges to polymer chemists but are unlikely to solve the problems associated with large-volume thermoplastics such as PE in the near term at scale and are not discussed in this dissertation further.

Advanced recycling methods promise higher quality polymers and/or higher value products over mechanical recycling. Whether this promise can be fulfilled depends on the plastic type, the cost of obtaining higher quality, and the scalability of creating higher value. I inquire whether, and to what extent, higher quality or value can meaningfully address the various plastic problems outlined before through two packaging case studies – PET bottles (in chapter 2) and polyolefins in flexible plastic packaging (in chapter 3). This choice clarifies the scope, contrasts two plastic types and broader polymer archetypes, and exemplifies two extremes of a spectrum of plastic packaging collection and sorting challenges. By shaping chapters around application groups within plastic types, I adopt a systems perspective that is sufficiently specific to investigate and quantify systemic barriers to recycling. PET and PE together constitute a third of plastic packaging, and 46% of all non-fiber polymers (Geyer et al., 2017), and insights from these case studies can inform the feasibility of recycling a large fraction of plastics and plastic waste. More importantly, PET and PE present two very different chemistries. Examining the implications of this difference on application types, diversity, and end-of-life possibilities explains what it would take to improve environmental outcomes for an industry as expansive as plastics.

1.4 Quantifying barriers to recycling more and better

The issue of quality, discussed so far from a macromolecular perspective, is situated at the intersection of market, policy, and technological barriers. Various authors, agencies, and industry groups have enumerated (Garcia-Gutierrez et al., 2023; Hopewell et al., 2009; US EPA, 2021) and explained these barriers: lack of collection infrastructure (Lebreton & Andrady, 2019; The Recycling Partnership, 2020), contamination in collected plastics (Roosen et al., 2020), material mixing in the sorting facility (Eriksen & Astrup, 2019), compatibility with other plastics ("APR Design® Guide", n.d.; Martínez Leal et al., 2020), property mismatch of the recycled content for intended secondary use (Demets et al., 2022; Grant et al., 2020), underdeveloped recycling markets (Meys et al., 2021) and value chain (Olatayo et al., 2023), and low demand for recycled plastics (Milios et al., 2018; OECD, 2018). However, none have attempted to quantify these barriers beyond pointing to estimates of abysmal recycling rates as evidence of these extant problems (Di et al., 2021; Geyer et al., 2017; Heller et al., 2020).

Quantifying systemic barriers to waste management is not easy. Waste, by its very nature, is a local problem – and local contexts are shaped by behavioral, historical, and political factors that resist quantification (Picuno et al., 2021; Pollans, 2021). Waste is also spatially distributed and materially aggregated. Beyond causing quality issues already discussed, this complicates attributing responsibility of waste management, and confounds collecting and interpreting information about the waste being managed. For this reason, qualitative studies in literature describe system-wide challenges across plastic types (Heiges & O'Neill, 2022). However, this broad scope denies the specificity needed to quantitatively assess the multitude of plastic types, sub-types, and applications that contend with collection, sorting, and recycling systems designed for select materials and products. At the same time, quantifying systemic barriers is necessary to advance practical, sustainable solutions to address rapidly growing plastic waste. In this dissertation, I focus on quantifying barriers to recycling plastics in two specific application-based contexts placed within the US plastics recycling system and inquire: why don't we recycle more plastics and what would it take to recycle more plastics? Within this purview, an economic perspective provides a unified framework to understand this question and construct an answer. From an economic perspective, the answer to the first question is simple: we don't recycle more plastics because recycling plastics is usually a net cost burden. Quantifying this cost burden is the first step towards quantifying barriers and answering the second question. Once quantified, weighing this cost against expected benefits clarifies the merits and limits of recycling as a solution to environmental concerns posed by plastics.

In chapter 2, I investigate the recycling of PET bottles. PET bottles benefit from PET's polyester chemistry as well as unique market characteristics and scalable collection pathways enabled by historical policy choices. I reframe the cost burden of scaling these pathways by evaluating the mismatch in recycled supply and demand. This approach has not been previously used to assess plastics recycling outcomes, but I demonstrate that it can inform policy design and examine incentive structures. By linking demand for recycled PET to quality constraints, I also discuss the limited environmental benefits of increasing value through monomer recycling. In chapter 3, I focus on flexible plastic packaging and use material flow analysis to understand the challenges in scaling up waste recovery for plastic films. I connect material flows to technological advances in recycling and explore whether increasing the value of recovered plastics via depolymerization methods is a scalable pathway. In chapter 4, I summarize the findings from the two very different case study contexts – PET bottles and PE films - and qualitative extend these insights to suggest a generalized framework. I then review the role of recycling in solving plastics' many problems and the conditions under which this role is fulfilled. To this end, I present a qualitative discussion on displacement of virgin plastics to assess if, and when, recycling *more* can be decisively linked with doing environmentally *better*.

1.5 Metrics: defining *more* and *better*

To quantitatively discuss barriers to recycling of plastics, in this dissertation, I quantify recycling outcomes, the costs of pursuing an outcome, as well as benefits of realizing outcomes. I define outcome as Schumann does in "Using Outcome Indicators to Improve Policies" (Schumann, 2016) and emphasize that this policy-relevant terminology is deliberate: historically, recycling of plastics in the US has been pushed by policy, rather than pulled by market demand, and as will be demonstrated in Chapters 2 and 3, policy continues to be a critical driver of recycling progress for many plastics. According to Schumann (Schumann, 2016) – outcome indicators measure the "progress towards a policy objective" – often indirectly – and are different from output indicators which measure what the policy produces to achieve its objectives. For example, in recycling, tax incentives for the purchase of recycling equipment were commonly implemented in many states (US EPA, n.d.). The output of such a policy, depending on how it is designed, may be an increase in recycling facilities, and the outcome will be, depending on factors within and beyond policymaking, an increase in recycling rates for relevant materials.

Recycling outcomes define what 'more' means. They are defined along policy objectives, and I distinguish three commonly stated objectives: increasing recycling, increasing circularity, and increasing landfill diversion. Recycling is the reuse of materials in the economy across all products that use it – recycling doesn't differentiate between closed-loop or open-loop applications and includes all material transformations. However, fuels are mixtures of chemical compounds intended to be combusted for energy and are excluded from this recycling definition. For example, in a lab-scale study, Xu et al (Xu et al., 2023) depolymerize polyolefin waste, hydroformylate and hydrogenize the hydrocarbon products to produce fuel fractions (intended to be refined to gasoline and diesel), and other chemicals that can be used to make plasticiziers, surfactants, and polyesters. Only the non-fuel chemicals are counted as recycling. For circularity, only the strictest level of product-to-product circularity is counted. Keeping up with the circular definition (one where materials can potentially be recycled repeatedly) this provides a narrower scope. For example, a PET bottle recycled into a PET bottle, discussed in Chapter 2, will contribute to the circularity measure but LDPE films recycled into decking will not. On the other hand, diversion from landfills is broader than recycling and includes all end-of-life fates that divert materials from landfills – including fuel and energy recovery. For each of these outcomes, a quantity outcome and/or rate outcome can be quantified. When a rate/fraction is estimated, the denominator quantity is explicitly mentioned and corresponds to estimates of waste generated for the plastic product type or material type, depending on scope.

Tracking what is being recycled into what is not easy, and the trade of recycled commodities is extremely opaque. Most gray literature estimates of recycling instead measure collection or recovery. I additionally define these terms explicitly. In this dissertation, collection refers to the first stage in reverse logistics where the material is collected with the intention of recycling – it could be collected in a commingled state (such as curbside collection of recyclables) or collected separately (such as film waste returned to retail bag return programs). Recovery is the stage at which non-target materials are removed (typically at a material recovery facility) and recovered material is baled. Collection and recovery can coincide in cases where the material is source separated such as in film return-to-retail. It should be noted that all these estimates hold great uncertainty – they are typically triangulated from numerous empirical sources of varying data quality, each using a host of approximation techniques. Spatial upscaling of local audits, temporal staggering of material flow from consumption to waste, data interpolation using socioeconomic correlates, etc. are inherently uncertain methods. In Chapter 3, I outline data gaps in waste generation, collection, and recovery data for flexible films, and discuss the challenges wrought by the diversity of plastic products and packaging in use.

The cost of pursuing a recycling outcome can be quantified on different scales (system costs and unit costs). I also distinguish between total costs and net costs (derived by subtracting total revenue from total costs). System costs are difficult to estimate, as they require bottom-up modeling of all value chain components of the system: this is undertaken for the US PET bottle recycling system in Chapter 2. Unit costs are typically normalized on a per-lb or per-tonne basis and are typically derived from facility-level estimates in techno-economic analysis in literature studies. In Chapter 2, I create a system as a network of recovery and recycling facilities and normalize total and net system costs by total recycled quantity to derive total unit costs and net unit costs. The price of recycled plastics is also defined on a per-tonne basis, to allow comparison with unit costs. In Chapter 3, where the flexible film recycling system is still in its early stages of development, literature estimates of unit costs of collection, recovery, and recycling operations are used. Beyond collected or recycled quantity, the cost of recycling can also be normalized by the benefits achieved. Such a normalized cost serves as a yardstick for the economic efficiency of a strategy and enables comparison across strategies to achieve an objective. In Chapter 2, the net cost of greenhouse gas (GHG) emissions abatement is estimated (in cost per tonne CO_2eq) and compared to the carbon price corridor to contextualize the role of recycling in reducing emissions from plastics production.

Recycling can offer several environmental benefits. Waste and litter reduction motivate why recycling is championed. Further upstream, decreased primary production is associated with various environmental benefits presented in life cycle studies (Lazarevic et al., 2010), with greater emphasis on projected GHG emissions reductions to link recycling to climate action. In this dissertation, I quantify GHG emissions as the environmental benefit metric of choice because it is well-researched in literature and can be inferred from energy consumption measures where data is scarce, especially for research-scale novel advanced recycling processes. I do not quantify litter or pollution reduction because not only does the community lack standard methods to estimate plastic litter/pollution, but the correlation between recycling and reduced litter or pollution is also unclear (Chaudhary et al., 2021). Instead, diversion from landfills reported as a recycling outcome can be considered an end-of-life benefit metric.

Linking more and better: when is more actually better?

Numerous reports have questioned whether recycling plastics is effective at all, and research studies have criticized the premise of circularity. This dissertation starts by acknowledging that recycling is a partial solution to many of the environmental problems caused by growing plastic use. From this point, now that I have defined how I quantify more and better, the purpose of the rest of this dissertation is to quantitively examine the conditions under which more recycling leads to better environmental benefits, and qualitatively discuss the feasibility of those conditions.

Chapter 2

PET Bottles

Disclaimer: Portions of this chapter are based on a manuscript accepted for publication in the Journal of Industrial Ecology by Basuhi R et al., titled "Evaluating strategies to increase PET bottle recycling in the US"

In the United States (US), PET packaging enjoys a collection rate of 29% (US EPA, 2020), more than any other plastic type in packaging. To compare, HDPE is collected at 18%, while <5% LLDPE and PP are collected for recycling. Much of this 29% comprises PET bottles[1], defined more precisely as "injection stretch blow molded PET containers" by NAPCOR (NAPCOR, 2019), a national trade organization for PET containers in the US. NAPCOR tracks PET in containers on the shelf, in the bins, in the bales, and in their various recycled forms. Evidenced by the degree of record-keeping, and high diversion rates (compared to other plastics), we can consider PET bottles to have a mature recycling system. It has been operational for over three decades (Welle, 2011); in 1991, the US Food and Drug Administration granted the first 'no objection letter' to the use of post-consumer recycled content in food packaging applications.

A different picture emerges when we compare US's collection statistics with other countries

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[1] Much of PET packaging comprises PET bottles as well. Other uses of PET in packaging include thermoformed containers and PET barrier layer in laminated films. While I could not find US or North American statistics of use share, European sources report that thermoforms and flexibles make up 20% and 7% of PET packaging use (by mass).

around the world. At least 16 European nations have PET packaging collection rates more than 60% - with five boasting > 90% collection (OECD, 2022a): Denmark, Germany, Belgium, Finland, and Lithuania. Outside of Europe, Japan and India also collect 87% and 80% of post-consumer PET bottles respectively.

This geographic difference in system performance for PET collection informs both the methodology and the rationale for an examination of US PET bottle recycling. On one hand, within the US, across plastic packaging types – PET bottles showcase modest recycling success. This can be attributed to a combination of product recyclability, historically supportive policies, and relatively robust end-of-life demand, allowing us to analyze how policies influence plastics recycling. On the other hand, PET bottle recycling in the US is abysmal compared to what's routinely possible in many other countries. We use this comparison and our analysis to understand what it would take to improve recycling and circularity in PET bottles in the US.

2.1 Context, Background

2.1.1 PET bottle recycling in the US

In the US, two types of reverse logistics mechanisms for PET bottle packaging support the PET recycling system. 73% of US residents[2] have access to curbside collection of recyclables (The Recycling Partnership, 2020) – an optional or automatic service, managed by local governing bodies, or contracted to commercial waste management services. Mixed recyclables – paper, metals, plastics – can be collected all together in a single stream or partitioned in dual and multi-stream programs (usually to keep the paper waste stream less contaminated). PET bottles are just one of the many plastic packaging types eligible in such curbside programs – PET thermoformed trays, clamshells, HDPE bottles, jugs and jars, PP tubs and LLDPE containers are accepted in several local curbside programs (Gendell & Coddington, 2020). Once collected and transported to a Material Recovery Facility (MRF), rigid plastics are sorted from the more abundant paper and metals, and then PET bottles are sorted from other plastic packaging types (Chaudhari et al., 2021). Bales of sorted PET bottles are sold to recyclers.

In addition to the curbside channel, 10 states in the US also have a deposit return system (DRS), also known as a "bottle bill" (and other terms such as deposit refund systems, container deposit systems, and more) where bottles can be returned for a deposit fee that is levied during sale of the beverage. The first such deposit bill in the US, in Oregon targeted cans and glass bottles, and pre-dates the use of PET bottles in beverage packaging (National Council of State Legislatures, 2022). The use of PET for bottles was first patented in 1973 and was not widely used until the 1980s, by which time, five other states had instituted deposit return systems with the objective of collecting recyclable aluminum cans and glass bottles (National Council of State Legislatures, 2022). PET bottles in these states benefited from existing incentive-based reverse logistics system. States differ in how they handle

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[2] In smaller, less populated counties, where curbside provisions are not supported, residents can be encouraged to drop-off their recyclables under drop-off recycling programs. Around 21% of the US population only has access to drop-off programs. With patchy reporting, we found that such programs led to negligible collection of plastics in MA and NC.

financial responsibilities, but the logistics of return is same: consumers redeem the deposit-fee incentive at select retail centers or dedicated deposit centers and the empty container is sent on to be recycled. Because returned PET bottles are sorted at the source, deposit-sourced PET bottle bales are less contaminated than their curbside collected counterparts (Welle, 2011). They also skip the sorting step and are directly sold as a 'higher quality' bale to recyclers.

Below, I visualize the flow of PET bottles from the point of waste generation to recovery and recycling in the US in a Sankey diagram (Figure 2-1). By partitioning into states that have DRS and states that do not, we see that the former generates 790 kt of PET bottle waste, of which approximately 300 kt (83%) is collected via DRS, 220 kt is collected by curbside collection, and 270 kt is sent to trash. The states without DRS cumulatively generate approximately 2250 kt of PET bottle waste, of which only 340 kt (34%) is collected through curbside channels while 1900 kt is sent to trash.

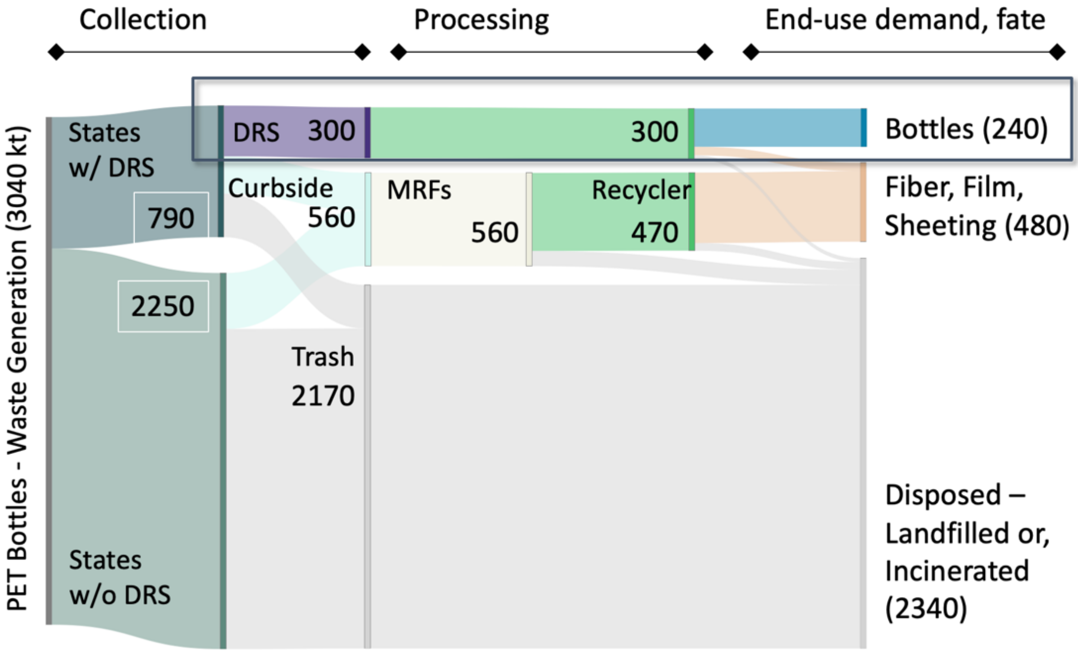

Figure 2-1: 2018 PET bottle material flow at end-of-life. DRS = Deposit Return System, MRF = Materials Recovery Facility. Data from publicly available reporting by NAPCOR, US EPA and The Recycling Partnership, 2020.

After collection and recovery, PET bottles are reprocessed, and used in various end-uses such as food and non-food grade bottles, sheets, fiber for textiles and other applications (NAPCOR, 2019). I partition this demand as circular (bottle back to bottle) and non-circular (bottle to non-bottle) applications. PET derived from DRS sources (purple) is preferentially used for circular bottle applications because it is less contaminated. Some DRS sourced PET bottles (55 kt) are also used in non-circular applications such as fiber, film, and sheeting along with some unavoidable processing losses that are disposed of. Curbside collected PET bottles (light blue) must be sorted and preprocessed at MRFs where losses due to contamination and sorting inefficiencies can reduce R-PET processing yields by 15% (560 kt to 470 kt). Imperfect recycling and mismatch in demand quality requirements further lower the quantity of curbside-derived R-PET used for non-bottle applications to approximately 425 kt. In 2018, two-thirds of the recovered PET bottles were used in non-bottle applications (orange, 480/720 = 67%; 720=240+480), which are seldom recovered for recycling. In 2021, a press release by NAPCOR indicated that the trend has more recently reversed (NAPCOR, n.d.) – more PET bottles were used as recycled content in bottle end-uses than other applications in 2020 and 2021 but actual numbers were not reported.

The net recycling rate is approximately 23.7% = (240 + 480)/3040, and the net collection rate, which is widely reported and does not include losses along the value chain, is about 28% = (300 + 560)/3040[3]. With 2340 kt (or 76.3%) of PET bottle waste generated ending up in landfills or incinerated, I highlight the opportunity for efficiency and improvement in the US PET recycling system.

2.1.2 Quality of R-PET: reprocessing considerations

Compressed and baled waste plastics, from curbside or deposit channels, are graded on quality based on contamination – from PET thermoforms, other plastics, and other materials. While some contaminants such as – polyolefins, PET thermoforms, residual liquids,

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[3] The more recent NAPCOR reporting also lumps in PET thermoforms with bottles (64 kt collected in 2021) and combined recovery rates for both types of PET packaging across US and Canada are estimated as 29%. However, Canada's individual collection rate is 34%.

aluminum cans, and some paper – are allowed at low levels, presence of contaminants such as PVC (polyvinyl chloride), low melting point polymers such as PETG (PET glycol), PS (polystyrene), PLA (polylactic acid), etc. can all lead to direct rejection of entire bales. While consumers can mistakenly place any plastic container in the curbside channel, DRS only accept eligible bottles that are made of PET and PE/PP caps – thereby providing a more predictable bale composition that is easily processed by recyclers (APR, n.d.).

Processors shred the bales, wash the shredded flakes to remove adhesives and organics, and use density differences to separate polyolefins and paper contaminants (Forrest, 2016). Recyclers can sell the shredded flakes as is or process them into R-PET pellets for higher quality. Extrusion conditions use high thermal and shear forces, leading to oxidative degradation of the chains, and lowering the intrinsic viscosity (IV) of the polymer. IV is a flow property that influences processing (and to a lesser extent, mechanical properties) of the product/packaging, and IV specifications differ across applications (Scheirs, 1998) – fiber extrusion for textiles (0.4-0.7 dL/g) is less than film/sheet extrusion (0.7-1 dL/g) as well as injection stretch blow-molded containers like bottles and jars (0.78-0.85 dL/g). Since many carpet and textile end-users acquire R-PET for polyester fiber, they can tolerate some degree of contamination and lower IV. However, this lower IV means that PET from these sources cannot be mechanically recycled back into bottles, earning such recycling activities the label of ‘downcycling’ (Welle, 2011).

To avoid downcycling, pelletizing and post-processing R-PET can be used to reduce contamination and restoring chain degradation. After float separation and drying, three types of contaminants may remain: small-molecule compounds that will be gaseous at extrusion temperatures, and materials that do not melt (metals, paper, etc.), and polymers that melt at temperatures below PET’s melting range. Modern extrusion lines have vacuum degassing units and melt-filters to address the first two types of contaminants (EFSA Panel on Food Contact Materials, Enzymes and Processing Aids, 2022). Contamination by paper particles is detrimental (Alsewailem & Alrefaie, 2018). Likely contaminants in the last category, polyolefins, are only present in small quantities that compatibilize with PET, and even enhance some bottle properties (Itim & Philip, 2015). However, PVC, also in this category, produces hydrochloric acid that provokes uncontrolled chain scission in PET (Welle, 2011). Even without PVC, pelletizing also involves re-extrusion, and chain degradation is inevitable but minimized under vacuum conditions that curb oxidative pathways. Additionally, a post-processing step called solid state polycondensation (Karayannidis et al., 1993) is employed to extend chain lengths and restore intrinsic viscosity.

An interesting aspect of quality is color – a purely aesthetic consideration that does not affect processing or mechanical properties during use. Alvarado Chacon et al. found that particle contamination was linearly correlated with worsening color parameters and the slope depended on type of PET. Source-separated PET (such as DRS) had lower particle contamination compared to PET collected with other materials, and therefore, produced clearer bottles with no yellowing (Alvarado Chacon et al., 2020). Studies also link discoloration to polyamide contamination from barrier layers (Berg et al., 2016). A more direct source of color is the presence of colored PET bottles – but such bottles are sorted into the colored PET bale at the MRF. Hafsa assesses recyclers' willingness to pay for/accept colored PET bales and shows that demand-price relationships for PET critically depends on color (Hafsa, 2022). Lower value for colored PET bales also indicates downcycling.

Quality is a continuous property across multiple dimensions – color, intrinsic viscosity, mechanical and processing properties all influence the final demand for R-PET and are only partially mapped by studies scattered in literature. Thoden van Velzen and colleagues study the impact of 10 contaminants on R-PET properties and find implications for crystallization (Thoden van Velzen et al., 2016). They deduce that EVOH cross-links with PET, PVC and ink particles causes chain degradation along with yellowing from degradation products, and PE makes the recycled PET hazy. Their survey of compositions of waste PET bottles from various sources in Netherlands also supports Eriksen et al's observations on cleaner deposit streams in the Danish context (Eriksen et al., 2019). López et al. (López et al., 2014) performed repeated recycling experiments to show that cleavage of ester bonds decreases molecular weights, viscosity, dynamic moduli, and toughness. Curtzweiler et al. investigate the mechanical properties of PET sheets and find increase in stress at proportional limit as well as stress at yield in the machine direction when 40% recycled content is used (Curtzwiler et al., 2011). Lee et al. study blends of recycled and virgin PET in fiber applications and observe that recycled PET has lower thermal stability but higher crystallization rates (desirable in fiber spinning), and a 30/70 blend shows virgin-like mechanical properties (Lee et al., 2021). For bottle-grade PET, numerous studies also evaluate solid state polycondensation reactions (Cruz & Zanin, 2006; Fitaroni et al., 2020), identify chain extender candidates (Tavares et al., 2016) and report restoring molecular weight and intrinsic viscosity losses incurred in the recycling process (Duarte et al., 2016; Rastin et al., 2016).

To simplify our assessment, and constrained by information available, we approximate this multi-dimensional quality continuum as two quality levels – a ‘high’ food-grade quality fit for circular end-use as well as non-circular end-uses and a ‘low’ grade suitable only for non-circular uses. Given more information, the supply-demand interaction method can be extended to many more quality levels. We also restrict the high food-grade quality to deposit-sourced bottles.

2.1.3 Deposit Return Systems (DRS)

DRS for beverage packaging is not unique to the US: globally, there are 50 bottle deposit return systems (including 10 in the US), covering approximately 400 million people worldwide (Reloop Platform, 2023). Different systems differ in types of PET packaging covered and excluded, ownership of collected material, system finance, distribution of financial responsibility, handling of unredeemed deposits, and deposit fees (OECD, 2022a). While all these factors can be qualitatively understood as influencing return rates in a system, only deposit fees can be quantitatively linked with the available information. Therefore, I plot and study the relationship between return rate (also called redemption rate) and deposit fees (Figure 2-2).

I find that higher deposit fees are correlated with higher redemption rates. Adjusted for price parity, US has a lower deposit fee of 5-10c on average, compared to many EU nation-states. Figure 2-2 presents a rough approximation of the expected redemption rate outcome

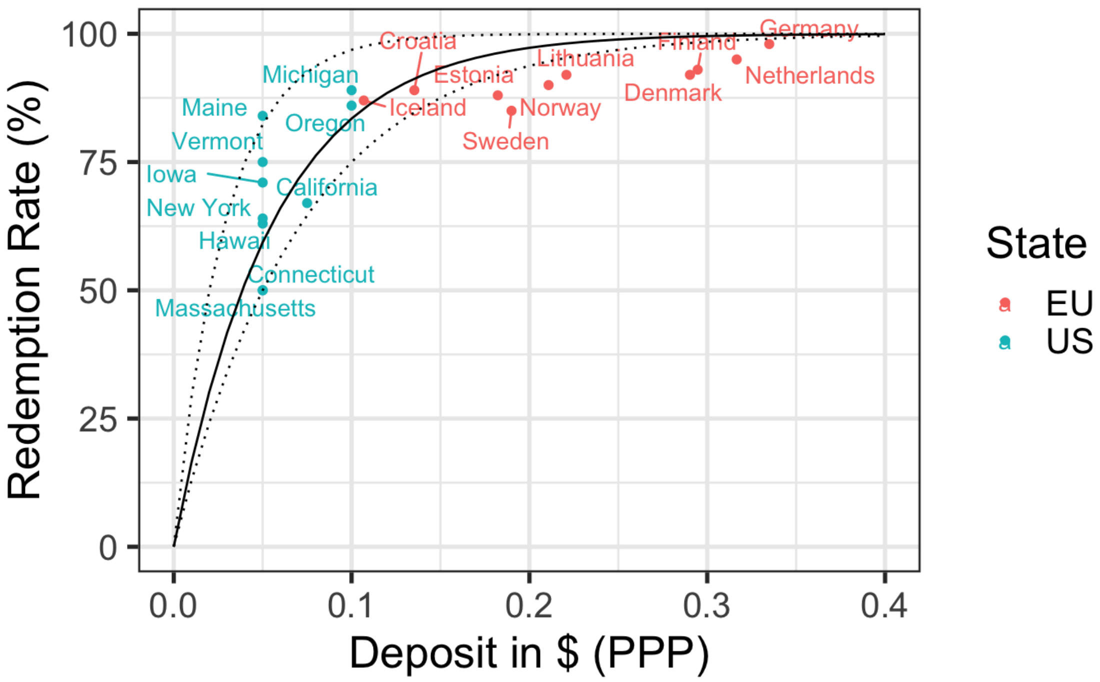

Figure 2-2: Deposit return system redemption rate as a function of deposit fees for 10 states in US (blue) and 10 countries in EU (red). Redemption rate denotes the fraction of eligible PET bottles returned by consumers to acceptable redemption locations such as participating retail centers or dedicated deposit centers to redeem the deposit fees; the deposit fee is charged upfront during the sale of the beverage. Data from 2019-2020 (Reloop Platform, 2020). PPP = Purchasing Power Parity

of a deposit return system based on deposit fees; significant variability in redemption rates for seven states at 5c as well as similar redemption rates for multiple states in the 80% range both indicate that factors beyond just the consumer incentive play a role in the success of a deposit return system. However, institutional, and logistical factors as well as social norms vary widely across and within states, so I base my analysis on this preliminary estimate of the deposit-fee redemption rate relationship.

Deposit return systems are not unique to beverage packaging either: lead-acid batteries, motor oil, waste tires, and electronics have similar take-back systems in place in many states. As a result, there exists significant theoretical scholarship on the cost-effectiveness of deposit return systems in waste management policy (Walls, 2011). In this policy literature, researchers argue that when the cost of providing disposal services is covered by local taxes, each household's decision to generate waste is decoupled from their contribution to tax revenue that pays for the disposal of said waste (Porter, 2010). In other words, households face no disincentive to generate additional trash (Palmer & Walls, 1997). Therefore, options to mitigate the disposal externality include (1) taxing disposal directly, (2) encouraging diversion from disposal by subsidizing recycling and (3) taxing production of products that incur disposal costs (also called, advanced disposal fee). Reviewing downstream policies, many scholars state that a Pigouvian tax on legal disposal is likely to incentivize illegal, unsafe burning or littering (Palmer & Walls, 1997), and be politically unpopular. Still, numerous unit-based disposal pricing policies have been instituted, and a meta-regression analysis finds that weight-based unit fees are effective in reducing waste disposal – especially when a compostable stream is available (Bel & Gradus, 2016). However, for lightweight and non-point-source pollution such as post-consumer plastic waste, monitoring and enforcement of plastics-specific disposal taxes presents numerous administrative challenges (Calcott & Walls, 2000). Moreover, recovery facilities and recyclers do not condone diversion from trash if it contaminates the quality of their "recyclable" stream, and option (1) – taxing disposal is not favorable for plastics. Palmer, Sigman, and Walls compare a deposit return system against options (2) and (3) and find that a "deposit-refund can achieve specified waste-reduction targets at roughly half the marginal social costs of a recycling subsidy or an advanced disposal fee" (Palmer et al., 1997).

A deposit-refund system provides incentive for both source reduction and recycling, provides higher quality PET bales, and engages the consumer as a "supplier" in the waste value chain directly. In theory and practice, deposit return systems are an attractive option. Therefore, in this chapter, I investigate the costs of expanding deposit return systems across the US. I additionally explore what these costs mean for producers, consumers, public actors, and materials recovery facilities.

2.1.4 Supply-demand modeling of recycled plastics.

In economics, interaction between supply and demand is the fundamental principle that governs prices and quantities of goods in a market. Supply represents the quantity of a product that producers are willing and able to offer for sale at various price levels. Demand reflects the quantity of a good that buyers are willing and able to purchase in the market, considering their preferences, and the prevailing market conditions. Supply curves and demand curves are mathematical abstractions of these concepts and price elasticity is used to approximately characterize how quantities supplied or demanded respond to changes in price (Nelson, 2013). For instance, inelastic supply indicates that sellers remain insensitive to price changes, and an elastic demand curve implies that buyers' decision to buy is responsive to price changes. Econometric data analysis informs estimates of price elasticities (Baltagi, 2021) – studies find that short run elasticities (in the short term) depend on time required to adjust production and consumption patterns while long run elasticities reflect technological changes and market regulations (Nelson, 2013).

Recycling is an economic activity. However, historically, recycling of some materials was spurred by market forces, while others were urged by waste management policies. For many scrap metals, "recycling has been undertaken for virtually as long as these materials/metals have been used, and typically in the absence of any policy intervention" (Söderholm & Ekvall, 2020) and scholarship on supply-demand modeling of scrap aluminum, copper, etc. explores how scrap and primary markets interact to inform prices (Mikesell, 2013; Zink et al., 2018). A discussion of recycled plastics markets is different in two ways – (1) recycled plastics have been pushed on to the market by local collection efforts, rather than pulled by interested demand actors, because virgin plastics are cheaper and more abundant than primary metals, and (2) recycled markets for plastics are dwarfed by the much larger virgin markets (because recycling rates are low) and any interaction is one-sided i.e. while virgin prices (dependent on oil prices) critically affect recycled markets, the converse is not observed (Jiang et al., 2015; Selmi et al., 2022). Supply-demand modelling of recycled plastics such as PET must account for these differences.

Research, policy, and practitioner literature alike agree that understanding and stimulating the market for recycled plastics is critical. But market-based studies are rare. In our literature review, we only came across one study that uses linear demand and supply curves to understand the impact of China's waste import ban on plastic recycling in Japan – but the authors only infer qualitative trends (Kumamaru & Takeuchi, 2023). In most economic studies of plastic waste recovery, waste sorting (Cimpan et al., 2016) and recycling options (Larrain et al., 2021; Singh et al., 2021; Volk et al., 2021) are investigated independent of common collection models (Valenzuela et al., 2021) and remain divorced from downstream demand conditions (Kahlert & Bening, 2022). A detailed system-wide environmental and socio-economic study (Andreasi Bassi et al., 2022) of PET recycling in the European Union (EU) analyzes future scenarios and highlights the role of demand in secondary markets. However, the study only considers fixed relative market sizes and does not model the price effects of demand. Material flow analysis maps plastic waste generated and recycled in various geographies (Heller et al., 2020; Kawecki et al., 2018; Lee et al., 2021; Nakatani et al., 2020) and comments on potential for circular practices (Klotz et al., 2023), but lacks market-based analysis to inform barriers and technological or policy solutions to those barriers.

2.2 Research Questions

Broader literature on environmental analysis of PET waste (Brogaard et al., 2014; Shen et al., 2010; Smith et al., 2022) acknowledges the benefits of recycling and recommends increased recovery (Lau et al., 2020; Meys et al., 2021; Zheng & Suh, 2019). But the means and costs of achieving higher recycling rates are seldom discussed. Policies such as recycled content mandates (De Smet et al., 2019) aim to boost circular recycling practices by setting targets for beverage producers and create demand (Carriere & Beavers Horne, 2020) for high quality R-PET. In the US, the proposed 'Break Free from Plastics Pollution Act of 2021', outlines recycled content targets of 25% by 2025, 50% by 2030, 70% by 2035, and 80% by 2040 ("Break Free From Plastic Pollution Act", 2021). While the future of this bill or allocated enforcement capacity is not certain, California has already instituted the same targets ("California Beverage Container Recycling and Litter Reduction Act", 2019) for 2025 and 2030 and several major beverage producers have also made recycled content commitments (Ellen MacArthur Foundation, 2021) that are aligned with the 2030 trajectory. In fact, for the near term to 2025, not enough PET bottles are being collected to meet this policy-driven demand for R-PET (Kahlert & Bening, 2022; Schneider, 2021). Therefore, in this chapter, I inquire: what is the net cost of expanding supply to meet policy-driven demand for circular R-PET in the US? This question is further broken down into three parts:

1. What is the net cost of expanding deposit return systems to meet recycled content mandates proposed for 2025, 2030 and 2035?
2. How does the net system-wide cost depend on demand factors?
3. How is the responsibility of the cost burden shared among stakeholders?

The answer to part 3 is then further contextualized by understanding how EPR fees and design parameters modulate demand for R-PET, and finally, environmental benefits are weighed against this cost to contextualize local waste management concerns against carbon policy.

2.3 Method Overview

Building supply curves for R-PET

In resource economics, material production costs and quantities are aggregated and visualized as supply cost curves. Such curves can then be used to approximate the cost of obtaining excess supply needed to meet demand. When waste is considered a resource for recycling, similar methods can be applied to understand policies and economics of waste recovery (Calcott & Walls, 2005; Porter, 2010). In this work, we model the US PET recycling system – including collection, sorting, and recycling, to build bottom-up supply cost curves for PET bottles. In the US, deposit legislation is a state-level policy action, while curbside collection parameters vary widely between counties and municipalities within a state. We include necessary granularity to capture state-level and local-level details. We estimate deposit collection costs from state-level deposit handling fees (see A.2). We regress a relationship between the cost to collect curbside mixed recyclables and local policy parameters based on census variables (detailed in A.3). We establish the costs of sorting, recycling, and transportation through process modeling, facility location, and distance optimization. An ordered list of these concatenated costs, disaggregated by R-PET supplier (municipality or reclaimer, for example), creates a deposit and curbside supply cost curve (like shown in Figure 2-3) that informs supply economics. All the data used in the study is summarized along with methodological details in the supplementary information.

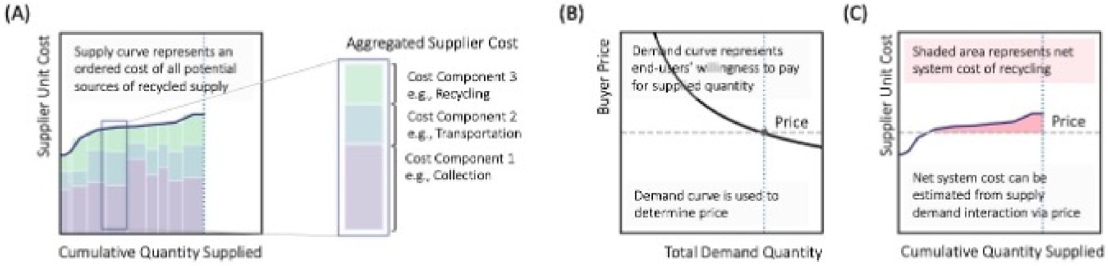

Figure 2-3: **Supply-demand interaction methodology.** (A) Supply curve schematic with cost components. The supply curve orders all material suppliers in increasing order of aggregated cost, which includes waste collection and processing steps. The cumulative quantity supplied forms the horizontal axis, while the marginal unit cost of each supplier forms the vertical axis. The width of each bar is the quantity produced by a supplier (e.g. a sorting facility) and the height of each bar is the cost of the supplier per tonne of final recycled output. (B) The demand curve represents the between R-PET price and the quantity of R-PET demanded by consumers at that price; the dotted vertical line is the total cumulative R-PET supply inferred from the supply curve. Price is estimated from the demand curve and is represented by the horizontal line. (C) Supply-demand interaction is used to evaluate net system costs. The net cost of a recycling system is the area shown in pink and represents the difference between the unit cost and R-PET price for all loss-making suppliers.

Estimating demand curves for R-PET

Resource demand is usually modeled as demand curves that abstract expected changes in consumption as a function of price using elasticity estimates. However, demand for recycled PET is not only dependent on virgin PET markets (OECD, 2018) but is increasingly policy-driven as well (Kahlert & Bening, 2022). To accommodate both market and policy factors, we model demand for R-PET as a function of virgin PET prices, and recycled content requirements that mandate minimum recycled content usage in PET bottles. This functional relationship and related data are further elaborated in appendix B.2. We further include quality considerations by delineating demand by grade: bottle end-users need high-quality (food-grade) R-PET and therefore limit their use to deposit-sourced recycled material while non-bottle end-users can use any R-PET source available. Correspondingly, two R-PET prices are estimated – a food-grade R-PET price and a low-grade R-PET price.

Supply-demand interaction in future policy scenarios

Recycled material supply is largely price inelastic (Mansikkasalo et al., 2014); this is because waste generation and recycling activities are pursued independent of recycling market forces. In the case of PET bottles, where post-consumer recycling is often a municipal service, consumers are unaffected by PET bale prices. Moreover, as a critical component of a city or state's waste management strategy, post-consumer material recovery is subsidized through public funds when unprofitable. Therefore, in our methodology, suppliers determine R-PET quantity (Figure 2-3 A), and buyers determine the price at which the total supplied R-PET is purchased (Figure 2-3 B,C). When R-PET price decreases, demand increases as buyers are willing to purchase more recycled material. Therefore, price and quantity have an inverse relationship. Consequently, when more R-PET quantity is supplied, R-PET prices fall. Therefore, the success of supply-side interventions critically depends on sustained demand for R-PET. Because future demand is highly uncertain, we also explore sensitivity of our results to key demand factors – virgin PET price, recycled content requirements, and demand for non-circular end-uses. We vary virgin PET prices between 1000 and 1800 USD/tonne (ICIS Chemical Business, 2019) and recycled content between 40-80%.

We analyze hypothetical supply increases via deposit system expansion to meet the demand set by policy-based recycled content targets (25% recycled content by 2025, 50% by 2030, and 70% by 2035). The combination of demand and supply side policies form a 'circularity roadmap' for increasing PET recycling. Figure 2-4 visualizes these scenarios. First, in 2025, all 9 states with pre-existing deposit programs increase the value of the deposit fee to 10c and include all PET bottles under the deposit system. In 2030, the remaining 39 states implement a 5c deposit fee on all bottles, and in 2035, all states have a 10c deposit fee on all PET bottles. This combines a 'national beverage container program' with a refund value 'not less than 10c', as proposed by the Break Free from Plastics Pollution Act ("Break Free

From Plastic Pollution Act", 2021).

Metrics used to evaluate future policy scenarios

We evaluate scenarios across four metrics: net system costs, R-PET cost gap, circularity rate, and recycling rate. Net system cost estimates the total subsidy needed for both the curbside and deposit systems to meet all the costs that exceed the revenue from the sale of recycled products. The average unit cost of producing R-PET is partially met by the R-PET price (paid by R-PET buyers). The difference between this average R-PET cost (per tonne) and the price is the R-PET cost gap. In the long run, if the R-PET cost gap widens and is not subsidized, components of the wider recycling system can shut down. Circularity rate is the proportion of generated bottle waste that is recycled into bottle end-use. Recycling rate is the proportion of generated bottle waste that is recycled for use across both bottle and non-bottle end uses. Taken together, these four metrics summarize scenario outcomes and explain the effectiveness of implementing demand-side policy strategies in stimulating the R-PET market.

2.4 Results

We visualize demand and supply cost curves for the baseline and future hypothetical scenarios in Figure 2-4 B,C. We use system costs and recycling rates in 2018 for calibration and report projected results against the 2018 baseline. First, in section 2.4.1, we estimate the recycling rates, circularity rates, R-PET price, and R-PET cost gap for the baseline and three DRS expansion scenarios in the PET circularity roadmap. The success of supply-side interventions such as DRS critically depends on sustained demand for R-PET, as demand determines the R-PET price, and R-PET price impacts net system costs. Because future demand is highly uncertain, we also explore sensitivity of our results for the 2035 nationwide DRS scenario to key demand factors. In section 2.4.2, we vary virgin PET price and recycled content requirement for circular bottle end-use, and in 2.4.3, we study the effect of increased demand for non-circular end-uses such as recycled polyester. An important consideration in implementing DRS is the negative impact on material recovery facilities

(MRFs) that rely heavily on PET and aluminum bales for revenue, and are vital components of the overall recycling system (Waste Dive, 2022). In section 2.4.4, we estimate the impact of DRS implementation on MRF profitability. Finally, extended producer responsibility (EPR) for plastic packaging has increasingly become central to the discussion of recycling costs and economic viability in many states in the US (Sustainable Packaging Coalition, 2022). In section 2.4.5, we quantify the EPR fees that producers need to be charged in different scenarios such that the fees cover the costs of recycling PET bottles.

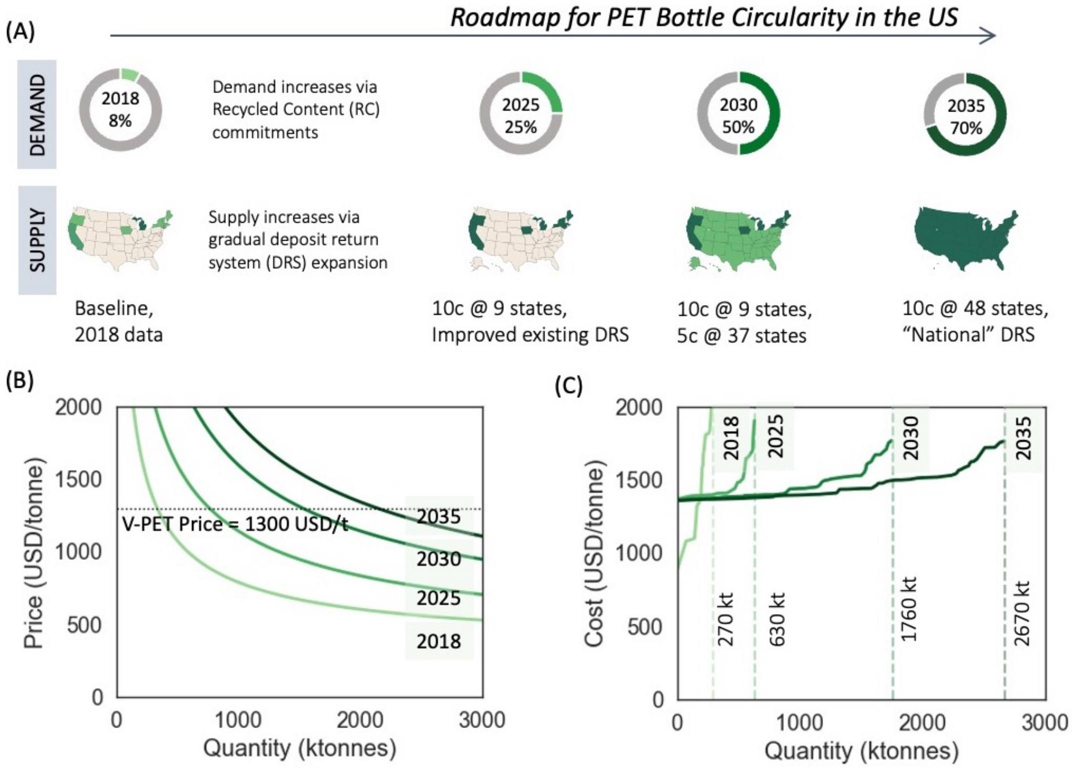

Figure 2-4: Roadmap for PET Bottle Circularity in the US. (A) Circularity roadmap following recycled content targets to expand supply using DRS (deposit return system) expansion to 48 contiguous states in the US. Detailed scenario supply and demand parameters are listed in appendix A. Recycled content targets are informed by proposed federal legislation in Break Free from Plastic Pollution Act ("Break Free From Plastic Pollution Act", 2021) (B) Demand evolution with recycled content targets moving the demand curve to higher quantity. A long-term average of 1300 USD/tonne is used to construct bottle end-use demand for 2025, 2030, and 2035 along with the respective recycled content targets. (C) Deposit supply curves show DRS expansion for scenarios in the roadmap.

2.4.1 PET circularity roadmap scenario cost analysis

In 2018, 720 kt of R-PET is placed on the market (recycling rate is 23.6%) and only a third (240 kt) is used for bottle end-use (Figure 2-5A). We find PET recycling unprofitable at a total net system cost of 210 million USD (Figure 2-5B). Handling fees for deposit centers and municipal collection and processing fees paid to MRFs are the implicit subsidies that balance this cost burden. Food grade R-PET price of 1430 USD/tonne is lower than V-PET price (1600 USD/tonne), and a R-PET cost gap of 450 USD/tonne is estimated for the deposit collected R-PET (Figure 2-5C).

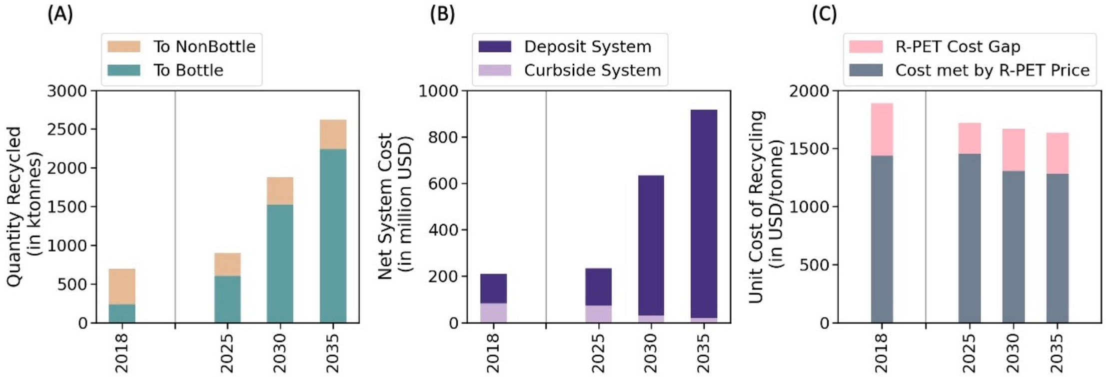

Figure 2-5: DRS expansion quantities and cost. (A) Quantity recycled to bottle and non-bottle end uses, (B) Net system cost for curbside and deposit channels in 2018, 2025, 2030, and 2035 scenarios, (C) Food-grade R-PET price and R-PET cost gap; R-PET cost gap is the difference between average cost of supplying R-PET and R-PET price determined by demand. The virgin PET price is $1600/tonne in 2018 and $1300/tonne in future scenarios.

Implementation of state-wise deposit system expansion and fee increase drives up R-PET supply and recycling rates (Figure 2-5A). The PET recycling rate increases from 23.6% in 2018 to 82% in 2035 (PET recycled quantity increases from 720kt to almost 2650 kt) and the circularity rate improves more than 8x (8% to 68%). Demand from non-bottle end-use is kept constant and as a result, the fraction used for non-bottle end-use declines from 67% in 2018 to 14% in 2035 (Figure 2-5A). Due to the higher quantity of recycling in the national DRS scenario, annual net system cost increases from 210 million USD in 2018 to 920 million USD in 2035 (Figure 2-5B). However, we find that the cost per tonne of PET recycled is lowered (Figure 2-5C), due to economies of scale. The recycling system still requires subsidies, but the R-PET cost gap is lower (down from 450 USD/tonne in 2018 to 360 USD/tonne in 2030, 2035).

2.4.2 Sensitivity to PET bottle demand factors

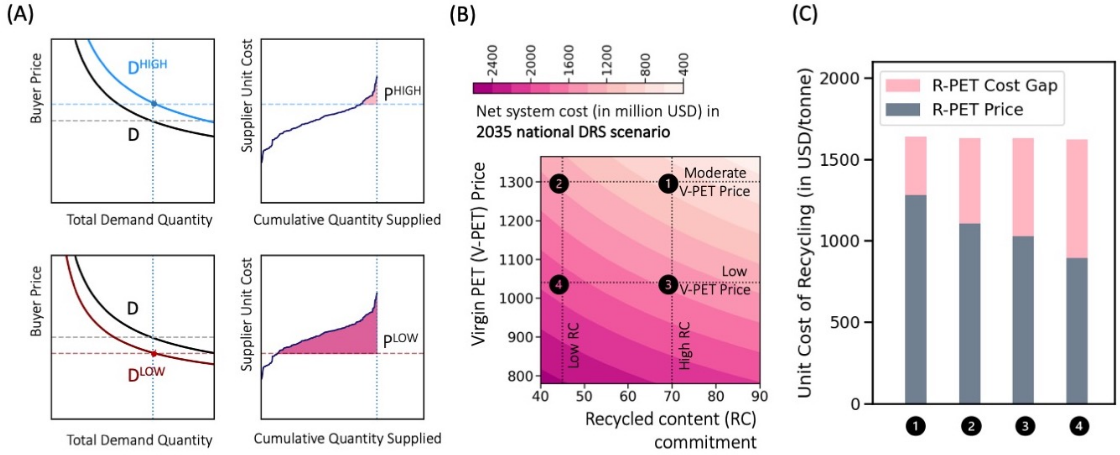

Figure 2-6: (A) Schematic showing effect of changing demand on R-PET price and net system cost of recycling. Demand for R-PET is influenced by V-PET price (lower V-PET price lowers R-PET demand) and recycled content commitments (higher commitments mean higher R-PET demand). When R-PET demand is high (D_{HIGH} compared to D), R-PET price for a given quantity of supply increases. Higher R-PET price P_{HIGH} covers more of the cost of supply and decreases net cost of recycling. When R-PET demand is low (D_{LOW} compared to D), R-PET price is lowered (P_{LOW}) and net cost of recycling increases. (B) Total net system cost (in million USD) required to support demand for 2035 national deposit return system expansion supply scenario under different virgin prices and recycled content (RC) commitments. (C) Unit cost of recycling (per tonne of R-PET produced) differentiated into R-PET price and R-PET cost gap for selected demand cases: 1 = 70% RC, 1300 USD/tonne V-PET Price, 2 = 45% RC, 1300 USD/tonne V-PET Price, 3 = 70% RC, 1040 USD/tonne V-PET price and 4 = 45% RC, 1040 USD/tonne V-PET price.

The net system costs are sensitive to demand conditions. The trade-off between R-PET price paid by buyers and the R-PET cost gap, usually covered by government, is modulated by R-PET demand (Figure 2-6A). R-PET demand may fall short of the targets outlined in the circularity roadmap, due to a) the availability of cheap and consistently high-quality virgin PET and b) changes in recycled content targets set by policy. V-PET prices are volatile – between 2018 and 2021, prices have varied between 1000 USD/tonne and 1800

USD/tonne (ICIS Chemical Business, 2019). As the virgin sector loses market share to recycled products, V-PET prices may decrease, eroding R-PET demand and increasing net system costs.

Net system costs can more than double from 920 million USD (Figure 2-6B, case 1) to 1.94 billion USD (Figure 2-6B, case 4) if V-PET prices near 1000 USD/tonne and the recycled content target drops 50%. In such cases, R-PET price paid by buyers only cover 55% of the cost of collecting and processing R-PET (Figure 2-6C), and a cost gap of 770 USD/tonne is expected. Maintaining demand for R-PET is critical to improving recycling and circularity rates in a cost-effective manner – policies such as strong recycled content mandates and taxes on virgin material usage support R-PET demand when voluntary commitments fail, or virgin prices fall.

2.4.3 Sensitivity to recycled polyester demand

Globally, polyester makes up over 50% of the fiber used in textiles (Textile Exchange, 2022) and is chemically similar to PET used in beverage bottles. Sustainability commitments made by clothing brands (Textile Exchange, 2021) can increase R-PET demand from bottles, as textile-to-textile recycling remains underdeveloped (Juanga-Labayen et al., 2022). We simulate a 100% increase (relative to 2018) in demand for R-PET fiber for the textile sector, competing with circular bottle-to-bottle recycling. Increased competition for the same recycled material raises prices. We find that the resulting increase in R-PET price can cover the costs of the deposit system expansion to a greater extent and bring down the total subsidy needed from 920 million USD to 508 million USD (Figure B-8 in appendix B). However, the circularity rate drops from 70% to 52% in the 2035 nation-wide DRS scenario. When multiple end markets exist, competing interests can distort the effect of circular policy tools. For instance, a recycled content mandate for bottles may lose its utility if non-bottle end uses capture most of the available supply. Bottle manufacturers concerned about rising competition for material can reduce their materials costs and increase recycled content rates by supporting extended producer responsibility (EPR) policies that allow them to retain ownership of post-consumer PET.

2.4.4 Extended producer responsibility (EPR) fees

Currently, the net cost burden of recycling is largely borne by taxpayers: municipalities subsidize collection and MRF processing while states often pay handling fees for deposit systems paid to retailers and deposit centers. Packaging extended producer responsibility (EPR) laws, already set up in four states (Colorado, Maine, Oregon, and California) with varying degrees of stringency and scope (Sustainable Packaging Coalition, n.d.), seek to transfer the externalized cost burden of waste management and recycling back to packaging producers. Implementation of EPR laws can enforce a financial obligation in the form of EPR fees from the producers. At a minimum, the EPR fees must be designed to cover the R-PET cost gap we calculate in our analysis.

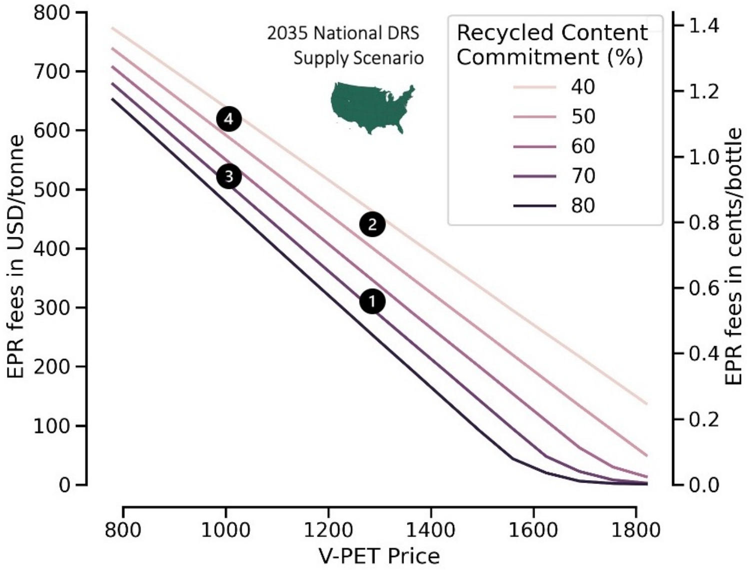

Figure 2-7: Extended Producer Responsibility Fees. Estimated EPR fees (expressed both in USD/tonne and cents/bottle) to cover total net system costs for the 2035 national deposit return system scenario as a function of virgin PET price and recycled content commitments EPR = Extended Producer Responsibility. Assumed average weight of a PET bottle= 31.7g (see appendix B.5)

For the 2035 scenario, we estimate the value of EPR fees required to cover the cost gap (not including administrative charges) as a function of virgin price change and recycled content commitments (Figure 2-7). In the circular roadmap to 2035 (70% recycled content target and 1300 USD/tonne V-PET price – case 1 in Figure 2-7) under an EPR regime, EPR fees for beverage producers would be 300 USD/tonne of waste generated from their bottles. This cost falls within the wide range of EPR fees implemented in Europe: 330 EUR/tonne for virgin PET bottles in 2021 by France's CITEO (CITEO, 2021); 104 EUR/tonne for transparent colorless PET bottles and 596 EUR/tonne for transparent colored PET bottles in 2022 for Belgium's Fostplus Green Dot program (Fostplus Belgium, 2022). If beverage producers pass this cost to consumers, it is approximately 1c per PET bottle placed on the market (see calculation in appendix B.6.5). This is in addition to a 10c redeemable deposit fee. Previous literature has shown evidence of consumers willing to pay a small price for sustainable packaging (Herrmann et al., 2022) and that deposit fees have not dissuaded consumption (Reloop Platform, 2023).

Low V-PET prices necessitate high EPR fees to fund the increasing cost gap. When V-PET price is low at 800 USD/tonne, the fees need to be as high as virgin prices (left-most part of Figure 2-7). If fees are not sufficiently high, beverage producers and bottle manufacturers may find it cheaper to use virgin material instead of buying R-PET. Policies that ensure high V-PET prices (>1500 USD/tonne) and adherence to recycled content targets (>70%) can help pay for the system without the need for large EPR fees. Eco-modulation of EPR fees based on recycled content has also been discussed (Laubinger et al., 2021) to drive demand for bottle-grade R-PET while ensuring that the cost of expanded supply is met.

2.4.5 Impact on curbside collection and MRFs

When a state institutes a deposit return system, residents are more likely to take the PET bottle back to the retail store or a dedicated deposit center instead of placing it in the curbside recyclables collection bin. This effect is already seen in states that currently have deposit return systems – only between 2-9% of beverage bottles are collected via curbside programs in the 10 states with deposit programs (Container Recycling Institute, 2022). As deposit systems expand, curbside PET bottle and aluminum can quantities destined for MRFs shrink. Since MRFs rely heavily on PET and aluminum bales for revenue, many have expressed concerns about deposit return system expansion (Waste Dive, 2022).

If MRF processing fee hikes, made to cover for loss in bale revenue, prove untenable to municipal actors, deposit system expansion can impact the recycling of other materials in the curbside stream as well. To investigate this unintended consequence, we estimate the net unit processing costs (MRF costs – MRF revenue normalized by waste input mass) for all MRF-based recyclables. We find that, on average, net processing costs increase by 13% from 54 USD/tonne in 2018 to 61 USD/tonne in the 2035 scenario. Similarly, median net processing costs increase from 76 USD/tonne to 80 USD/tonne. Current processing fees range between 40-80 USD/tonne (The Recycling Partnership, 2020).

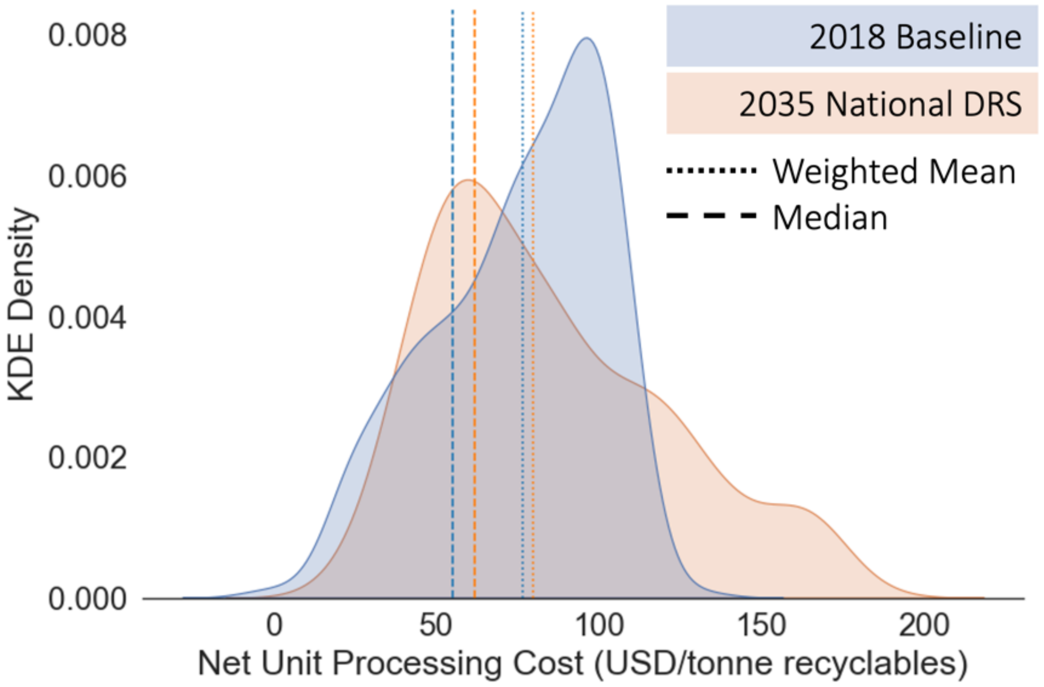

Figure 2-8: Distribution of net unit processing cost (USD/tonne) of simulated MRFs in 2018 and 2035 National DRS scenarios. Weighted mean is weighted by mass of recyclable waste input to each simulated MRF.

However, a comparison of averages or medians belies a more drastic change in distribution across MRFs (Figure 2-8) and we estimate that the total cost of keeping MRFs viable by subsidizing the additional cost increase is approximately 373 million USD, across all recyclables (not just PET bottles). A larger tail extending into a much higher processing fees (150-200 USD/tonne) represents many small simulated MRFs and indicates that reduced bale revenue from valuable material fractions is compensated by decreasing MRF service radius in less populated regions.

This simplified analysis is derived from distance-optimized locations and sizes for MRFs in each state and equates net unit processing costs to processing fees paid by municipalities. While such a simplification does not account for the local politics of waste management, negotiations between multiple public and private parties, contract lengths, siting challenges, or other qualitative considerations, it allows us to approximate the effect of national deposit return systems on local material recovery facilities. We underscore that when crafting legislation for deposit system expansion, recognizing the cost burdens of various stakeholders (OECD, 2022a) in the interconnected recycling value chain is crucial.

2.5 Discussion

In this chapter, I analyze the role of policy-driven demand in improving PET bottle circularity and estimate the cost of expanding supply to meet this demand. I find that national deposit return system can increase PET bottle recycling rates from 24% to 82%, supplying approximately 2700 kt of recycled PET annually. With stability in demand, I estimate that this PET bottle recycling system can achieve 65% bottle-to-bottle circularity, at a net cost of 350 USD/tonne PET recycled. This analysis can inform ongoing efforts to design and implement extended producer responsibility for plastic packaging. On a methodological note, I illustrate how reframing plastics recycling barriers as recycled supply and demand considerations can quantify trade-offs across stakeholders for better decision-making.

The basis of all our estimates are the costs stacked on the recycled supply curve: curbside collection in sub-county scales is linked to material recovery facilities within a state[4] while deposit return systems are implemented state-by-state. Both curbside and deposit channels

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[4] This is a modeling simplification that helps optimization of MRF locations, and not strictly true. Waste is sometimes shipped across state lines for communities near state borders.

send PET bales to plastics recyclers connected on a national scale, supplying R-PET to domestic[5] producers. This supply perspective reveals the distributed spatial scope and structure of waste management decision-making across the waste value chain. PET bottles represent one waste stream, among many, that operate within these waste management structures shaped by decades of policymaking at local, state, and federal levels. I'll now place the method and results of our PET bottle analysis within this wider market and policy context to better understand how these various cost estimates inform stakeholder actions.

2.5.1 Stakeholder implications

Stakeholders in the waste value chain hold different priorities. Local policymakers work within their budgets to provide waste management services to residents, deciding between diversion or disposal for many products and materials, and are constrained by transport logistics and availability of recovery options (Porter, 2010). Material recovery facilities focus on sourcing clean post-consumer waste with a higher fraction of valuable materials that are worth separating (Chang & Wang, 1995). Deposit return systems are reverse logistics channels that states negotiate with retail centers and producers (OECD, 2022a). Recyclers expect high quality bales of PET bottles so they can maximize yields and value of their recycled outputs. Producers wish to grow demand for their products – typically consumer goods such as beverages, apparel, carpets, and more – increasing demand for packaging as well.

Demand for products (and packaging) by consumers translates to derived demand for PET by the producers; the choice between using virgin or recycled PET comes down to the cost of obtaining material that is compatible with existing processing lines optimized for using a specific virgin PET resin. Following this logic, we model the demand for R-PET as 'substituting' virgin PET when a producer is willing and able to use R-PET instead of V-PET. We argue that producers' collective willingness to incorporate more recycled content is influenced most by recycling policy and posit the policy-driven demand for R-PET. Policy-

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[5] Domestic producers can be regional, national or multi-national who use the R-PET in domestic production activities within the US.

driven demand implies that without the policy driver (recycled content mandates in this case study), there is no incentive to increase recovery and recycling rates. We quantify the barrier posed by this lack of incentive in our sensitivity analysis of the nation-wide deposit return scenario. A lower recycled content mandate (of even 50% down from 70% in 2035), signals a weak demand, and increases net system costs from 920 million USD to 1.2 billion USD, and lower virgin PET prices can further erode demand for recycled PET, push this to 2 billion USD. Virgin prices are a significant, but exogenous factor, to the recycled PET market (Selmi et al., 2022). However, high rates of PET bottle recycling, simulated in this study, are likely to lower PET prices in the short-term. Even with policy-driven recycled demand, such virgin-recycled market interactions must be carefully studied as recycling rates ramp up – our approach presents a first order approximation of outcomes of such interactions without endogenous modeling.

With a policy-driven demand of 70% recycled content in PET bottles in 2035 and a steady V-PET price of 1300 USD/tonne, our results suggest that the net cost of the PET recycling system supported by nation-wide deposit return system is 920 million USD. Compared to 2018, our baseline scenario, this cost is 4.4x higher – but raises the recycling rate from 24% to 82% (a 3.4x increase), and bottle-to-bottle circularity rates from 8% to 68% (>9x increase). Currently this net cost is indirectly funded by municipalities and states to offer residents collection services, pay MRFs' processing fees to sort recyclables, and retail centers handling fees to run deposit programs. Moreover, MRFs losing revenue streams such as PET bottles and aluminum adds additional financial pressures. Many municipalities would need to increase processing fees as well. If the processing fees are higher than disposal fees (landfill tipping fees or incinerator gate fees) – from the municipality's perspective – recycling (more broadly, not just PET bottles) is no longer cost-effective. The distributed, burdensome responsibility shouldered by local and state waste management actors can make the 'improved' PET bottle recycling system unattractive and untenable.

Imperfect return rates in deposit systems generate revenue as unredeemed deposits (i.e., when a customer forfeits deposit value by not returning the PET bottle). When a 10c deposit fee is implemented nationwide, we find that close to 1 billion USD in unredeemed deposits will be potentially available (Figure B-9 in appendix B). While this is approximately equal to the total net system cost of 920 million USD, it also offers counterproductive incentives – maximizing this revenue would mean minimizing collection via deposit return systems. Current handling of unredeemed deposits varies widely: Connecticut, Massachusetts, and Maine use it for general funds; New York gives producers a fraction to cover system costs while Iowa allows producers to keep it and operate the deposit system and Michigan uses it to support retailers who handle deposits; others like California and Vermont earmark it for specific purposes such as beverage container recycling fund or clean water programs Reloop Platform, 2020. The degree to which these funds are allocated back to deposit return systems is unclear, but relying on unredeemed deposits to motivate state or municipal actors to enact deposit legislation undercuts the recycling objective. In such cases, the deposit legislation can devolve into a tax on packaging[6], which is suboptimal to appropriately incentivized deposit-return systems (Walls, 2003).

In practice, sub-optimality concerns program design and implementation that are not considered in this study. For example, accessibility and convenience of a deposit return system depends on the type, number, and distribution of collection points (OECD, 2022b) – dedicated but sparsely situated depots dissuade redemption, while mandating all retail centers selling beverages in eligible containers to accept returns can encourage consumer participation. Grocery retailers, a critical component of the reverse value chain, are among the most vocal detractors of deposit return systems (alongside MRFs whose revenue streams are directly cannibalized) (Waste Dive, 2021). Common grievances of retailers relate to compensation: handling fees are neither dynamic[7] nor reflective of the true costs of operating deposit return systems (Calabrese et al., 2021). States also have different configurations of financial and operational responsibilities that can influence return rates and are critical points of contention among stakeholders.

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[6] A packaging tax can be argued to be better or worse depending on what the policy objective is. If the objective is to recycle more, it is arguably worse than a deposit return system. If the objective is to use less packaging, and the tax rate (as a % of product price) is high enough, then it is better than a deposit return system. (Walls, 2003)

[7] For instance, Connecticut, with the lowest redemption rate in the US, has not updated handling fee structures for over three decades since its inception in 1978. Recent legislation that goes into effect in 2024 will increase handling fees, expand beverage types covered, and increase deposit fee to 10c (CT.gov, 2022).

Debates about the cost burden of recovering plastics routinely stump recycling action. If we distill the various arguments, the problem can be simplified to one of responsibility. In the US, currently, the responsibility of packaging waste largely falls on public systems. For the PET bottle case where I quantify the total cost of expanding deposit return systems, I argue that distributed public responsibility across state and local actors is unlikely to incentivize improved PET bottle recycling outcomes. In other words, optimizing for cost is not enough when systemic barriers deter progress – and our analysis quantifies such barriers. Systemic barriers to plastics recycling can be traced to misaligned policy structures, where costs, responsibilities, and benefits are not appropriately linked by incentives (Milios et al., 2018).

2.5.2 Producer responsibility and circular demand

Extended Producer Responsibility (EPR), varyingly described as a law, concept, or principle, aims to shift the cost burden of end-of-life responsibility from public actors to producers. Perspectives differ on the scope of policy objectives: while some limit it to transferring financial responsibility of waste management activities back to producers (in line with the "polluter pays" principle), others insist that EPR must address upstream packaging design to converge towards more better recycling or sustainable outcomes. The latter objective derives from the concept of eco-design[8] and is commonly implemented as eco-modulation of EPR fees – where the fees levied on producers varies according to a predefined criteria relating to sustainable design principles. For example, EPR schemes Belgium and Netherlands vary fees by packaging recyclability, France additionally also varies its fees by recycled content (Laubinger et al., 2021).

In the US, interest in packaging EPR is growing; 4 states have passed legislation implementing EPR for packaging while 11 other states introduced EPR bills in 2023. Fee structures, producer definitions, timelines, and administrative details differ but many include design for recyclability and recycled content among eco-modulation criteria (Sustainable Packaging Coalition, n.d.). In theory, defining fee levels based on recyclability is simple once elements

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[8] Some also extend the 'polluter pays' metaphor and eco-design concept to upstream pollution during production and manufacturing stages.

of product design that compromise end-of-life recovery have been identified. In practice, surveying diverse formats and functions of packaging is tedious. For PET bottles, relatively standard product designs simplify such surveys, and industry initiatives maintain simple design guides drawing from decades of research on recyclability (APR, n.d.; EPBP, n.d.). Varying fees by recycled content, on the other hand, is not straightforward.

Though recyclability and recycled content criteria are both lumped together under 'eco-design' principles, the ways in which they influence the recycling system are different. Improving design for recyclability increases quality and (possibly) quantity of recycled supply; increasing recycled content creates a demand signal. Using policy-driven demand, we quantify this signal and derive EPR fees as the net cost per tonne of recycled PET with a nation-wide deposit return system. We also emphasize that low V-PET prices can distort these demand signals. In our analysis, EPR fees, levied on beverage producers, to meet the net cost of the 2035 PET bottle recycling system can be between 300-650 USD/tonne depending on market and policy factors. If virgin prices are around 1300 USD/tonne and 70% recycled content mandate is imposed, material costs for the producer from both recycled and virgin PET is also approximately 1300 USD/tonne, and EPR fees amounts to a 23% tax on material costs. If virgin PET prices fall and content mandates are lax, the EPR fees needed to maintain the recycling system will have to shoot to 60% of material costs without fees (650 USD/tonne on 1100 USD/tonne). In other words, EPR fees will have to make up for low demand reflected by low R-PET price to keep the expanded R-PET supply viable.

When EPR fees is tied to recycled content, it can foster circular bottle-to-bottle demand. When the costs and benefits are borne by the same group of interested actors (here, producers), recyclability is prioritized. A deposit return system, then, provides a practical mechanism to limit non-recyclable designs, monitor and track returns, and support data-driven decisions on material ownership. Moreover, when industry actors responsible for packaging (usually in consortia such a producer responsibility organizations or PROs) assume financial and/or operational responsibility of deposit return systems, misaligned incentives can be curbed to some extent by adding a packaging tax that is tied to collection rates (provided the packaging tax is more than the unredeemed deposit) (OECD, 2022a).

Therefore, if circularity is the objective, EPR and deposit return systems provide a complimentary policy framework to support circular outcomes. However, the premise that product circularity (such as bottle-to-bottle recycling) is environmentally superior originates from studies penalizing downcycling/open-loop recycling (Huysveld et al., 2022; Rigamonti et al., 2020; Vadenbo et al., 2017) due to quality degradation (Helbig et al., 2022). Geyer and colleagues (Geyer et al., 2016) show that circularity, or closed loop recycling, is not inherently better than open-loop recycling and recommend that recycling policies should be guided by environmental impact reduction potential instead. Several critiques of circular economy initiatives also question the link between circularity and environmental benefits (Harris et al., 2021; Saidani & Kim, 2022). We quantify this link to better clarify recycling objectives.

2.5.3 Assessing environmental benefits: recycling and circularity.

In our PET bottle study, we differentiate demand by circular and non-circular end-uses. We argue, however that, if virgin PET is displaced in both circular and non-circular end-uses, differentiating between circular and non-circular end uses is not meaningful. We simplify non-circular demand[9] as ‘low’ quality, allowing it to absorb curbside collected material. As we limit our analysis to mechanical recycling, from an GHG emissions standpoint, the difference between recycling to circular and non-circular end-uses is then a difference in emissions from (a) collection and (b) sorting. If non-circular/open-loop recycling are assumed to be bottle-to-fiber like, virgin PET processing and recycled PET reprocessing are equivalent as both require solid stating (low viscosity requirements eliminate the need for solid-state polycondensation). Environmental impacts of differences in collection and sorting are minor, compared to the GHG emissions rendered from virgin PET production and bottle manufacturing. Estimates from California suggest that curbside collection produces 0.17 kgCO_2eq/kg PET bottle collected, while drop-off (deposit return system) produces 0.22 kgCO_2eq/kg. Estimates of GHG emissions allocated to PET recovery at MRFs vary: 0.04 – 0.1 kgCO_2eq/kg PET bottle collected are reported in the US context (Chaudhari

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[9] This simplification is not strictly correct. PET used for sheeting, depending on application, can have higher intrinsic viscosity requirements than even PET bottles.

et al., 2021; Uekert et al., 2023) but are low. Mechanical recycling estimates also hold some uncertainty (0.27-0.4 kgCO_2e/kg) but remain far lower than virgin PET production (2.2 kgCO_2e/kg) (Uekert et al., 2023).

The 2035 nation-wide deposit return system can reduce between 56-64% of GHG emissions associated with virgin PET production if virgin PET is displaced. Compared to the 2018 baseline, where 16-18% virgin PET emissions are saved by recycling if virgin PET is displaced, the proposed deposit return expansion curbs 2.3-2.7 MtCO_2e in emissions annually (in 2035) from the PET supply chain, even as PET consumption increases from 3040 kt to 3400 kt. If 82% recycling rate can be achieved, the degree of circularity has no bearing on environmental benefits (here, GHG emissions) reaped. In other words, even if fiber end-uses use more R-PET in the 2035 scenario, if high recycling rate can be maintained, and if virgin PET is displaced, the environmental benefits fall within the same range. Circularity is not environmentally superior, but the value of circularity lies within the first conditional clause. Circular demand aligns incentives, costs, and benefits, simplifies responsibility, and can therefore potentially sustain an efficient recycling system (Calcott & Walls, 2000) with high recycling rates. Put simply, if PET bottles are recycled into open-loop applications, producers are less likely to support recycling activities financially and operationally without strong economic signals (in the form of packaging taxes). Policy-wise, EPR fees tied to collection/recycling rates can and do provide such signals in several EU states (OECD, 2022a). However, without circular demand incentives, if the domestic demand sink cannot absorb all the collected R-PET, either due to its total size or other quality constraints, the policy can lead to excess costs not factored into EPR fees. More drastically, collected R-PET can be stranded without a destination. While the global demand for (mostly virgin) PET used as polyester in fiber applications is double the solid-stated resin used for bottles, within the US, only 0.47 Mt (mostly virgin) PET is produced for fiber compared to 2.1 Mt for bottle applications. Much of fiber processing and textile manufacturing is overseas, with a significant fraction in China. Export of plastic waste has numerous well-documented negative consequences and is both undesirable and heavily discouraged (Brooks et al., 2018). Therefore, combined with higher sensitivity to R-PET price, domestic demand for R-PET from fiber end-uses simply cannot grow R-PET supply. From a GHG emissions standpoint, if non-circular demand exists, open-loop recycling is not inferior even if the end-use application is not recyclable (like textiles). But circular demand is necessary to incentivize and sustain high recycling rates.

Quality degradation and mismatch are common challenges in circular applications, and our analysis simplifies these considerations for analytical tractability. Studies show there is a significant drop in mechanical and barrier properties of R-PET after several extrusion cycles (Nait-Ali et al., 2011). We do not include quality degradation in once-recycled PET bottles and assume that they can be recycled many times. Literature on PET recycling suggests that more than 50% (up to 90%) recycled content inclusion is dilute enough to not cause severe property degradation after many cycles (Brouwer et al., 2020). However, this article only offers theoretical limits based on empirical observations on contamination, and experimental process validation is lacking. For greater percentage inclusion, chemical recycling or monomer recovery (Chaudhari et al., 2021) which can revert PET back to monomers and yield virgin-like PET, must be considered.

The broader supply-demand methodology in our analysis is agnostic to the recycling method used, and I adapt the analysis to include chemical recycling of PET. In this discussion, I extend the existing results by considering high and low circularity cases within a sensitivity analysis of some key variables – the high circularity (60-90%) case uses chemical recycling for bottle-to-bottle recycling while the low circularity case (40-60%) finds mechanical recycling sufficient. On the demand side, the high/low circularity is encoded by limiting/not limiting recycled content inclusion. Virgin price is varied between 1000 USD/tonne and 1600 USD/tonne. On the supply side, nation-wide deposit return expansion is considered, and collection, sorting and reprocessing steps are all kept same. Chemical recycling is simply assumed to have a higher economic and environmental cost as additional steps for chemical depolymerization, monomer recovery and repolymerization must be added. Depolymerization of PET can be achieved by several solvents and enzymatic reagents, and Uekert et al. assess and compare the various processes on cost, energy consumption and GHG emissions basis (Uekert et al., 2023). I choose PET glycolysis for use in this extended analysis, as it has the lowest economic and environmental cost of all surveyed processes. On a per

Table 2.1: Parameter ranges for sensitivity analysis on quality and chemical recycling

Parameters	Min	Max
Virgin Price (USD/tonne)	1000	1600
Demand Pivot for Bottle-Grade R-PET:		
Without Chemical Recycling	0.4	0.6
With Chemical Recycling	0.6	0.9
Additional Cost for Chemical Recycling (USD/tonne)	100	400

tonne basis, chemical recycling via glycolysis is approximately 400$/tonne more expensive and emits 0.8 tCO_2eq more than mechanical recycling (Uekert et al., 2023). With time and greater market share, chemical recycling costs may come down the experience curve, but unless process-heat is eliminated or substituted with low-carbon fuels, emissions will persist. Therefore, I consider the 400$/tonne as the maximum additional cost, going down to 100$/tonne (minimum).

I simulate 300 scenarios, varying all the parameters in Table 2.1. Figure 2-9 shows the results of this sensitivity analysis, with x-es denoting high circularity scenarios and o-s denoting low circularity scenarios. Total GHG emissions abated (on the x axis) for the PET recycling system with nation-wide deposit expansion is higher for lower circularity scenarios with mechanical recycling. On the y axis, we construct a metric to measure the cost of environmental benefit (GHG emissions abated). Given the available R-PET supply, higher circularity costs more per tonne CO_2eq abated and has lower overall emissions abatement. If high circularity is only possible with chemical recycling, low V-PET costs can raise the cost of emissions reduction via recycling to up to 800$/tonne-$CO_2eq$.

To put the numbers in Figure 2-9 into perspective, carbon prices, a common yardstick for emissions abatement cost in climate discourse, fall in the range of 61-122 USD/tonne (World Bank, 2022). In other words, at low and even moderate V-PET prices, then, recycling is a highly cost-inefficient means of emissions abatement. This perspective allows us to re-examine the various problem-frames recycling (and circularity) is invoked to solve. Circular demand can pull material away from landfills, saving landfill space. It can also abate emissions, albeit lesser with highly energy intensive chemical recycling. However, from an emissions reduction standpoint, when chemical recycling becomes necessary, unless virgin

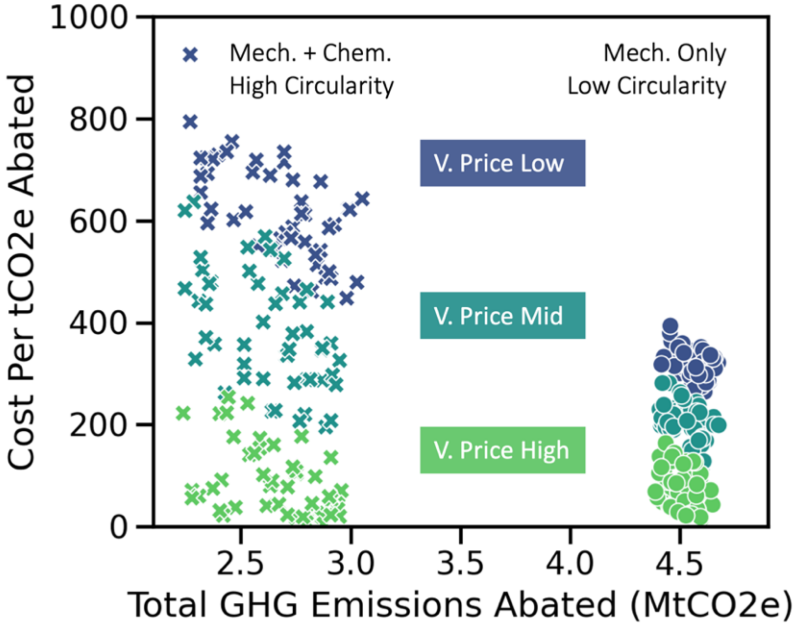


Figure 2-9: Cost per tonne CO_2eq abated and total emissions abated through recycling of PET bottles in the US.

materials are expensive or are taxed separately, circularity is more expensive per unit of abatement achieved. This result urges us to rethink the ends to which we justify or motivate circularity –at least in the PET bottle case, our analysis provides a quantitative measure of costs and benefits to inform this rethinking.

Estimating emissions abatement from recycling involves an important assumption that cannot be taken for granted: displacement. A simple calculation suggests that an 82% PET bottle recycling rate would mean that up to 60% of domestic V-PET production capacity will have to be displaced by new recovery and recycling infrastructure. While this presents a significant opportunity to decouple from virgin feedstocks in the long term, there is no past evidence of production capacity of virgin plastics shutting down due to recycling. Gray literature suggests that low V-PET prices are weathered by virgin PET producers better than recyclers (OECD, 2018). In fact, virgin PET production capacity in the US is projected (IEA, 2018) to rise over the next decade even as bold commitments from companies and aggressive recycling targets from states proliferate. Displacement, as Zink and Geyer

(Zink et al., 2016) point out, is a market-based phenomena, and I explore what this entails for plastics recycling in Chapter 4.

2.5.4 Contextualizing cost estimates

In the US, a rapidly shifting plastics policy landscape contends with the dual challenges of availability and use of recycled plastics (US EPA, 2021). In this case study about PET bottles, we quantify supply-demand interaction effects to analyze the cost and circularity implications of expanding deposit return systems to meet 70% recycled content inclusion requirements in bottles. Under current policies, public stakeholders face collective costs of up to 2 billion USD to increase recycling rates from 24% to 82%. Robust demand can halve these costs to 920 million USD – but maintaining robust demand as recycled supply floods the domestic PET market and lowers V-PET prices will prove difficult without appropriate policy levers. Moreover, placing the cost burden on struggling and strained public waste management systems is unlikely to boost recycling efforts.

Shifting the responsibility to producers under EPR frameworks that incentivize circular demand through eco-modulation can align costs, benefits, and outcomes. This analysis reframes the total costs needing subsidy (in millions of USD) as per tonne excess costs met by EPR fees – this reframing explores how EPR, deposit return systems, recycled content mandates and virgin taxes interact in a PET bottle recycling system that stands to displace a significant fraction of domestic V-PET production. Under strong circular demand, our EPR fee estimates of 300-400 USD/tonne fall within the range of existing EPR fees levied in many European countries. If beverage producers pass this cost to consumers, the per tonne cost translates to approximately 1c per PET bottle placed on the market (in addition to the 10c deposit fee). This cost can be interpreted in two ways: it is a negligible fraction of the average beverage product and cannot be observed against usual inflationary forces. However, it also points to the sheer number of PET bottles used for beverage packaging – a number that is obscured when mass units are used throughout the analysis. Beverage container sizes vary - but an average weight of 20-30g can be assumed (Becerril-Arreola & Bucklin, 2021); this suggests that more than 100-150 billion PET bottles are sold every year in the US. Critics of deposit return systems argue that emphasis on recycling conflicts with opportunities to reduce or reuse while proponents point to reduced littering as evidence of its utility (OECD, 2022a).

Beyond visualizing PET costs across three different scales: nation-wide, per tonne, and a single bottle, we also situate our cost numbers in the larger PET value chain and inquire about environmental benefits of circularity. We conclude that creating circular demand is critical for incenting large-scale changes in the PET bottle recycling system that overcome systemic barriers. However, if higher circularity cannot be met by existing mechanical recycling processes and needs energy intensive chemical recycling technologies, we show that pursuing circularity for circularity's sake can be misguided.

Chapter 3

Polyolefins in Flexible Plastic Packaging

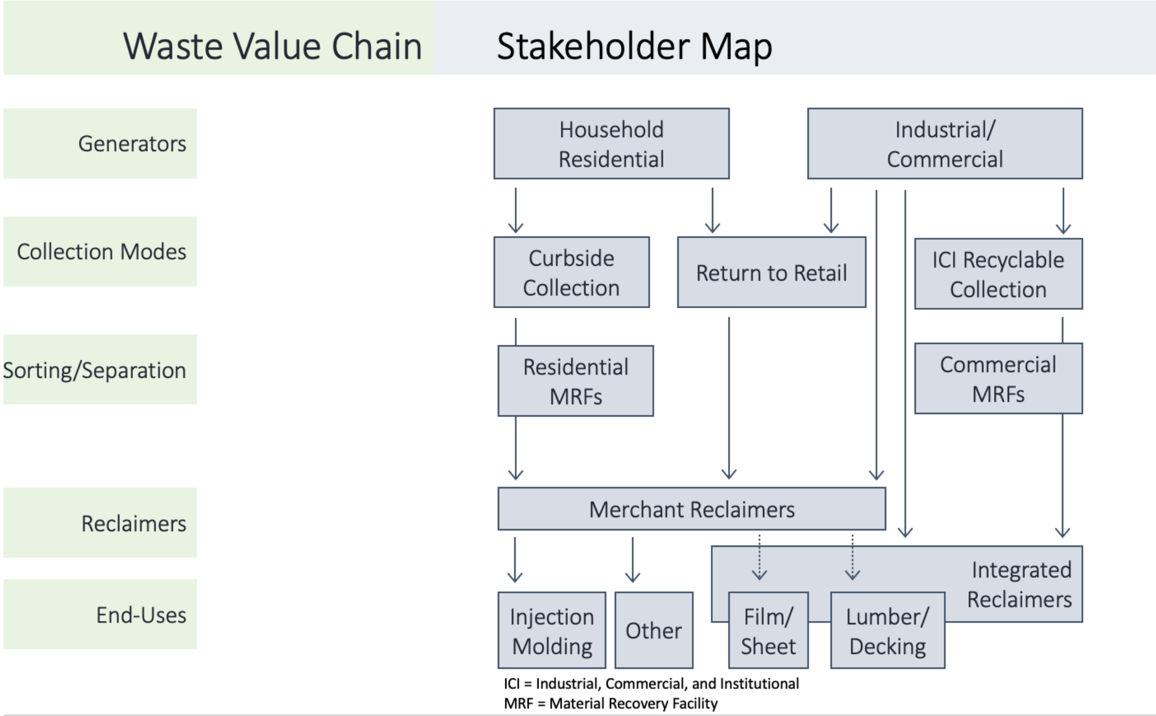
Plastic films comprising flexible plastic packaging are collected for recycling at far lower rates (<5%) than PET bottles (24%). Unlike PET bottles for beverage packaging, they are not associated with one function – they can be used for food-contact packaging (like meat packs or cereal bags), for bagging purposes (grocery/merchandise bags), for secondary packaging (such as around a crate of water bottles), or even tertiary packaging (in retail back-of-store handling). This functional diversity of flexible films is achieved by exploiting a range of material chemistries, layer designs, and processing properties. The inexpensiveness, usefulness, and ubiquity of flexible films is simultaneously a testament to advances in materials chemistry (Morris, 2022), innovations in process design (James F. Macnamara Jr, 2020), and a mundane reminder of the pervasiveness of plastics in modern life.

Polyethylene (PE) and to a lesser extent, polypropylene (PP), are the most common polymers used in flexible plastic packaging. PE and PP can both be mechanically recycled, albeit with structural changes in the polymer matrix that make it difficult to use it back in films (Horodytska et al., 2018). However, the complexity of this stream arises from the diversity of (i) polymer subtypes within PE and PP, (ii) other materials used in multi-layer film structures, and (iii) multitude of waste generation sources. Numerous technological solutions have been proposed to advance circular economy targets: compatibilizing polymers together (Kaiser et al., 2018), de-inking printed films (Horodytska et al., 2020), solution-precipitation recovery of polymers in multi-layer design (Walker et al., 2020), advanced recycling of polyolefins, including fuel/feedstock recovery (Li et al., 2022). In this chapter, I quantify barriers to recycling flexible plastic packaging by mapping film waste material flows, assessing cost of recovery, and comparing it against value of recovered products.

3.1 Background, Context

3.1.1 Flexible plastic packaging recovery in the US

Flexible plastic packaging is used for household consumer goods, as well as for industrial and commercial packaging applications. Therefore, the generated flexible film waste must be collected from residential, as well as industrial/commercial/institutional (ICI) sources. Three collection modes are available in the US serving the various waste generators, as shown in Figure 3-1. Unlike PET bottles which are accepted in curbside bins across US, The Recycling Partnership reports that only 50 local communities accept monolayer PE film in their curbside collection programs (The Recycling Partnership, 2021). In fact, misplaced flexible packaging is considered a common contaminant in curbside collected recyclables, and consumers are discouraged from throwing it alongside other recyclables ("Plastic Bag and Film Recycling", n.d.). Return to retail collection mode allows consumers to return select, clean, PE monolayer flexible plastic packaging (such as bread bags, newspaper covers, grocery and retail bags, dry cleaning bags, etc.) to participating retail centers. Like deposit return systems, return to retail programs were established by state laws mandating that retail stores that issue plastic bags also facilitate consumer return (National Council of State Legislatures, n.d.) and were then extended to other clean consumer film packaging. However, unlike deposit return systems, there are no monetary incentives (such as deposit fees) to encourage return.



ICI = Industrial, Commercial, and Institutional
MRF = Material Recovery Facility

Figure 3-1: Waste Value Chain Stakeholder Map

Commercial settings provide clean, concentrated sources for PE films from secondary and tertiary packaging (such a pallet wraps or retail back of the store packaging) and are either collected alongside cardboard packaging by commercial waste collection companies or directly handled by interested reclaimers. Reclaimers convert the recovered waste PE from flexible films into plastic pellets that are either sold (by merchant reclaimers) or used in recycled products (by integrated reclaimers). For instance, Trex, a decking company, advertises eco-friendly decking options (Trex, n.d.) made from reclaimed wood and recycled plastic bags. In 2022, Trex reported using 153 kt of plastic films in their products (Trex, 2022). Information on other reclaimers or end-uses is scarce, but More Recycling estimates that in 2018, approximately 46% of recovered PE film was consumed by the lumber/decking industry, 34% was made back into film, and 20% was used in injection molding and miscellaneous applications (More Recycling, 2020).

3.1.2 Mechanical recycling and limited end-uses

Recovered PE films are shredded into flakes, washed to remove dirt or contaminants, dried and mechanically recycled by melting and reforming into PE pellets (Horodytska et al., 2018). With clean film, the washing and drying step can be forgone saving energy and cost. Because end-users prefer recycled PE from recovered films only if it is much cheaper than virgin PE, reclaimers in turn prefer clean, transparent film that can be directly shredded and recycled. The strength of this preference can be inferred from the differentiated grades of recovered PE film bales sold on the waste market. Figure 3-2 shows the prices of Grade A, B, and C film – as defined by the Institute of Scrap Recycling Industries (ISRI). From these definitions, grades can also be mapped to collection modes – grade A corresponds to clear secondary or tertiary packaging from commercial sources, grade B is usually from return to retail sources where colored film is also accepted and grade C is MRF sorted film. Low value of MRF sorted film as well as frequent tangling of MRF equipment and contamination of other high value streams such as paper or HDPE bottles leads to low acceptance of films in curbside programs that supply recyclables to MRFs (Cimpan et al., 2015). Notably, PP in film waste is seldom recovered or recycled.

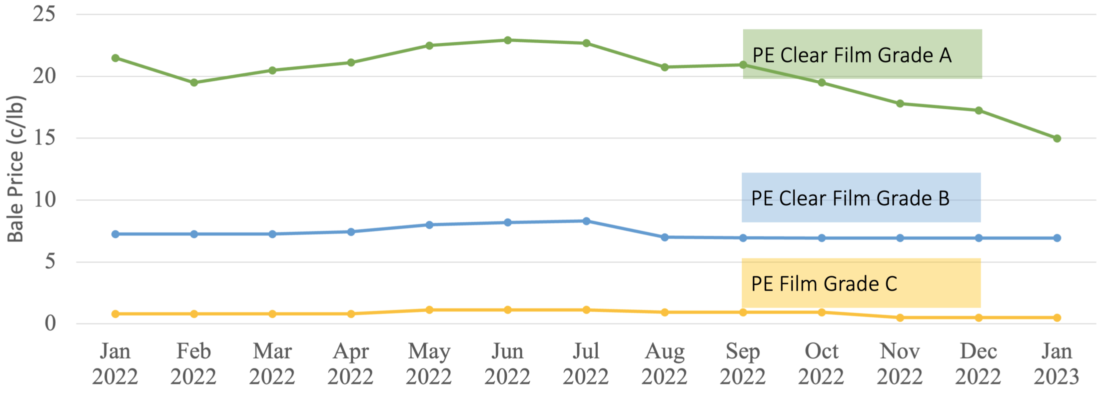


Figure 3-2: Bale price over time for flexible PE grades. Data gathered from Resources Recycling for period 2022-2023.

Recycled film PE resins find limited end-uses. For flexible packaging alone, desirable ranges for film thickness, transparency, stretchability, resistance to tear, barrier to air and moisture, printability, etc. vary widely. It is this degree of customization and variation, in addition to the thermomechanical changes during the recycling process itself, that often disqualifies use of mechanically recycled PE from films back into films. Many potential users of LDPE/LLDPE remain skeptical of recycled PE quality, and without standardized datasheets, only those end-use applications that can tolerate batch-to-batch variation in PE properties or secure a known, reliable supply of end-of-life film waste provide viable sinks (More Recycling, 2020). Moreover, end-users from decking and injection molding applications demand recycled PE as a cheaper alternative to virgin PE. But virgin PE can be inexpensively tailored to an application's desired properties, further eroding potential use cases for mechanically recycled PE from films.

3.1.3 Advanced recycling of polyolefins (PE/PP)

Unlike PET, which can be chemically recycled by depolymerization to monomers followed by repolymerization, polyolefins do not produce olefin monomers at economical yields when deconstructed (Rahimi & García, 2017). Instead, a mix of C_1-C_{50} hydrocarbons is produced by random C-C bond cleavage (also called cracking) of the long polymer chain (each macromolecule contains 10^4-10^6 C atoms). Thermodynamics and kinetics dictate the relative abundance of various saturated, unsaturated, and aromatic hydrocarbons in the product stream and this distribution can be tuned by selecting suitable temperatures, pressures, catalysts, residence times, reactor types, etc. Numerous reviews on this topic have collated studies investigating these parameters (Anuar Sharuddin et al., 2016; Faust et al., 2023; Hu et al., 2022; Lopez et al., 2017; Peng et al., 2022). Broadly, reviews agree that: higher temperatures and/or longer residence times lead to greater extent of cracking, producing higher yields of liquid and gaseous products; catalysts lower temperatures needed to achieve the suitable degree of cracking, and an increase in acidity and porosity of the catalyst can accelerate cracking; reactor design influences level of coking, catalyst deactivation, and must be optimized for heat and mass transfer alongside product selectivity and high conversion efficiencies.

Cracking[1] or C-C bond cleavage can be industrially propagated by many thermal processes:

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[1] 'Cracking' is a petrochemical terminology and it refers to the breaking down of complex or long hydro-

pyrolysis, hydrogenolysis, hydrocracking, gasification, liquefaction, etc (Scheirs & Kaminsky, 2006). Pyrolysis refers to thermal cracking (300-750 °C) in the absence of oxygen, and catalytic pyrolysis additionally makes use of catalysts to improve selectivity and yields. Sometimes, molecular hydrogen is added to the process to allow for milder reaction conditions (250-450 °C) and increase degree of saturation in products. Depending on catalyst type, cracking reactions may either be classified as hydrogenolysis, or hydrocracking based on dominant cleavage mechanisms (Liu et al., 2021). Hydrocracking requires bifunctional metal-acid catalysts (such as $Pt/WO_3/ZrO_2$ + HY) and chain scission is mediated by carbocation chemistry, while hydrogenolysis uses monofunctional metal catalysts (like Ru or Pt) that forms complexes with C-C and H-H linkages before cleaving them. Reviewing various hydro-conversion processes involving plastic wastes, A. Kots et al. (A. Kots et al., 2022) show that the faster carbocation chemistry owing to acid functionality in hydrocracking produces lighter gasoline range hydrocarbons and is typically limited to lower residence times to limit the extent of cracking. On the other hand, hydrogenolysis produces heavier diesel range hydrocarbons, waxes, and lubricants. Gasification, which occurs in the (limited) presence of air or oxygen, presents the other extreme of thermal cracking where almost all C-C bonds are cleaved, and the major product is syngas ($CO+H_2$). Compared to pyrolysis or hydro-conversion, gasification also requires higher temperatures: 700-1200 °C (Tan et al., 2023). Liquefaction refers to solvothermal processes where solvents facilitate mass and heat transfer by dissolving and distributing the plastic waste feedstock (Liu et al., 2021). In microwave pyrolysis, microwave heating is used for energy transfer (Suresh et al., 2021). Process technologies differ in reaction conditions such as temperature, pressure, and residence time, and influence reactor design parameters as well as determine energy and capital needed to produce desired hydrocarbon fractions from plastic waste at scale.

Conversion of waste plastics to chemicals and fuels by cracking them has been pursued as an alternative to mechanical recycling for more than three decades. But terminologies used for said conversion has evolved over time. In 1990, Scott et al. describe a fluidized bed process for "fast pyrolysis of plastic waste" (Scott et al., 1990). Hadjilambrinos writes: "pyrolysis processes are used to reconstitute plastics into oil products and various gases

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carbons into shorter, light hydrocarbons.

(Hadjilambrinos, 1996). These products can be used as either feedstock for resin manufacturing, or as fuel". Scheirs and Kaminsky describe feedstock recycling as "cracking of polymer chains", classifying it into pyrolysis (thermal cracking in the absence of oxygen), hydrogenation (thermolysis with hydrogen), and gasification (to produce syngas) (Scheirs & Kaminsky, 2006). Rahimi and Garcia explain that "tertiary (chemical) recycling uses a chemical process to recover the petrochemical components in plastics" in their review, classifying processes by plastic type instead (Rahimi & García, 2017). Xia and Han emphasize the value perspective and define chemical upcycling as "a chemical process that breaks down parent materials to produce value-added products" with examples such as "smaller fuel-range hydrocarbon and low-molecular-weight wax for applications such as aviation fuels and lubricants" (Xia & Han, 2023).

Despite all the different ways in which advanced recycling and fuel recovery technologies are discussed in literature, from a process perspective, advanced recycling pathways for polyolefins can all be investigated as polymer deconstruction processes followed by optional separation, upgrading and purification steps. Figure 3-3 shows sequence of four process steps that will require capital and consume energy – (1) cracking (lowering the carbon number, may use H_2 if hydrocracking), (2) distillation/hydrocarbon separation (separating the relevant hydrocarbon fractions), (3) chemical upgrading (further processes to refine or convert to other chemicals typically using other reactants), and (4) product recovery (separating products from byproducts). While step (1) is necessary to convert polymeric macromolecules to lower hydrocarbons for fuels or chemicals, studies may or may not use one or more of steps (2)-(4) depending on the product the process is optimized for (dotted lines and boxes in Figure 3-3).

In academic literature, advanced recycling processes are often investigated to assess economic feasibility and environmental sustainability of large scale implemetation. Researchers use techno-economic assessments to demonstrate that proposed advanced recycling processes can create high value products (Tan et al., 2023) from currently non-recycled plastics (often including flexible plastic packaging). Such assessments either estimate net present value for an advanced recycling facility or calculate minimum selling price of a major prod-

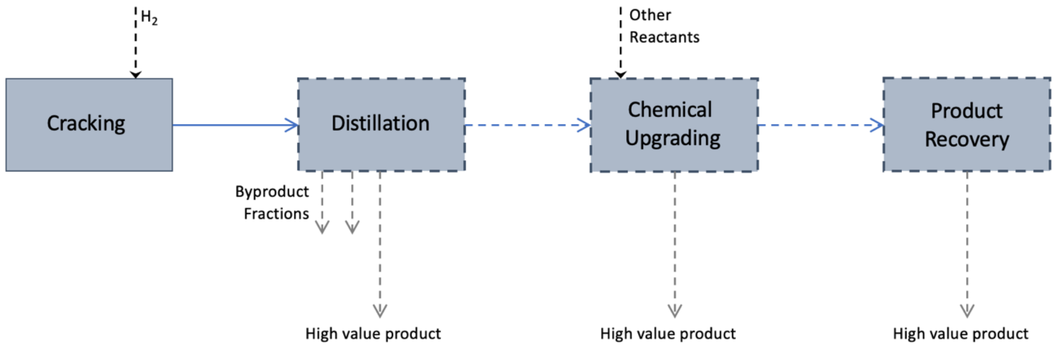


Figure 3-3: Advanced recycling steps.

uct and compare it against conventional products. Variations among studies are numerous and assumptions about plastic waste and product composition, product separation needs, capital expenditure, feedstock and product prices influence metrics of feasibility (Keller et al., 2022). Beyond economics, life cycle assessment methods are used to understand the environmental impacts of advanced recycling methods (Gear et al., 2018). Most studies adopt a waste perspective, comparing advanced recycling against incineration or landfilling and assuming displacement credits for products and byproducts. The former produces higher GHG emissions while the latter produces more solid waste than advanced recycling. The underlying rationale for the waste perspective is that advanced recycling diverts non-recyclable plastic waste from the disposal fate. However, many studies of this perspective assume full displacement, which is a market-based phenomena, and assume the nature of the displaced product without considering full compositional requirements for sellable products/byproducts. One recent study (Yadav et al., 2023) instead adopts a product perspective, comparing the GHG emissions of chemical feedstocks derived from plastic waste against their petrochemical counterparts.

From a product perspective, plastic waste provides an alternative feedstock to the petroleum refining and petrochemical industry. Researchers select process types and parameters to optimize for pyrolysis oil (usually C_5-C_{20} hydrocarbons), fuel fractions like diesel or gasoline, naphtha as feedstock for olefins and chemicals, or even specific hydrocarbon fractions for

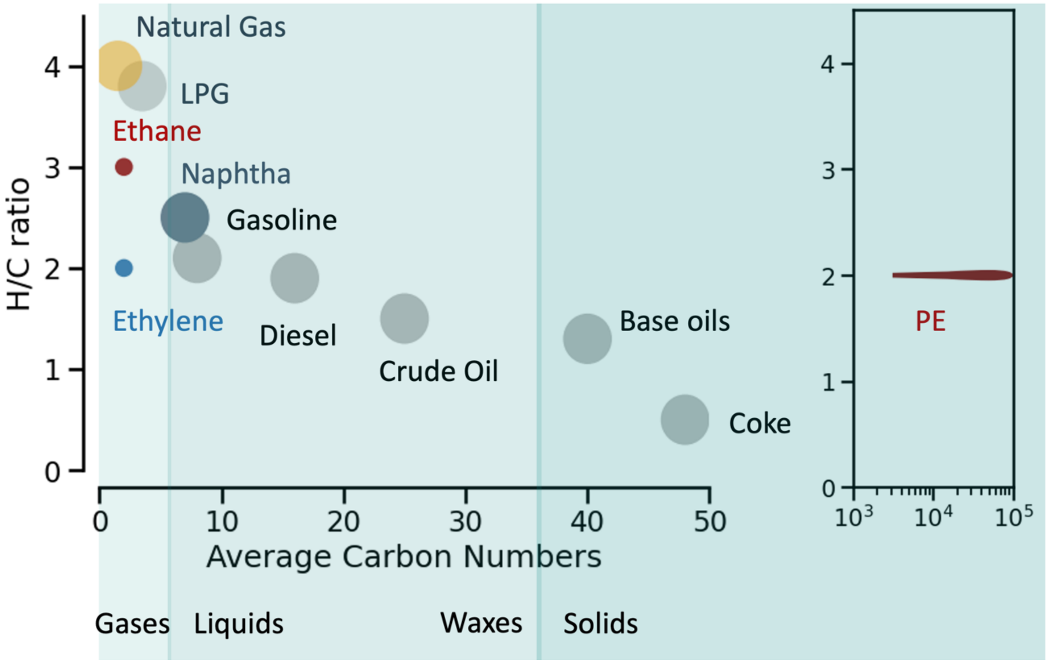


Figure 3-4: C1-50 hydrocarbons in the economy.

waxes and lubricants. To emphasize the inspiration from and connection to petrochemistry, I adapt Figure 3.1.3 from Speight (James G. Speight, 2021) that places various fuels and chemical feedstocks according to their hydrogen and carbon composition.

3.2 Research Questions

In this chapter, I assess the techno-economic barriers to recovering and diverting hard-to-recycle flexible plastic packaging from disposal. A sequence of three research questions steers this assessment:

1. What are the types and volumes of flexible packaging flows in the US?
2. What is the cost of increasing recovery of flexible plastic packaging waste?
3. What is the techno-economic potential of proposed advanced recycling processes in increasing recovery at scale?

3.3 Method Overview

3.3.1 Mapping material flows for flexible plastic packaging

Material flow analysis (MFA) is a methodological framework in environmental science and industrial ecology used to systematically investigate and quantify the flows and stocks of materials within a defined system (Brunner & Rechberger, 2016). This approach can provide a comprehensive understanding of material consumption, use, and disposal. In this study, the system boundary extends from waste generation to waste disposal, but an additional layer of product type is added to display the complexity of flexible plastic packaging waste. This study was conducted in three stages: data collection, visualization, and reconciliation with multiple iterations between the latter two.

Systematic collection, classification, and analysis of flows of flexible plastic waste in the US required consulting numerous gray literature sources. We found that data gaps are common and most of the existing data lacks a strong empirical basis. This is common with waste flows that are not closely monitored. Beyond suspect data source quality, we also found that data-points are not temporally (year of estimation) or spatially (geographic scope is sub-national) consistent. Moreover, without a clear classification scheme within flexible films, flows are aggregated based on custom defined classes relevant to the goal of each study, are also inconsistent across studies. Details on defined classes/groupings, data sources, and assumptions can be found in Appendix C.1.

Material flows are commonly visualized as Sankey diagrams – with flows 'flowing' between nodes that represent processes or stocks. Layers in a material flow diagram correspond to a series of nodes at the same life cycle stage and we define 3 such stages – flexible film use product type, waste generation, and waste recovery. The nodes within a stage represent some method of grouping of flows. In this material flow analysis, I group flexible film use by product types (bags, shrink film, etc.), generation by waste generator type (residential, C&I retail, etc.), waste recovery by destination (residential MRF, direct collection, etc.). Full list of nodes can be seen in Table 3.1 below.

Table 3.1: Material Flow Analysis Nodes

Generator	**Product Grouping**	**Collection/Disposal**
C&I Services	Retail Bags	Return to Retail
C&I Retail	Heavy Duty Shipping Bags	Residential MRF
C&I Hospitality	Box Liners	Commercial MRF
C&I Manufacturing	Storage Bags	Other Collection
Agriculture	Roll Wrap	Disposal
Residential	Shrink Bundling	
	Stretch Films	
	Pillow Pouches	
	Lay Flat Pouches	
	Stand-up Pouches	
	Woven bags	
	Wraps & Others	

Lastly, due to the degree of data scarcity, data reconciliation is performed manually. Datatable for final flow values can be found in Appendix C.2.

3.3.2 Estimating supply curves for recycled PE

We build a supply curve for recycled PE from flexible plastic packaging waste by estimating the recovery and recycling costs and volumes for various collection modes and waste generators respectively. The stakeholder map in Figure 3-1 and material flow mapping in section 3.4.1 inform our assumptions for the availability, cost, and quality of flexible plastic waste collection services currently available in the US – and are invoked to calibrate the baseline scenario. We also additionally model three other scenarios of increased flexible film packaging collection across the three collection modes – commercial collection, return to retail, and curbside collection.

Modeling recovery costs and quantity

Residential waste generators can place flexible plastic packaging waste in curbside collection channels or return to retail channels. In the baseline scenario, curbside mode is minimized to reflect widespread practices encouraged to not contaminate recyclables. Cost of curbside collection varies between 150-250 USD/tonne as modeled in Chapter 2 (and described in

Appendix A) - we use the average estimate of 200 USD/tonne. Sorting costs are added to curbside collection (commercial collection and return to retail are assumed to collect sorted plastic films). Lin et al. estimate the cost of sorting flexible plastic packaging from single stream recycling to be 82 USD/tonne, but this does not account for contamination of other waste streams on account of introducing flexible plastic packaging (Lin et al., 2023). We round this up to 100 USD/tonne to include costs of additional quality control for other waste streams. This produces Grade C film, and does not include additional cost of sorting within flexible plastic packaging to improve quality to Grade A or B. Quantity recovered in the baseline is estimated by assuming a set-out rate (0.02), curbside access fraction (0.7), MRF recovery efficiency (0.5), and plastic film fraction (0.8) in the bale based on current reports (RRS, 2020; RSE, 2014; Tanimoto, 2020; The Recycling Partnership, 2021). For the increased curbside collection scenario, set-out rate is artificially increased to 40%. Note that our cost estimates do not account for behavior change strategies such as information campaigns, policy mandates, etc.

Cost of return-to-retail system is difficult to estimate. An American Chemistry Council (ACC) report of a 5-month pilot program in Seattle (ACC, 2022) has reference costs in the range of 800-1000 USD/tonne of flexible plastic packaging (including transportation) - we use a lower cost of 600 USD/tonne to account for scaling up of the pilot program. The Recycling Partnership (The Recycling Partnership, 2021) reports that on average, 7% and 3% of retail and non-retail bags (box liners, storage bags, and wraps) are returned to retail centers – and we use this for the baseline scenario. For the increased collection, these return rates are artificially increased to 40% across the categories. Participation behavior is not modeled, and therefore, excludes the cost of changing behavior/incentivization needed to increase return rates.

Commercial waste generators, depending on their size and location, can be using curbside collection (e.g., small downtown establishments in cities), commercial collection (larger establishments in sparser regions), or direct contracts with integrated reclaimers (large retail stores or hospitals with clean film separately collected). To model all possibilities, commercial waste generators are binned by their average sizes before they are assigned a collection mode, a probability of access to the collection mode, and participation rates. Average sizes of commercial establishments in the 4 C&I categories are estimated by scaling data from California counties reported by CalRecycle to all counties in all states using NAICS sector-wise employment data (following methodology in Meyer et al., 2020). Figure 3-5 (A) visualizes the various parameters used to model commercial collection in the baseline scenario. In the increased commercial collection scenario, probability of access to service is increased as shown in Figure 3-5 (B). Average size of household recyclables (all materials) in tonnes per year is also visualized to provide reference for tonnes generated.

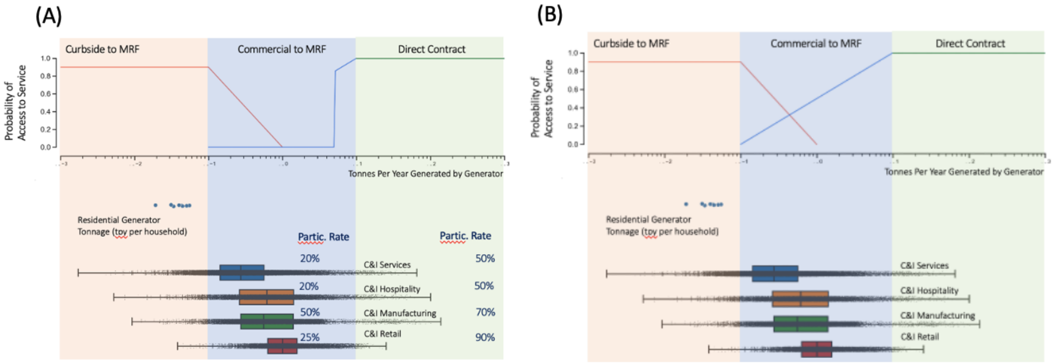


Figure 3-5: Collection model parameters visualized.

Curbside to MRF channel quantities is combined with residential generators, and same costs are assumed. To estimate the cost of separate commercial collection of flexible films $(Cost_i^{com})$, a simple functional relationship is used relating $(Cost_i^{com})$ to Q_i using total fixed (F) and unit variable (v) costs. Online cost quotes by the firm Waste Management suggest that $F = 1600$ USD and $v = 60$ USD/tonne including storage and transportation for weekly collection.

$$(Cost_i^{com}) = F/Q_i + v \tag{3.1}$$

Direct collection costs are not published, and our estimates are based on an interview with a supply representative from Trex, an integrated reclaimer of plastic film waste. Quoted costs were 100-200 USD/tonne. We also assume a constant recycling cost of 400 USD/tonne for collected, sorted, flexible film following economic estimates (Larrain et al., 2021; Lin et al., 2023).

3.3.3 Generalized techno-economic assessment (TEA)

We review 29 studies of advanced recycling and pyrolysis processes in literature that report feasibility or techno-economic analysis. We tabulate assumptions about feedstock type, feedstock costs, other operating costs, process steps, yields, product prices, energy consumption, capital expenditures, and more. A detailed breakdown of these studies can be found in datatables in Appendix D.

Generalized techno-economic assessment

In this study, we present a simplified techno-economic assessment generalized across process technologies using five parameters whose ranges and interdependencies are inferred from literature review (and discussed in results section 3.4.3).

Scalability of advanced recycling processes producing hydrocarbon feedstocks or chemicals depends on their relative economic attractiveness against conventional means of production. Therefore, the sole economic metric of interest investigated in this study is the minimum selling price (MSP) of the main product, which is then compared against historical prices. As a comparative measure, price includes profit margins, implicit subsidies, as well as variability due to the larger energy and chemicals markets and is also more accessible than marginal costs of many petrochemical producers. Minimum selling price of a product is calculated as –

$$MSP = \frac{TCI * \frac{r*(1+r)^n-1}{(1+r)^n-1} + OpEx - Coproduct\ Revenue}{Annual\ Production\ of\ Main\ Product} \tag{3.2}$$

In the numerator, total capital investment (TCI) is annualized by interest rate r over a period N and added to operating costs ($OpEx$) from which revenue from coproduct sale is subtracted. Constant values for r (0.1) and N (30) are assumed.

Total capital investment (TCI) depends on annual capacity of the facility as well as process design and steps. This dependency is modeled with two parameters.

- Annual capacity of the modeled facility (in ktonnes) is denoted by C.
- Total capital investment per tonne of annual capacity at 100 ktonnes ($TCI_{per\ t}^{@100kt}$) is used to isolate and capture the capital needs for the process at scale.

Then,

$$TCI = TCI_{per\ T}^{@100kt} * 10^5 * (\frac{C}{100})^m \tag{3.3}$$

Where m is used to capture the economies of scale relationship between capital costs and annual capacity. Empirical evidence gathered from existing petrochemical facilities indicates that m is typically 0.6-0.7 (Seider et al., 2017).

Operating costs are estimated as a sum of total feedstock costs (FC) denotes unit feedstock cost per tonne as the main variable cost driver) and other operating costs (from a constant unit estimate of 400 USD/tonne).

$$OpEx = (FC + 400) * C \tag{3.4}$$

Because advanced recycling processes often create multiple products, one product is considered main product, while others are relegated to co-products. Product yield is assumed to be $\zeta > 0.4$; a single co-product yield and price is assumed to represent the aggregated yields $(1 - \zeta)$ and weighted price (π). The simplifies multi-product considerations without loss of generalization.

$$Coproduct\ Revenue = (1 - \zeta) * C * \pi \tag{3.5}$$

And,

$$Annual\ Production\ of\ Main\ Product = \zeta * C \tag{3.6}$$

Lastly, feedstock costs are linked to advanced recycling capacity that can be feasibly realized. Feedstock costs for flexible film packaging are approximated based on recovery costs (not including mechanical recycling) estimated in 3.4.2. A functional relationship between total flexible plastic packaging recovery amounts (A) and recovery cost is characterized:

$$FC = f(A) \tag{3.7}$$

To relate total amount recovered (A) and average capacity (C) of an advanced recycling facility, number of facilities across US is arbitrarily set to $p = 10$, with the assumption that any fewer would necessitate long transportation distances and high feedstock costs. Moreover, advanced recycling is pursued once mechanical recycling is no longer feasible, and the amount sent to be mechanically recycled is set to 2019 baseline (MR). Therefore, $C = (A - MR)/p$, and MR = 300 kt.

Scenario Descriptions

For the generalized TEA, we assess four scenarios by varying system economic parameters $TCI_{per\ t}^{@100kt}$ and feedstock cost (FC) which are linked by annual capacity. Scenarios with lower capital costs are assumed to correspond to fuel or pyrolysis oil production, and higher capital costs with refined chemicals production. Based on assumptions on products, technical parameters are also restricted based on literature review of advanced recycling processes and their TEAs. The scenarios and their parameters are described in Table 3.2:

Table 3.2: Recycling process TEA scenarios and associated parameters

S	$TCI^{@100kt}_{per\ T}$ (USD/annual tonne)	Feedstock Cost (USD/tonne)	Product Yield Ranges	Average Co-product Price
LL	500	500	0.4-0.9	200-400
LH	500	1000	0.4-0.9	200-400
HL	2500	500	0.4-0.7	200-700
HH	2500	1000	0.4-0.7	200-700

3.4 Results

3.4.1 Mapping flexible film waste flows in the US

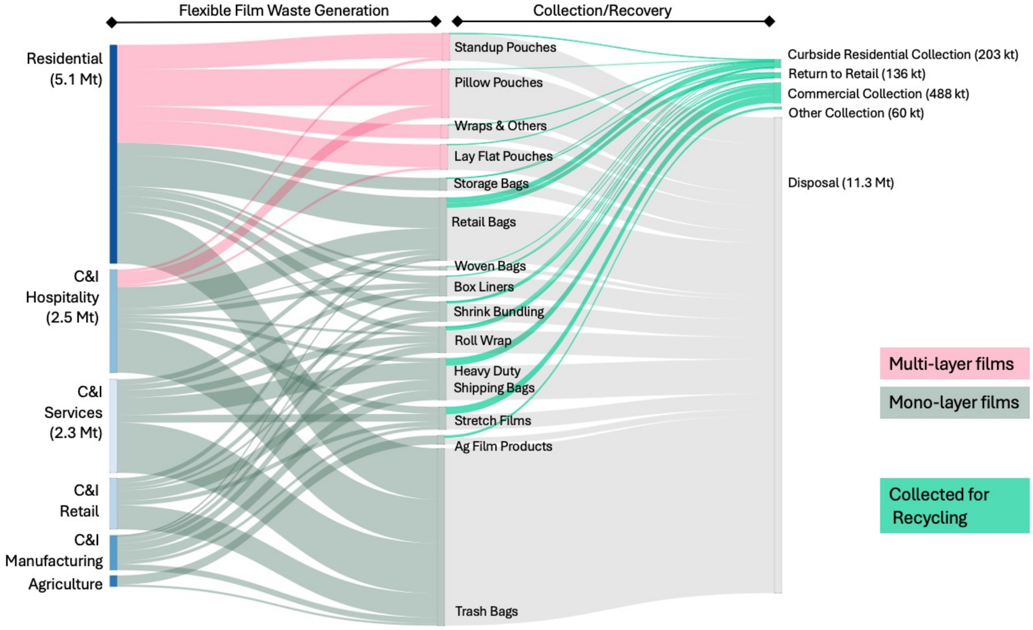


Figure 3-6: Material flow mapping of flexible packaging waste visualized by waste generator and product grouping for US, 2018. Flow values are tabulated in appendix C.2

Despite widespread recognition that flexible film waste is a growing concern, estimates of waste generation and recovery are sparse, incomplete, and inconsistent in the US. We present a detailed generator/product type cross-section of flexible film waste generation in 2019 to identify opportunities for reduction, recovery, and recycling. Total waste generation estimates vary between 8-12 Mt, of which only 41-55% is from residential sources. The remaining flexible packaging waste is generated by commercial establishments and industrial centers (C&I), which have been grouped into four sectors: hospitality (2.5 Mt), services (2.3 Mt), retail (1.2 Mt) and manufacturing (0.8 Mt). Considering product types, we see that residential sources have greater fraction of multilayer packaging, as expected from primary food and grocery packaging which often needs tailored multilayer profiles for food preservation, moisture/oxygen barriers, and labeling. C&I services, retail, and manufacturing use flexible PE films for secondary and tertiary packaging, which is typically monolayer, and provides clean end-of-life waste for recovery.

Recovery rates for plastic film waste are low (450 kt/8500 kt = 5.3%) and domestic recycling rates are even lower (3.8%) when export (140 kt) is excluded. Most of the waste is collected from commercial collection (270 kt) – which includes both commercial MRFs and direct contracts between waste generators and reclaimers. Return to retail (70 kt) and curbside collection (are channels available to residential sources as well as smaller commercial establishments in C&I hospitality (such as restaurants) and services (office buildings) – but with data gaps, only total collection volumes are known, and the flows from each of the sources are highly uncertain. Because it is impossible to know data fidelity of estimates based on numerous assumptions made by various authors and practitioners – we flag flows that lack information or have conflicting information. This helps us map the flexible plastic packaging waste system for further analysis while noting important data gaps. Next, we use this mapping to estimate the cost of recovering PE film waste and explore scenarios of increased recovery and recycling.

3.4.2 Improving PE film collection and recovery.

Figure 3-7 visualizes the cost of recovering waste PE films in the current recycling system, and 3 other hypothetical scenarios, colored by collection modes. The three hypothetical scenarios progressively increase participation along the three collection modes, i.e. greater commercial recovery, higher return to retail access, and higher residential recovery.

Literature estimates suggest that the cost of recycling PE from recovered film – including washing, drying, and reprocessing is approximately 400 USD/tonne (Larrain et al., 2021).

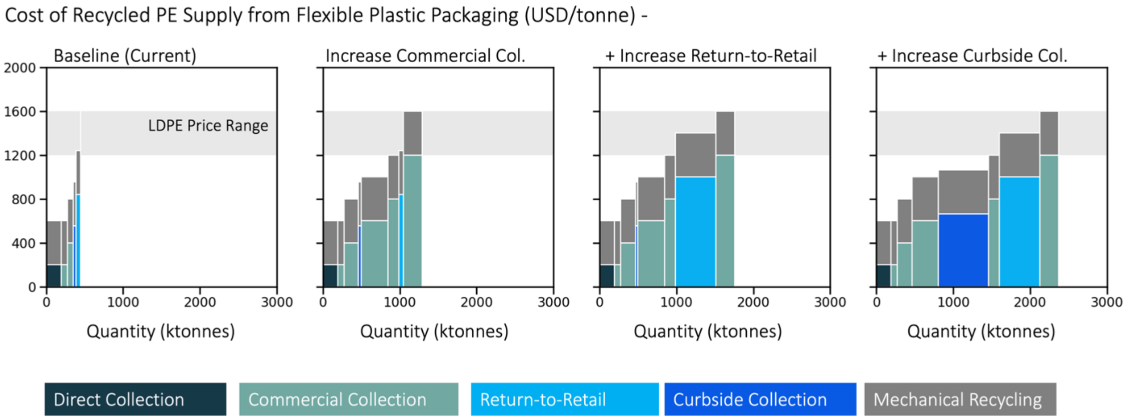


Figure 3-7: Cost of recycled PE supply from flexible plastic packaging

The gray bar in Figure 3-7 stacks this cost on the recovery (collection + sorting) costs. Demand for mechanically recycled PE must account for quality loss compared to virgin LDPE, and consequently, we expect recycled PE prices to be 20% lesser than virgin LDPE to be cost-competitive for end-users. This 20% quality margin is inferred from Grade A PE film bale with clean film recycling costs. Comparing it against the average price of virgin LDPE in the US (between 2016-2019), we find that increasing recovery for mechanical recycling of waste films for use in PE end-uses can be achieved by increasing commercial collection (to approx. 850 kt) but return-to-retail and curbside collection levers are, on average, not price-competitive unless virgin LDPE prices are high. While an increase in recovery from 450 to 850 kt can approximately double the recycling rate from 3.8% to 7%, even with the necessary reverse logistics channels in place, flexible film packaging recycling rates will continue to be low.

It should be noted that estimates on the cost curve in Figure 3-7 do not include behavioral levers or policy changes needed to improve recovery, instead focusing on the cost of reverse logistics, inferred, and collated from diverse sources. Actual costs are likely to be higher, especially for the hypothetical scenarios because the setting up of these scenarios depend on incentives to create the additional supply of recovered films. Incentives can be regulatory (landfill bans), fiscal (subsidizing collection or recycling), or market-based (reclaimers can pay more for the waste films if the material is valuable). However, cost of monitoring regulation compliance can be prohibitively expensive, and without demonstrated demand for recycled PE from flexible films, recovery efforts can be misplaced. Expensive, lower quality, and inconsistent supply of recycled PE is not attractive to end-use manufacturers that contend with low profit margins, and only modest increases in recycling rates can be achieved by improving collection activities. In other words, in the current recycling system, flexible film waste collection rates are not higher because collecting any more film waste is not *worth* it.

3.4.3 Assessing the techno-economic potential of advanced recycling.

When mechanical recycling cannot drive demand for end-of-life plastic waste and divert it from disposal fates, advanced recycling is championed (Li et al., 2022). To assess the techno-economic potential of advanced recycling processes for polyolefins in flexible plastic packaging, we first collate data from techno-economic feasibility or assessment reports in literature. The full TEA database of advanced recycling processes that treat PE, PP contains 29 reports, includes studies from 1996-2023, and is tabulated in Appendix D. We review key techno-economic parameters, discuss similarities and differences in assumptions and how they influence results, and identify interdependencies between parameters. Building off of this review, I simulate parameter interdependencies within a simplified, generalized techno-economic model to assess whether and to what extent advanced recycling can incentivize greater recovery of flexible plastic packaging waste.

Key techno-economic considerations and parameters

Objectives of technoeconomic assessments are two-fold – (1) demonstrating technical feasibility of producing a desired product at substantial yields, and (2) assessing the cost-competitiveness of the product against other production pathways (Uekert et al., 2023). In our advanced recycling TEA database, only few, recent studies producing chemicals do both (Hernández et al., 2023; Li et al., 2022; Xu et al., 2023). More often, and especially for many pyrolysis TEAs producing fuels, studies assume yields from previous literature and approximate process analysis for a hypothetical facility (Fivga & Dimitriou, 2018). Metrics for cost-competitiveness can vary. On a product basis, estimating the minimum selling price for the main product while assuming co-product benefits can be a direct indicator but, on a facility-basis, profitability is assessed from net present value calculations. Such calculations use assumptions of product yields and prices which can be varied across scenarios. In our own generalized techno-economic model, we adopt a product perspective, and seek to estimate minimum selling price but for this initial review, we include both product-based and facility-based studies to include a larger set of datapoints. Figures 3-8 and 3-9 visualize the various considerations discussed below.

Products, Yields, and Prices

Cracking of polyolefins produces a range of hydrocarbons, and further specificity in this range can be achieved by varying temperatures, pressures, residence times, and catalytic activity. As a result, advanced recycling processes can have a wide variety of products. Moreover, different studies may use different names for their products, and several product categories can overlap. In Figure 3-8, I broadly categorize products as – pyrolysis gases, pyrolysis oil, fuels, naphtha, BTX and aromatics, olefins (including ethylene and propylene), and other chemicals. Note that these categories are not mutually exclusive or exhaustive but are used for simplification and aggregation.

Many researchers report production of pyrolysis gases and oils along with solid residues such as waxes, char, or coke (Faisal et al., 2023); some characterize the composition of these phases (Liu et al., 2021). Several TEAs assume that, depending on the carbon number distribution in the pyrolysis oil, it can be sold as synthetic crude oil alternative for further refining or as a heavy distillate fuel oil without refining. Typically, these products fetch the lowest price. When the carbon number distribution is further limited to specific fuel range hydrocarbons, typically using catalysts, TEAs assume that gasoline or diesel may be displaced and use higher prices (than crude oil) for said fuels. This is an optimistic assumption; fuels such as gasoline and diesel have important fuel properties (such as octane or cetane number) that depend on chemical composition beyond carbon number distribution (paraffins, olefins, iso-alkanes, cycloalkanes, etc), and TEA studies do not ascertain whether fuel-like products comply with compositional and property requirements for sale as a fuel. While numerous studies have shown that fractions from plastic pyrolysis oil can be successfully blended with diesel for automobile combustion (albeit with higher NO_x, CO_2, and CO emissions) (Jahirul et al., 2022), we find that TEA studies do not accurately model distillation requirements to retrieve or purify such fractions (Hernández et al., 2023; Larrain et al., 2021). Instead, studies routinely conflate fuel-range hydrocarbons with the specific, marketable fuels (Bora et al., 2020; Li et al., 2022), overestimate fuel yield by equating it to liquid yield (Fivga & Dimitriou, 2018; Sahu et al., 2014) and underestimate capital costs required for the facility's operation by not considering fuel separation and refining stages (Kulas et al., 2023; Riedewald et al., 2021). To be clear, fuel recovery is not considered recycling in this dissertation as it violates material cycling. However, since the resulting hydrocarbon mixes can also serve as alternative feedstocks for petrochemicals production, we include TEAs of such cracking processes to include relevant datapoints.

Naphtha, a less refined mixture of hydrocarbons in the gasoline-range (C_5-C_{12}), is a common petrochemical feedstock, and some TEAs assess naphtha as a major product (Kulas et al., 2023; Larrain et al., 2021; Yadav et al., 2023). But naphtha is a low-value product, sometimes priced lower than the crude oil from which it is derived, and to increase value of the final product, researchers optimize process parameters to produce higher value chemical feedstocks such as olefins (such as ethylene, propylene) and BTX rich aromatics (Yadav et al., 2023). However, targeting specific hydrocarbon fractions without further reforming leads to low yields (0.12-0.19) (Gracida-Alvarez et al., 2019; Yadav et al., 2023), and further reforming adds capital costs, presenting tradeoffs that require detailed process modelling (Yadav et al., 2023). Same processes can be adapted to synthesize functional hydrocarbons to produce higher value chemicals further down the petrochemical value chain. Li et al. (Li et al., 2022) employ hydroformylation and hydrogenation of pyrolysis oil fractions to produce monoalcohols and diols. Similarly, oxidative degradation can be used to convert PE and PP into diacids (Bäckström et al., 2017; Pifer & Sen, 1998). Typically, functionalization adds additional purification and upgradation steps that can be capital intensive. However, Wang et al. show that catalytic oxydegradation and hydrogenolysis can directly produce aliphatic dicarboxylic acids as precursors to polyesters, paints, and coatings (Wang et al., 2023). Studies also convert polyolefin wastes to waxes and lubricants under milder cracking conditions (A. Kots et al., 2022; Cappello et al., 2022; Hackler et al., 2021; Liu et al., 2021). Waxes contain long chain hydrocarbons that can be further functionalized to produce higher value surfactants (Kanbur et al., 2021; Xu et al., 2023). Few studies producing chemicals have conducted thorough TEAs and most processes are in early-stage research. In our generalized assessment, we use these examples to ascertain product yields and prices for high value chemicals. We identify main products and co-products based on yield, and estimate minimum selling price (MSP) of the main product (as a function of yields and co-product prices) that can be compared against product prices.

Prices of chemicals fluctuate greatly. The petrochemical industry derives hydrocarbon feedstocks from the fossil-based energy sector. As a result, petrochemical prices depend on oil and gas prices that are influenced by energy supply/demand imbalances dictated by geopolitical interests and macroeconomic conditions. Moreover, lower the product price, greater the relative impact of this fluctuation. For instance, ethylene price, depending on time and location, may vary between 500 to 1400 USD/tonne (Yadav et al., 2023). Price uncertainty presents significant risks to investment in advanced recycling facilities, and to capture this uncertainty we use price ranges.

Operating Costs and Feedstock Costs

Operating costs include variable costs of raw materials (feedstock as well as other consumables), catalysts, process fuel, and utilities, and fixed operating costs such as labor, taxes, insurance, and maintenance. Of these, assumptions about fixed operating costs are usually a linear function of other costs (Cappello et al., 2022; Westerhout et al., 1998), following principles in cost estimation and can be considered largely independent of the process itself. These assumptions are not further investigated here as their rationale can be traced to existing petrochemical production processes and related financing decisions. We find that energy costs, including process fuel as well as utilities for cooling and heating, are usually a very small fraction of total operating costs when reported separately (Hernández et al., 2023; Larrain et al., 2021; Xu et al., 2023). Even though energy consumption is significant (8-10 MJ/kg plastic waste), this is likely due to the use of pyrolysis off-gases consumed as fuel for cracking energy needs (Gracida-Alvarez et al., 2019; Jiang et al., 2020). However,

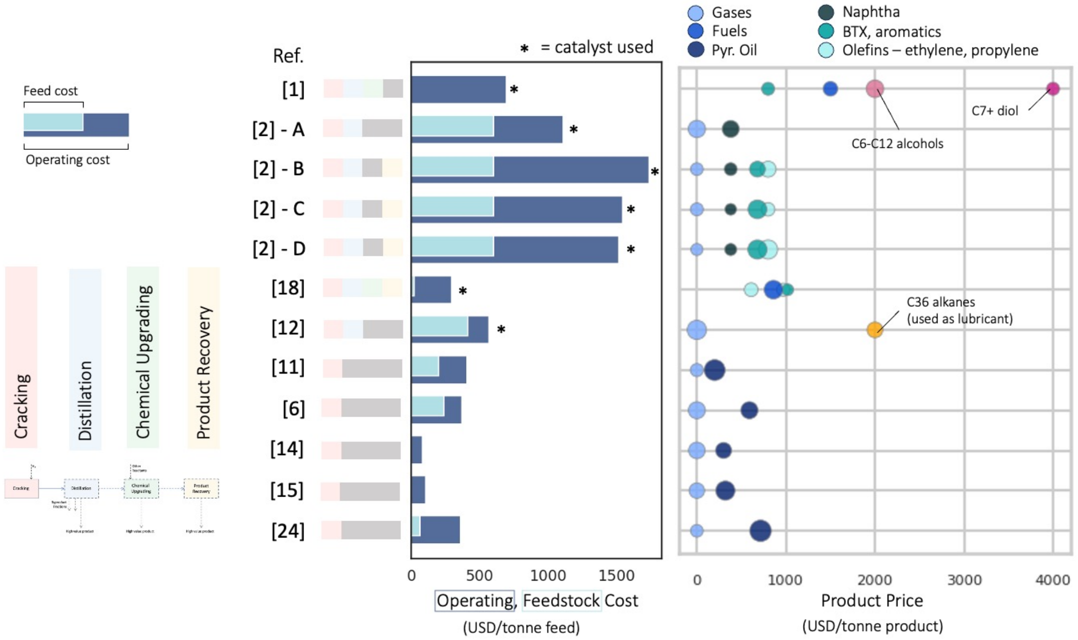


Figure 3-8: Visualizing assumptions about operating costs, feedstock costs, product yields, and product prices found in select studies from review database.

some studies estimate much higher energy costs: Gracida-Alvarez et al. (Gracida-Alvarez et al., 2019) extensively model extractive distillation requirements and find that 50% of operating costs are energy-related, suggesting that production of refined chemicals likely has much higher energy costs due to separation. Costs for catalyst purchase vary as catalyst type and loading differ between processes but because catalysts are recovered and reused over many cycles, this cost is usually small (Cappello et al., 2022). Consumables such as H_2 or syngas in hydrotreatment (Gracida-Alvarez et al., 2019; Li et al., 2022), also add to costs depending on process stoichiometry, and are appropriately accounted for (Almohamadi et al., 2021; Cappello et al., 2022; Larrain et al., 2021).

On the other hand, assumptions about the costs for procuring plastic waste feedstock, a necessary raw material, is inconsistent across studies (see Figure 3-8). These assumptions can be categorized into three classes; some studies assume that (a) plastic waste feed has a net negative cost as tipping fees is paid to the advanced recycling facility, (b) plastic waste feed is available for free, and (c) plastic waste feed has a net positive cost (100-600 USD/tonne). The rationale for a negative cost lies in considering advanced recycling to be an effective strategy for diverting currently non-recyclable plastic wastes from waste disposal in landfills or incineration which require tipping fees in the order of 30-60 USD/tonne (EREF, 2022). The critical flaw in this reasoning is that landfills and incinerators do not require separating plastics from other waste fractions. Some researchers then argue that, if the cost of separation is of the same order as disposal fees, plastic waste can be separated from other wastes and obtained for free. This argument is supported by the fact that material recovery facilities have concentrated bales of misplaced and/or non-recyclable plastic wastes that they can sell at near-zero cost (instead of paying disposal fees for it). However, this source is limited in quantity, and it cannot be scaled. Beyond MRFs, for most individual waste generators, the cost of disposing lightweight plastic waste seldom exceeds the effort needed to separate it from mixed plastic waste and transport it to merchant reclaimers. Therefore, scaling up collection and sorting efforts requires recovery mechanisms and infrastructure that can only be built out if plastic waste has non-zero value. In other words, plastic waste must be priced greater than zero, and to meet this price, advanced recycling facilities (or stakeholders subsidizing such facilities) must pay a non-zero advanced recycling feedstock cost. The exact cost of obtaining plastic waste feedstock depends on quantity and quality of feed desired as well as specific policy contexts – for the US, in the absence of any policies covering flexible packaging, our analysis in 3.4.2 shows how a supply cost curve can estimate such costs. Because such assumptions significantly impact total operating costs (i.e., higher feedstock costs raise operating expenditures), and determine the feasibility of scaling advanced recycling in the US, I include feedstock cost as one of the key parameters.

Capital Costs and Annual Capacity

Beyond operating costs, capital costs also influence profitability and inform decisions about investment in scaling advanced recycling technologies. Research in chemical engineering and process design suggests that capital costs scale sub-linearly with capacity (usually measured in tonnes per year or TPY) – and an exponential factor of 0.6-0.7 is assumed (Towler & Sinnott, 2021). In other words, economies of scale are expected. Figure 3-9 plots capital cost estimates per unit capacity against annual capacity. We find that assumptions on

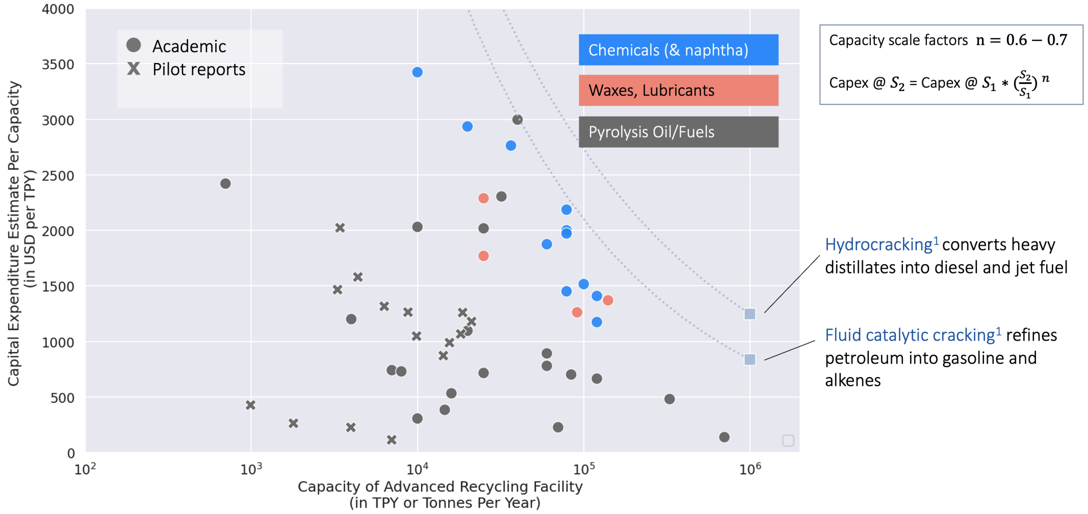


Figure 3-9: Capital cost per annual tonne of various advanced recycling processes investigated in literature visualized by scale of operation assessed. Data for large-scale cracking processes is adapted from Towler and Sinnott 3rd edition (Towler & Sinnott, 2021)

capital costs vary widely across literature and these variations can be traced to (a) process requirements and reactor design (b) number of steps modeled – i.e., product separation or additional refining, and (c) annual capacity sizes, and (d) system scope definitions, including siting, etc. We find that there are contradictions within and across variations – owing to inconsistent and incomplete assumptions – and without real data on established large scale processes to benchmark against, it is unclear which assumptions are correct. However, comparing against similar petroleum and petrochemical processes (blue squares) and using the appropriate scaling exponent from process design textbooks, we find that almost all capital cost estimates are low, and some, specifically the pilot plant estimates are off by almost two orders of magnitude.

From this brief review, we characterize capital costs in the generalized meta-assessment by two parameters: capital cost per TPY @ 100 ktonnes and capacity of the average facility (in TPY). Capacity of the average facility is linked to feedstock costs, as higher feedstock costs incentivize higher recovery of plastic waste, ensuring consistent volumes in sourcing. Capital costs for the average facility is estimated by scaling the cost level set at 100 ktonnes per year with the average capacity supported by feedstock costs.

Product-based generalized techno-economic assessment

The four scenarios we assess in this generalized techno-economic assessment are devised based on capital costs and feedstock costs (see Table 3.2. Higher capital costs are ascribed to technologies that produce refined chemicals, and lower capital costs for production of pyrolysis or fuel oils; higher feedstock costs allow greater collection and larger capacities, while lower feedstock costs can only sustain lower collection costs and lower capacities. In Figure 3-10, we visualize minimum selling price (MSP) for main product as a function of product yields and co-product prices. Product yields above 0.7 for refined chemicals are assumed to be outside of expected range and greyed out; co-product price beyond 400 USD/t for pyrolysis oil production is assumed to be out of range and similarly greyed out.

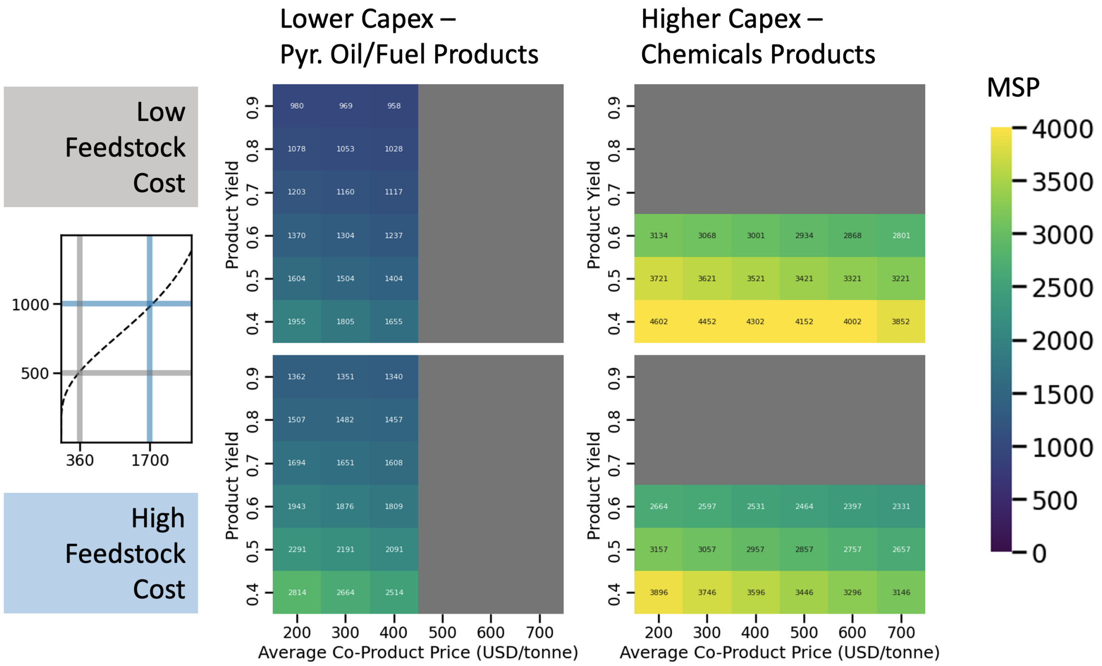


Figure 3-10: Minimum selling price as a function of product yields and co-product prices for 4 scenarios outlined in Table 3.2; high and low feedstock costs for fuels and chemicals production respectively.

Lower the MSP, greater the likelihood of technology viability as it allows more room for profitability. We find that, for oil or fuels product, the lowest MSP is approx. 980 USD/tonne (low feedstock cost) and 1360 USD/tonne (high feedstock cost). At low capex, feedstock costs influence operating margins more. On the other hand, for higher capex chemicals production, we find that, at low feedstock costs minimum selling prices are higher (>3100 USD/tonne) than at higher feedstock costs (>2600 USD), indicating that economies of scale is more influential than operating margins.

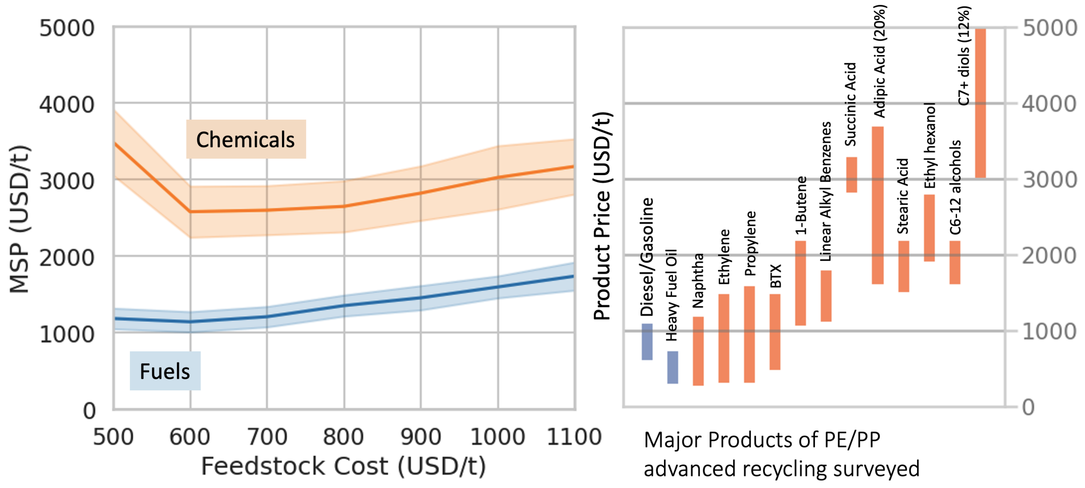
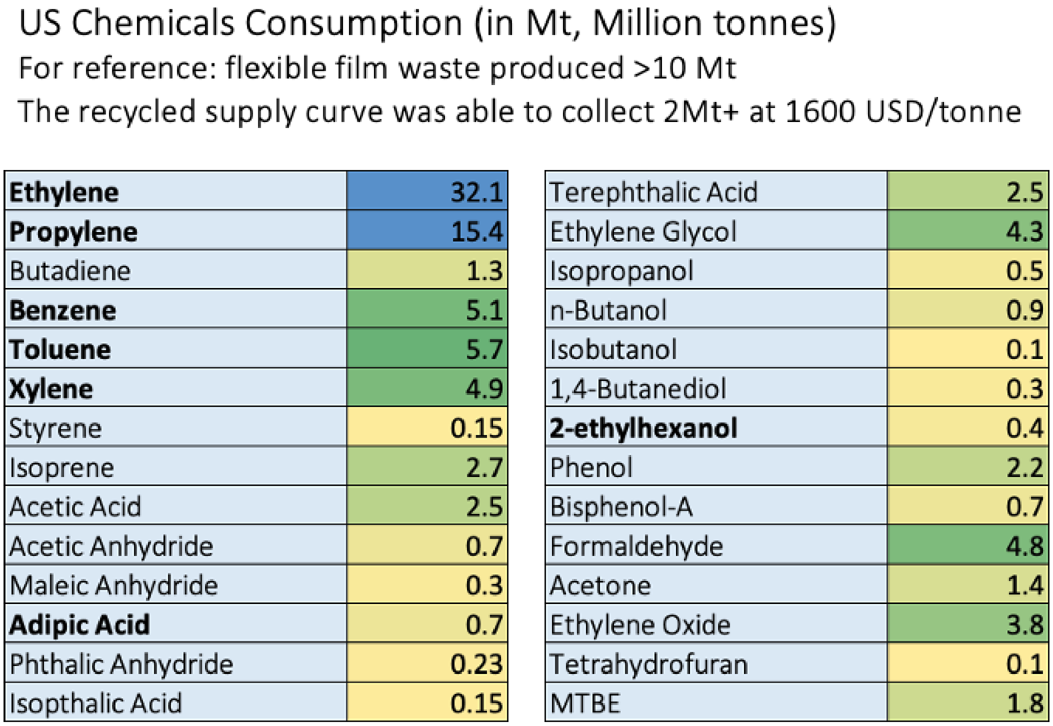


Figure 3-11: Minimum selling price as a function of feedstock price for generalized assessment of fuels and chemicals production.

Figure 3-11 visualizes variation in MSP as a function of feedstock costs for the 'fuels' and 'chemicals' cases. Plastics-to-fuels are only competitive with low feedstock cost for production of refined fuels like diesel or gasoline. Moreover, the monotonic relationship suggests that higher feedstock costs do not provide sufficient economies of scale benefits to offset direct operational expense of procuring feedstock. Chemicals production from plastic waste is more competitive in the 600-800 USD/tonne feedstock cost range, with MSP ranging between 2300-3000 USD/tonne. However, such high MSPs exclude most large volume starter chemicals such as ethylene, propylene, BTX, etc and is only accessible to more refined chemicals such as important dicarboxylic acids, alcohols, and diols fall within this price range. Figure 3-12 shows the list of all (C, H, O) containing chemicals produced at volumes greater than 1 Mt globally, along with the US consumption volumes (adapted from Nicholson et al. (Nicholson et al., 2023)).

Among the major products surveyed, only adipic acid and 2-ethylhexanol have sufficient



US Chemicals Consumption (in Mt, Million tonnes)

For reference: flexible film waste produced >10 Mt

The recycled supply curve was able to collect 2Mt+ at 1600 USD/tonne

Ethylene	32.1	Terephthalic Acid	2.5
Propylene	15.4	Ethylene Glycol	4.3
Butadiene	1.3	Isopropanol	0.5
Benzene	5.1	n-Butanol	0.9
Toluene	5.7	Isobutanol	0.1
Xylene	4.9	1,4-Butanediol	0.3
Styrene	0.15	**2-ethylhexanol**	0.4
Isoprene	2.7	Phenol	2.2
Acetic Acid	2.5	Bisphenol-A	0.7
Acetic Anhydride	0.7	Formaldehyde	4.8
Maleic Anhydride	0.3	Acetone	1.4
Adipic Acid	0.7	Ethylene Oxide	3.8
Phthalic Anhydride	0.23	Tetrahydrofuran	0.1
Isopthalic Acid	0.15	MTBE	1.8

Figure 3-12: US Consumption of Petrochemicals (adapted from (Nicholson et al., 2023))

market volumes to absorb additional plastic waste, provide competitive prices and support high feedstock costs to incent higher recovery. Even then, compared to the volume of flexible plastic packaging waste generated, assuming perfect displacement and increased mechanical recycling, maximum economically feasible recycling rates are <15% in the absence of external policy support or market incentives.

3.5 Discussion

3.5.1 Quantifying the limits of mechanical recycling

Numerous studies (Klotz et al., 2023; Lau et al., 2020) point out that flexible plastic packaging is a challenging plastic waste stream, but none quantify “how big” the challenge is. The lightweightness of the packaging material and the distributed nature of its use make the economic case for recycling flexible films bleak: more than 100,000 pieces of film must be collected to make 1 tonne of a film bale (RSE, 2014). Sorting film waste from recyclables is difficult as they tangle with machinery (Lubongo & Alexandridis, 2022) and most MRFs consider it contamination. In this study, we quantify the barriers to mechanically recycling flexible plastic packaging by first mapping the waste flows in the US and then estimating the cost of increased recovery and recycling. Comparing the cost of (mechanically) recycled PE supply with price of virgin LDPE (usually 1200-1600 USD/t), we find that only 850 kt of potentially $\geq$ 10 Mt of plastic waste can be recovered cost-competitively without additional incentives or subsidies. Our estimate is larger than what is currently recovered from flexible packaging waste (450 kt) and suggests that demand is lacking.

In Chapter 2, we described demand for recycled material as the willingness and ability of producers and manufacturers to substitute virgin material in their products. For waste from flexible packaging, this demand is sensitive to price as manufacturers prefer recycled material primarily for their inexpensiveness. The inexpensiveness of LDPE constrains prices achievable for recycled material – and demand below this price cannot meet the cost of increased supply beyond 850 kt, and thus it cannot incent large-scale change in the current recycling system to increase diversion of flexible packaging waste from landfills.

What we have quantitatively demonstrated in this chapter has been qualitatively discussed from numerous perspectives. In policy texts, lack of demand is investigated as the primary market failure of recycling (Milios, 2018; Nicolli et al., 2012). In polymer and process chemistry literature, increasing demand is instead discussed in terms of increasing value by producing higher value products. Researchers argue that mechanical recycling constitutes downcycling in quality and value (Ellis et al., 2021), and instead recommend considering plastic waste as a chemical feedstock (Jehanno et al., 2022) whereby advanced recycling technologies convert plastic waste into high-value fuels and chemicals (Xu et al., 2023).

3.5.2 Economic and environmental potential of advanced recycling

Advanced recycling promises demand for hard-to-recycle plastics including flexible plastic packaging and create value from plastic waste. To understand the extent to which novel technologies can incentivize higher recovery and diversion rates of flexible plastic packaging waste, I review TEAs of advanced recycling processes (and pyrolysis studies that produce fuels). I then use a generalized TEA model to delineate the conditions under which increased recovery can be expected and highlight these conditions to inform optimization of technical and economic parameters in experimental studies.

In economic assessments, value is characterized by price. Therefore, in our generalized TEA, we estimate minimum selling price as a function of system, economic, and technical parameters. We find that price ranges for large volume commodity chemicals that can be technologically feasible sinks for flexible plastic packaging are not economically viable, and that many refined chemicals are not produced in large enough quantities to match the scale of plastic waste generation. Current research is focused on using the plastic waste feedstock in innovative ways, but this approach is inefficient in finding relevant products that can competitively use alternative feedstocks. Instead, this analysis suggests that only few large volume chemicals can provide significant demand pull for system-wide change and improved recovery outcomes, but such chemicals are not produced at sufficient yields. Advances in catalytic chemistry, including a better understanding of reaction pathways, can help design processes with fewer steps and higher yields.

Advanced recycling also exposes trade-offs between waste diversion and GHG emissions. Yadav et al. (Yadav et al., 2023) show that chemical feedstocks and chemicals produced from catalytic pyrolysis of waste plastics are more emissions intensive than their fossil-derived counterparts. In their study, plastic-derived naphtha, BTX chemicals and ethylene produce 2x, 1.5x and 3.8x more GHG emissions than their fossil-derived naphtha, BTX, and ethylene. LCA studies of plastic-derived fuels report minor savings or emissions, depending on counterfactual assumptions: when plastic-derived fuels are assumed to avoid incineration fates, they are expected to provide GHG savings (Gear et al., 2018). However, production of plastic-derived fuels also release the carbon contained in plastics to the atmosphere, and given the short lifetimes of plastics used in packaging applications, such a release is just as instantaneous as carbon emissions from fossil-fuels relative to the geological timescales of carbon sequestration.

Chapter 4

Outlook

Disclaimer: Portions of this chapter are based on an article published in Resources, Conservation and Recycling by Basuhi R et al., titled "Environmental and economic implications of US postconsumer plastic waste management" and Appendix E: "Overview of Recycled Plastics Supply and Demand: Identifying the Critical Market Bottlenecks for Closing the Loop" written by Basuhi Ravi et al. for a report titled *Recycled Plastics in Infrastructure: Current Practices, Understanding, and Opportunities* published by the National Academies of Sciences, Engineering, and Medicine.

Plastics and their products are numerous in variety, and I examine two specific types of plastic packaging. PET bottles have been the poster child for plastic recycling for over three decades, and my analysis explores the cost of improving recycling rates through policy-driven demand. This perspective is rooted in existing legislation and expected consumer pressures as beverage producers contend with the dual challenges of availability and use of recycled PET. Flexible plastic packaging, on the other hand, is largely considered non-recyclable, and our supply perspective quantifies the cost barriers to recovery in the current recycling system. While advances in recycling technologies are expected to erode these barriers. I demonstrate that the mismatch in the scale of waste generation and advanced recycling capacities, even in the best-case scenario, leaves sizeable gaps in the circular economy ideal.

The two chosen case studies present extremities of the US plastic packaging recycling spectrum. However, despite dissimilar demand characteristics, the supply perspective critically examines the full system-wide costs of recycling and suggests a common denominator. Large-volume **thermoplastics used in plastic packaging are too inexpensive to be recycled cost-effectively**. The extent to which this common barrier limits recycling rates for a plastic type and product type then depends on the policy context and attendant market realities. In this chapter, I first discuss the need for market-based studies to quantitatively examine plastic waste as a resource. I then extend the insights from case studies in Chapters 2 and 3 to argue for a generalized supply-demand framework for all plastic types and packaging formats. Within this generalized framework, I describe the role of policy in overcoming barriers to recycling. Then, I consider the larger petrochemicals value chain and examine displacement factors beyond the recycling system that temper the expected environmental benefits of recycling. I also briefly describe alternative end-of-life fates for plastic waste in the US. Lastly, I conclude with the numerous information gaps I encountered in my study to highlight the need for information transparency to enable better analytical inquiries.

4.1 Waste as a resource & the need for market-based studies

Using waste "as a resource" is a central tenet of the industrial ecology discipline and underpins the ecological metaphor within. The circular plastics economy discourse can be understood as reframing this principle to propose actionable policymaking towards using plastic waste as a resource (De Smet et al., 2019). However, I find that current modeling efforts that seek to inform actionable strategies lack a distinct methodology to operationalize the plastic "waste as a resource" viewpoint. Material flow analyses characterize plastic flows in the economy, projections based on such flows emphasize the implications of continued growth, life cycle assessments estimate the environmental impacts of treatment and disposal pathways, and systems studies integrate impacts across flows under various optimistic and pessimistic scenarios. All recommend greater recycling, champion a holistic approach to recycling policy, and underscore the need for better markets for recycled plastics. But none engage these insights in their analytical studies. I posit that this is because the plastics recycling literature lacks a framework to consider plastic waste as a resource or commodity of prospective value.

In this dissertation, I borrow the concept of production cost curves from resource economics to fill this gap and quantify barriers to using plastic waste as a resource. I adapt this concept to build recovery cost curves for two plastic packaging types based on parameters reflecting recycling logistics, technology, and policy in the US. This concept can be easily generalized. Future work can use the framework exemplified by the two case studies to study other plastic products and packaging systems to advance effective solutions to the plastic problem.

4.2 Overcoming barriers

The quantitative supply-demand perspective adopted in this dissertation investigates barriers to clarify logistical and technological considerations that are unmet in the current recycling system for PET bottles and polyolefins in flexible plastic packaging. These considerations can be generalized:

- Plastic waste feedstock quantity barrier must be overcome to improve the availability of sorted polymer waste feedstocks at scale. This requires coordinated changes in collection behavior and policies.
- Plastic waste feedstock quality barrier must be overcome to increase the availability of clean, reasonably sorted polymer feedstock, and is a function of downstream collection policies and sorting capabilities as well as upstream product and material design.
- Recycling market barrier must be overcome by finding cost-effective and environmentally sound destinations of comparable scales for end-of-life plastics. For mechanical recycling, the height of this barrier depends on feedstock and recycled product quality relationships relevant to target application requirements. For advanced recycling, process viability depends on product yields and feedstock needs but trade-offs between waste diversion and GHG emissions pose a critical challenge.

The role of policy Plastic waste and recycling policy has and can play a significant role in overcoming these barriers. Figure 4-1 visualizes the impact of supply-side policy push and demand-side policy pull on the supply and demand curves.

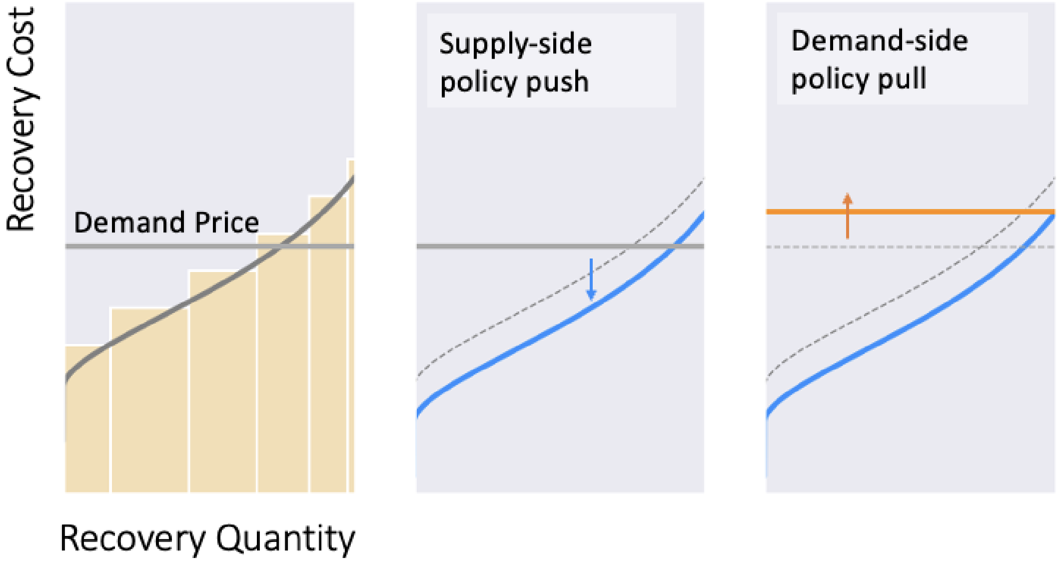


Figure 4-1: (A) Recovery supply curve and demand price. (B) Supply side policy push (such as subsidies) reduce cost of recovery, and (C) demand side policy pull increases price of recovered and recycled plastics.

For materials management, policy instruments can be classified based on implementation strategy (Bengtsson et al., 2010) – regulatory/command and control (seeking compliance), economic (creating market-based incentives), or informational/educational (enabling informed choices or altering choice architectures). In Table 4.1, I identify generalized policy instruments that are aimed at end-of-life handling of plastic waste.

Policy instruments differ in means and intended effect, and therefore, different instruments impact different stakeholders in the plastics value chain. To better understand the role of policy, Figure 4-2 shows relevant stages in the plastics value chain, as well as important variables that influence supply-demand dynamics for recycled plastics. Regulatory instruments upstream of use can impact the producer directly or indirectly (by targeting sales at retailers) while downstream product takeback requirements and landfill bans typically affect collection agents that must ensure waste is handled as regulated. Upstream regulatory action such as bans are expected to reduce waste generation and possible recycling (Q_{col}), while downstream action is predicated on increased recovery (Q_{col}). Informational instruments target consumers and responsibility for accurate information dissemination can be placed both upstream (product labeling by producers) and downstream (local consumer outreach). Depending on polymer type and product recyclability, informational instruments are either used to increase quantity (Q_{col}) or improve the quality of recycling (therefore impacting price, p_r).

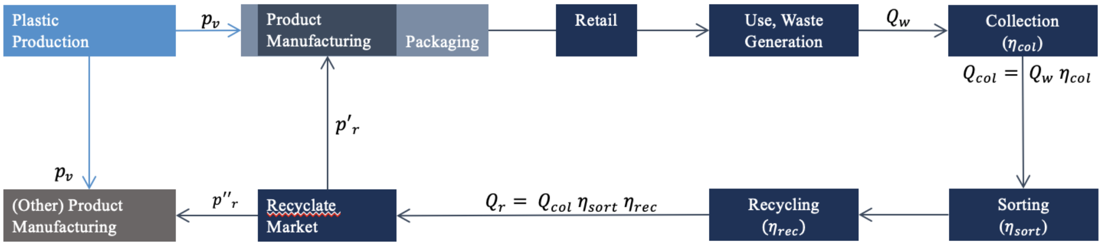


Figure 4-2: Cradle to cradle plastics value chain and relevant recycling variables for a generalized framework.

Economic policy instruments provide incentives that can be modeled as supply and demand responses. In Chapter 2, I model supply curves to simulate deposit return system expansion. Other instruments like pay-as-you-throw are intended to increase recycling but because such policies are not differentiated by product or recyclability, their effect on specific plastic types may be mixed – it can increase quantity but compromise quality due to greater contamination likelihood. Beyond quantity and quality, some instruments directly impact the cost drivers of recycling activities. Recycling subsidies can lower the cost of supply – making recycled plastics more desirable to price-elastic end-users, while demand-side instruments influence end-user preferences for recycled plastics by modulating recycled plastic prices relative to virgin plastic prices. Recycled content mandates and tradable recycling permits are both quantity-based policies that let the market set optimal prices for achieving recycled content targets, while virgin resin taxes directly influence producers' choices to use more recycled content.

Table 4.1: Policy instruments aimed at end-of-life handling of plastic waste.

Number	**Policy Instrument**	**Expected primary effect**	**Means**	**Type**
1	Deposit Return Systems	Expand collection to increase recycling	By making it easier to recycle	Economic
2	Product Take Back Requirements	Expand collection to increase recycling	By making it easier to recycle	Regulatory
3	Landfill bans	Landfill diversion	By making it difficult to trash/landfill	Regulatory
4	Pay-as-you-throw	Landfill diversion to increase recycling	By making it expensive to throw into trash	Economic
5	Advanced disposal fees, Environmental product taxes	Reduction in use	By making it expensive to buy, discouraging to buy	Economic
6	Product bans	Reduction in use	By making it difficult to buy	Regulatory
7	Recycling subsidy (direct or indirect)	Expand collection and encourage recycling activities	By making it cheaper to recycle and obtain recycled plastics	Economic
8	Virgin resin taxes	Discourage virgin plastics use and encourage use of recycled content	By making it expensive to use virgin plastics	Economic
9	Recycled content mandates	Develop recycled end-use markets and encourage use of recycled content	By making it expensive to not use recycled plastics (fines)	Economic
10	– Tradeable permits (like emissions trading system)	Develop recycled end-use markets and encourage use of recycled content	By making it cheaper to use recycled plastics (permit credits)	Economic
11	Product Labelling	Increase recycling quantity and quality	By informing consumers	Informational
12	Consumer Outreach	Increase recycling quantity and quality	By informing consumers	Informational

Policies addressing the concerns surrounding plastics range from local, community-driven actions to regional, national, and international scales. The Nicholas Institute's Plastics Policy Inventory tracks public policy documents and finds that approximately 900 policies that cover plastics across all life cycle stages have been enacted since 2000 (Nicholas Institute, n.d.). While international and national attention derives focus from marine plastic pollution (Diana et al., 2022), regional and local scales approach policymaking as a waste management issue, and instruments often overlap with other municipal solid waste fractions (Porter, 2010). Therefore, not all plastics-related policies lend themselves to the type of quantitative study undertaken in this dissertation, but where possible, mapping policy objectives to barriers and policy implementation mechanisms to supply or demand-side actions can be a useful tool in the system modeling toolbox.

4.3 Displacement

Barriers to recycling plastics are most evident when their recycling metrics are contrasted against other large-volume materials in use today. Many metals such as copper, aluminum, nickel, etc. are recycled at rates greater than >50% (Reck & Graedel, 2012). Compared to such materials, not only are recycling rates for plastics abysmal, but the effect of recycling outcomes on primary (or virgin) production is also unclear.

Metals recycling is and has been motivated by diminishing natural resources (often under national interest) and increasing costs of mining primary materials (Ayres, 1997). To that end, the displacement of primary metals is a key indicator of the utility of recycling activities (Atherton, 2007). In plastics recycling, on the other hand, even though scholars point out that fossil resource use in plastics is a problem that must be minimized, the emphasis on downstream disposal outweighs the upstream problems. This is not surprising. From an economic standpoint, petrochemicals (including plastics) started as a byproduct of the massive, global, energy industry – where primary feedstocks (petroleum, coal, natural gas) are plenty and their extraction is largely driven by ever-growing energy consumption. On the waste side, however, the explosive growth of plastics in the last six decades has disproportionately burdened local waste management systems. Coupled with the short lifetimes of packaging applications, plastics' ubiquity makes them uniquely visible at end-of-life – the plastic waste problem appears more significant. While the two problem-frames hold different policy objectives, without displacement of primary materials, neither of these objectives can be effectively achieved. As Geyer et al. demonstrates in "Common Misconceptions about Recycling" (Geyer et al., 2016), diversion from disposal without displacement merely distracts from both upstream and downstream problems.

What is displacement?

Displacement, as a concept, stems from the desire to “close material loops” within the industrial ecology metaphor. More concretely, Zink et al. define it as “the change in primary production quantity (Q_p) caused by a change in secondary production quantity (Q_s))”, presenting a simple equation that belies a complex phenomenon (Zink et al., 2016):

$$d = \frac{-\Delta Q_p}{\Delta Q_s}$$

They further simplify the complex phenomenon using a toy model. Their toy model employs price responses of primary (virgin) and secondary (recycled) supply and demand within a partial equilibrium approach and derives conditions for zero, positive, or negative displacement. From this simple modeling exercise, Zink and Geyer posit that displacement can be zero if (a) either the primary market is unresponsive to selling prices, or (b) primary demand is insensitive to changes in the price of secondary material (Zink et al., 2016). While (a) is unlikely for competitive recycled commodity markets such as metals, markets for secondary plastics are seldom as competitive against virgin plastics due to structural reasons such as conventional procurement practices. Second, (b) is also highly likely in recycled markets where buyers of primary/virgin material show little willingness or ability to switch to secondary/recycled material. In other words, (b) is a condition on the (perceived and real) inferior quality of recycled material. Since quality is a significant challenge for recycled plastics (Demets et al., 2022), low displacement values translate to tempered environmental benefits are likely.

Is substitution displacement?

Extending the industrial ecology metaphor to early product life cycle assessment (LCA) methodologies, the ‘avoided burden’ or ‘substitution’ approach of accounting for multi-functional product systems was developed (Weidema, 2000). Note that the functional unit, then, is the product. In this view, recycling yields another output or function from the product system (across its cradle-to-grave life cycle) that is assumed to substitute an equal quantity of the same or different material through ‘system expansion’; experts debate on how far the system must be expanded to predictively incorporate all the consequences

(Finnveden et al., 2022). Notably, early studies of the recycling conundrum in LCA method development literature discuss market-based approaches, presenting elaborate arguments speculating on expected changes in the system based on supply and demand mechanics (Ekvall, 2000). In such cases, substitution factors abstract displacement. However, in much of plastics recycling literature, substitution is not predicated on market responses.

Even though substitution should be the LCA way of factoring in displacement, substitution factors used by plastics recycling and waste management LCA studies do not reflect displacement. Because empirically quantifying displacement is difficult and requires copious data on past trends, substitution factors rely on the next best alternative: variables that theoretically should influence displacement. Recycled quality is measured as technical substitutability (Demets et al., 2021; Golkaram et al., 2022), recycled/virgin price ratios (Rigamonti et al., 2020), or relative market shares (Eriksen et al., 2019; Horodytska et al., 2020) indicating market substitutability, and combinations of such fractions (Vadenbo et al., 2017) have been used in various studies. While various approaches can be justified in the absence of quantifiable evidence on displacement, substitution factors embody assumptions about the system-wide implications of recycling. In a review of more than 50 LCA studies about plastics recycling in the European context, Lazarevic et al. (Lazarevic et al., 2010) found that assumptions about virgin material substitution ratio (that varied between 0.2 and 1.0) significantly influenced environmental impact results. Therefore, market-based modeling of displacement is necessary to accurately estimate the substitution benefits of recycling. However, the supply-demand perspective used in this dissertation only provides a first step towards displacement modeling. To rigorously assess displacement, we must inquire how recycling affects production decisions upstream in the plastics value chain.

The larger petrochemical context

Ethylene is a starter chemical for many large-volume polymers. It can be produced from natural gas feedstocks as well as petroleum refining fractions - naphtha and gasoil (James G. Speight, 2021). Ethane from natural gas is the preferred feedstock for ethylene production to maximize yields and minimize byproducts. Heavier hydrocarbon feedstocks, on the other hand, require a lot of feed per tonne of ethylene produced (Levi & Cullen, 2018).

However, since byproducts are high-value chemicals such as propylene, BTX, etc, some producers choose to diversify their product portfolio and not optimize for ethylene alone. More importantly, prospects of higher yield or diverse product mix must be balanced with regional availability of feedstock. To understand whether higher recycling rates lead to displacement, we must understand decisions in the larger petrochemical supply chain (see Figure 4-3).

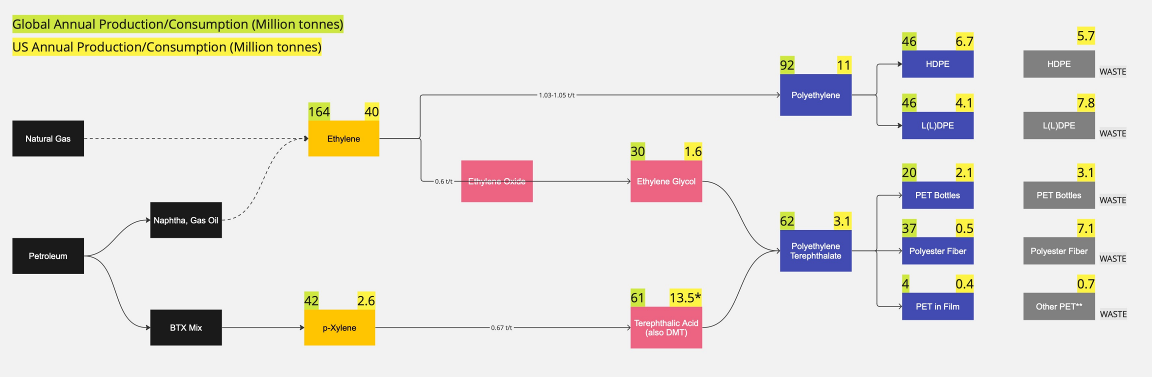


Figure 4-3: Upstream petrochemical feedstocks and value chain for plastics. Data from US Energy Information Agency, (Levi & Cullen, 2018), Amercian Chemistry Council, and US EPA

The EIA estimates that "rising demand and continuing growth in US natural gas production are projected to increase US ethane production" (US Department of Energy, 2022). Investment in export infrastructure for both ethane and ethylene continues to grow (US Department of Energy, 2022) as economics favors ethane over naphtha when crude oil prices are high. Moreover, from a refinery's perspective, the production of low-value naphtha as a petrochemical feedstock is subsidized by other lucrative fractions such as motor gasoline and diesel. Naphtha is priced lower than crude oil i.e. it offers negative margins. In other words, a drop in demand for either (driven by the clean energy transition) is likely to lower naphtha supply and increase naphtha prices, further making ethane an attractive option. However, ethylene is not the only major basic chemical; >C3 olefins and BTX, etc. are valuable petrochemical commodities that continue to rely on naphtha feedstocks. Besides vertically integrating refining and petrochemicals production, refinery actors elsewhere are also pursuing crude oil-to-chemicals pathways, bypassing the refinery step but the scalability required to slide down the experience curve is estimated to be decades away. In the short-term, two opposing forces are at work – as demand for fuel fractions from refineries drops, initially more naphtha may be produced, decreasing its prices but in the medium term, to maintain refineries' profitability, naphtha feedstocks are likely to become more expensive. This would increase the price of virgin polymers made from monomers derived from such feedstocks, and for our purpose, make recycled plastics more competitive.

However, ethylene is predicted to be increasingly made from ethane (US Department of Energy, 2022). Natural gas, the cleanest of all fossil fuels, is widely considered to be the bridge to the renewable energy future (IEA, 2023) – and ethane being its byproduct is expected to remain inexpensive and abundant. If ethylene capacity was driven by ethane abundance, the supply dynamics can clamp ethylene prices, including the downstream chemicals and plastics that derive from it. Therefore, in the US context, recycling plastics derived from ethylene becomes even less attractive. While naphthenic feedstocks can adjust production to produce more of the non-C2 streams, ethane is inflexible (other than rejection to natural gas streams). In summary, the ethane-ethylene landscape faces opposing supply and demand considerations. Recycling should (ideally) displace equivalent ethylene capacity in the US if growth in demand is largely due to short-lived plastic products and packaging that can also be easily recycled. But while domestic demand for ethylene is not growing much, US capacity for ethylene has grown significantly. This mismatch arises not from demand forces, but from feedstock supply dynamics for ethylene production. Moreover, trade complicates the picture in two ways. First, overseas ethylene demand is expected to grow faster than in the US. But more importantly, ethane-sourced ethylene offers higher margins than naphtha-sourced ethylene due to larger, inevitable changes in the world energy landscape.

If the generated waste is collected and recycled, it should (ideally) displace production. Traversing up the dendritic petrochemical value chain, displacing PET and PE production should (ideally) displace some fraction of ethylene capacity. However, the interactions between chemicals and energy supply and demand complicate predictions of displacement of virgin plastics or petrochemicals. When considering demand for recycled plastics, they not only pose a significant barrier to the expansion of recycling infrastructure in the short term but also challenge many of the assumptions made when assessing the environmental benefits of recycling.

4.4 Beyond recycling: alternative end-of-life fates

In this dissertation, I focus on recycling as a (partial) solution to the plastic waste problem. Within this scope, I discuss mechanical recycling, advanced recycling, and fuel recovery. However, these are not the only fates available to waste plastics, and a brief discussion of other fates is warranted to accurately describe recycling among the suite of possible end-of-life pathways.

Energy Recovery

Energy recovery, where non-recyclable plastics are combusted along with other municipal waste fractions, can drastically decrease the waste being sent to landfills while producing electricity and low-grade heat for local applications. Depending on landfill space availability restrictions, it is the preferred method of disposing non-recyclable municipal wastes (including plastics) in several counties in the US (Kaza et al., 2018). However, this pathway is criticized for several reasons. First, energy recovery releases the fossil-based carbon in plastics. For short-lifetime plastic packaging, the time from the extraction of hydrocarbons to the atmospheric release of carbon dioxide is not much longer than for direct use of fossil fuels. In LCA studies, the degree of adverse GHG impacts of energy recovery against recycling or landfilling depends on credits assumed for displacing electricity (Anshassi et al., 2021; Lazarevic et al., 2010) – and benefits wrought for a more fossil-intensive grid are expected to diminish over time as the US grid is expected to become greener. In the longer term, therefore, energy recovery pathways for plastic waste disposal will not be a sustainable solution. Figure 4-4 shows the total GHG emissions of combusting all non-recyclable waste plastics for energy recovery in the US as a function of the energy efficiency of electric combustion.

Some researchers argue that biobased production of feedstocks for plastics, and subsequent combustion for energy recovery, can be considered approximately carbon-neutral because

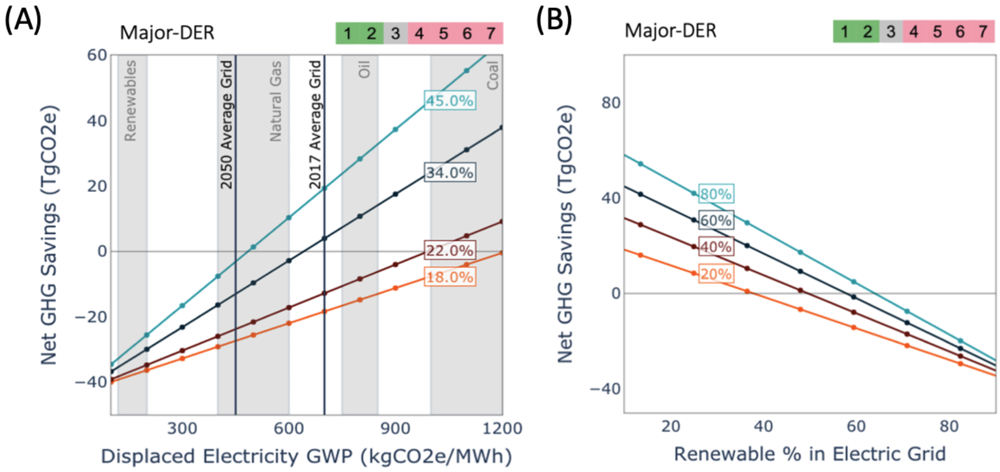


Figure 4-4: (A) Net GHG Savings as a function of GWP of displaced electricity in scenario Major-DER at 100% collection rate for different electric efficiencies energy conversion. (B) Net GHG Savings as a function of % renewables in electric grid in scenario Major-DER at 100% collection rate for different levels of CHP usage (H/P ratio = 5, total efficiency = 70% and electric efficiency of non-CHP plants = 30%). Positive is savings, negative is emissions. Figure reproduced from Basuhi et al., 2021, published in Resources, Conservation, and Recycling.

of embodied biogenic carbon (Meys et al., 2021; Stegmann et al., 2022b; Stegmann et al., 2020). However, the relative timescales of plastic waste generation from short-lived packaging compared to natural carbon cycles challenge this simplistic view. Critics also cite the numerous environmental justice issues caused by past energy recovery projects in the US (Bullard, 1993) to caution against further development of this pathway (Rootes & Leonard, 2009).

Biodegradation

Biodegradable plastics provide an alternative pathway where end-of-life plastics are degraded and assimilated in the environment at rates orders of magnitude faster than conventional plastics. Biodegradable plastics can be bio-based (such as poly-4-hydroxybutyrate) or fossil-based (polybutylene adipate terephthalate); they can be degraded by oxidation, hydrolysis, or enzymatic processes, and adhere to different standards for biodegradability (Kim et al., 2023). The critical challenge in developing and scaling biodegradable plastics for packaging is the paradoxical need to match the robust chemical, barrier, and mechanical properties of conventional plastics during use while also breaking down from environmental forces at end-of-life (Ghosh & Jones, 2021).

However, the objective of biodegradation pathways is different from recycling or incineration. The latter is pursued to decrease the quantity of waste sent to landfills and recover material and energy resources at end-of-life. Unless also compostable, biodegradation cannot directly divert waste from landfills or recover industrial resources. Instead, biodegradable plastics are designed to minimize the persistence of polymer particles in the environment and solve problems related to plastic pollution. While some studies argue that biodegradation can safely return carbon to the biosphere, numerous studies caveat that not all biodegradation products from degradable plastics can be considered environmentally benign (Degli Innocenti & Breton, 2020; Qin et al., 2021). More research on the assimilation and mineralization of fragmented, depolymerized plastic waste is required (Kim et al., 2023).

Reduce and Reuse

Reduce and reuse levers are direct actions that can reduce plastic waste and its numerous attendant problems, but their cost-effective implementation is not well-studied. Barnett et al. (Barnett et al., 2023) find that even though reducing waste generation is far more sustainable than recycling plastics, survey participants perceived the opposite, and a recycling bias is pervasive. Plastics are lightweight, and producers that use plastic packaging are incentivized to optimize use across economic and functional considerations – any additional source reduction would require added incentives. However, a study of outcomes of EPR laws across nations in Europe showed that reductions realized were insignificant (Joltreau, 2022). As for reuse, while some options have been designed (Ellen MacArthur Foundation, 2019), adoption of such products remains low. Because the convenience of plastic packaging lies in its disposability, an expensive, inconvenient alternative is less desirable.

Literature on recycling is vast and growing, but the same focus is not extended to source reduction and reuse strategies to combat plastic waste problems. However, projections of future plastic use and waste generation trends suggest that recycling will continue to be a partial solution (Lau et al., 2020; Zheng & Suh, 2019).

4.5 Quantification: a note on data and information

One of the constant challenges in my efforts to quantify barriers to plastic recycling was the lack of pertinent data and reliable sources of information. In academic literature, this challenge is most evident in material flow analyses of plastics in the US, where researchers have noted that granular data on collection practices, grades, and end-use applications is wanting (Di et al., 2021). In this dissertation, as I build off of uncertain material flows to understand the system interactions of waste supply and recycled plastics demand, this challenge is compounded. Such a systems perspective recognizes multiple perspectives and interests, traces relationships between interested stakeholders, and factors decision-making based on market signals modulated by policy. For plastics recycling, much of this information resides outside peer-reviewed literature. Municipal waste audits, policy position papers, state-level container refund accounting, trade association publications, corporate sustainability progress reports, and legislative records provide fragmented insights into the current workings of the plastics recycling system at large. There are two key caveats to using this class of information. First, source credibility is often undermined not only by the lack of a peer-review process but also because details about the scope and methodology of data collection are often unclear. Second, collecting information is expensive, and is often done towards a purpose (such as persuading public or policy action, influencing investment, etc.) that may or may not bias reporting and interpretation.

Plastic recycling has been a contentious topic in popular discourse for the period of this study (Crunden, 2021; Sullivan, 2022; Sullivan et al., 2022). To be objective in my analysis, I have attempted to corroborate and critically vet non-peer-reviewed information to the extent that it influences model results or major themes of discussion. When that proved difficult, I relied on sensitivity analysis to explore possible tipping points. Designing cost-efficient pathways to improving recycling outcomes in the US requires models that can incorporate policy and technology levers directly. To this end, this work strives to bridge academic literature focusing on technology development, life cycle assessment, and scenario study along with the many contextual factors embedded in gray and practitioner literature. However, despite my best efforts, data gaps present a significant limitation. Below, I highlight some limitations that can be addressed to gain a more complete understanding of the plastics recycling system.

Data Gaps

While there is national-level information about waste collection and some state-level accounting, organized municipal-level data on waste collection remains rare. Some state environmental departments (e.g., California) release county or municipal-level waste generation and collection estimates but comprehensive data across the US is non-existent. Data about participation rates, and municipal or county-level policies such as collection frequency, funding models, MRF contracts, accepted materials, etc. are also lacking. Such information can be instrumental in quantifying the impact of policies in improving collection and recycled materials supply. Understanding the effectiveness of these policies will go a long way towards designing strategies to overcome the main bottleneck in the recycling system: stagnant collection.

While new infrastructures and standards are likely needed to measure and collect some of the data described above, other kinds are traceable but are currently not reported openly. For example, deposit return systems involve monetary exchanges between municipalities, manufacturing, and redemption centers, and budgets and accounts can be queried to track bottles collected at redemption centers. In turn, this can help better design the network of deposit centers. For sorting, data on the costs of individual MRFs may be proprietary but processing fees, paid by municipalities to MRFs to cover their cost, serve as a proxy to analyze changing trends in packaging waste recyclability, MRF viability, and system effectiveness. For successful implementation of extended producer responsibility laws, collation and tracking of information on the collection and sorting of plastics will be imperative.

A large amount of waste generation and diversion takes place from commercial sources, but very little information exists about the volumes of these flows. California conducted a waste audit of commercial facilities in 2014 (and has mandatory commercial recycling rules) but remains the only state to release this information (CalRecycle, 2014). A better understanding of waste types and quantities from commercial sources as well as the economics of waste collection in business-to-business contracts can improve overall recycling rates by targeting point sources with promising economies of scale and existing logistics networks.

To comment on the economic viability of the system, it is crucial to know how much the different end-use markets might be willing to pay for recycled materials in different scenarios. Very little is known about the demand for various grades of plastics by end-users. We use price data with recycled plastic consumption trends to tease some insights into demand. However, better information is needed to predict future demand and understand the price elasticity of demand. While an individual company may not be willing to report how much they are willing to pay for plastics, information on the demand of different industries on aggregate can be collected and reported.

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Zheng, J., & Suh, S. (2019). Strategies to reduce the global carbon footprint of plastics. *Nature Climate Change*, *9*(5), 374–378.

Zink, T., Geyer, R., & Startz, R. (2016). A Market-Based Framework for Quantifying Displaced Production from Recycling or Reuse. *Journal of Industrial Ecology*, *20*(4), 719–729.

Zink, T., Geyer, R., & Startz, R. (2018). Toward Estimating Displaced Primary Production from Recycling: A Case Study of U.S. Aluminum. *Journal of Industrial Ecology*, *22*(2), 314–326.

Appendix A

Appendix to Chapter 2 - Supply

A.1 Building Supply Curves

We identify two types of reverse logistics systems for PET bottle packaging waste in this work: curbside collection and deposit return system. We model both for the 48 states in contiguous United States, covering 98.2% of the population. Figure A-1 shows the end-of-life value chain of PET bottle waste in these two types of systems from use and collection to recycling and depicts the modular make-up of the supply model. Each system and the operations within are discussed in detail in the following subsections.

A.2 Deposit Return System (DRS)

Deposit return system (DRS; also referred to as deposit refund scheme, beverage container deposit law or more informally as bottle bills in the US) is a packaging waste take-back system where a deposit fee is added to the sale price of the product. This fee can be redeemed when the consumer returns the empty packaging to the take-back structure in place (select retail centers, dedicated deposit centers). We model PET bottle collection from state-instituted deposit programs to PET bales, transportation to reclaimers and the conversion of bales into flakes and food grade pellets for subsequent use as recycled content.

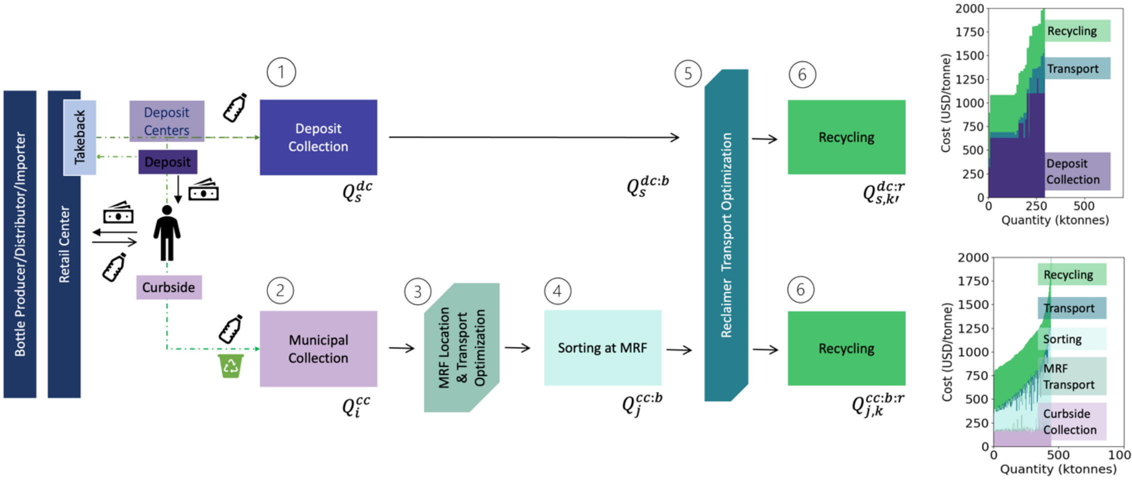

Figure A-1: Schematic of the U.S. PET recycling system along with labels for material flows, recycling supply chain actors and legend for indices used in the supplementary document to describe the methodology. MRF = Material Recovery Facility, CCSD = Census County Subdivision

A.2.1 Deposit Collection

We model the mass of PET collected by deposit programs in each state as the product of PET waste generation in that state (G_s), program coverage (c_s, percentage of containers generated that are eligible to be returned) and redemption rate (r_s, percentage of eligible containers returned via the deposit system).

$$Q_s^{dc} = G_s * c_s * r_s \tag{A.1}$$

Data for PET waste generation, for all states in 2018 is extracted from The 50 States of Recycling Report[1]. Data for coverage rate, redemption rate and deposit value for each state in baseline (2018) is compiled in Table A.1 below. Not all products packaged in PET bottles are eligible for refund under a deposit system. Therefore, coverage rate is typically less than 100%. Redemption rate also varies widely between states – from as low as 50% in Massachusetts, Connecticut to as high as 91% in Michigan. Data from US states and EU

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[1] Eunomia. The 50 States of Recycling; A State-by-State Assessment of Containers and Packaging Recycling Rates; 2021. https://www.eunomia.co.uk/reports-tools/the-50-states-of-recycling-a-state-by-state-assessment-of-containers-and-packaging-recycling-rates/ (accessed 2021-11-15)

countries show evidence of correlation between redemption rate and deposit fee (Figure 2-2). Capturing this correlation by curve-fitting, the redemption rate r_s is modelled as a function of deposit value shown in equation A.2:

$$r = f(d) = 1 - e^{-18d} \tag{A.2}$$

Where d is the deposit value per unit container (\$/bottle). We fit the function $r_s = 1 - e^{-a*d}$ to data on redemption rate (r_s) and deposit value (d) and use this to estimate the value for the parameter a (median value of a across deposit states and countries was estimated to be $\sim$18). With this functional form, when the deposit redemption rate is 0, there is no deposit program ($r_s = 0\ when\ d = 0$). It is assumed that the redemption rate would be 100% if there was an infinite deposit value ($r_s = 1\ when\ d = \infty$).

Table A.1: Handling fees, deposit value, coverage rates and redemption rates for states and territories with deposit return systems in the United States. Greyed out regions are not considered in this contiguous US model. Data from Container Recycling Institute and Global Deposit Book

STATE	Handling Fee (\$/bottle)	Deposit Value in 2018 (\$/bottle)	Coverage Rate (% of bottles eligible for deposit collection in 2018)	Redemption Rate (% of eligible bottles collected in 2018)
CALIFORNIA	0.02	0.075	0.87	0.67
CONNECTICUT	0.02	0.05	0.77	0.44
GUAM	0.03	0.05		
HAWAII	0.035	0.05	0.88	0.62
IOWA	0.01	0.05	0.65	0.65
MAINE	0.04	0.05	0.91	0.84
MASSACHUSETTS	0.02625	0.05	0.42	0.43
MICHIGAN	0.025	0.1	0.57	0.89
NEW YORK	0.035	0.05	0.78	0.65
OREGON	0.02	0.1	0.88	0.86
VERMONT	0.035	0.05	0.48	0.77

In our future scenarios, we gradually expand the deposit return system across the contiguous United States. Table A.2 shows the DRS implementation details and parameters.

Table A.2: Table S2: DRS expansion scenarios, and associated parameters.

Scenario Year	DRS Implementation	Deposit Fee	Deposit Coverage
2018	9 states marked black in Table S2; all other states do not have DRS	Same as in Table S2	Same as in Table S2
2025		10 cents for the 9 states in Table S2	90% for the 9 states in Table S2
2030	All states have DRS	10 cents for the 9 states in Table S2 + 5 cents for the other 37 states	90% for all states
2035		10 cents for all states	

A.2.2 Recycling

No sorting is assumed to be required in deposit collection which is source-separated by design, and bales of PET bottles are directly assumed to be sent to the recycling facility. The bales collected in each state are transported to a reclaimer, and the transportation distance is estimated by a linear optimization algorithm that minimizes transport distance, subject to capacity constraints. Each state can send bales to multiple reclaimers, and conversely multiple reclaimers receive bales from multiple states. Nationwide modeling of waste routing to PET recycling facilities (Box 5 in Figure A-1) is done simultaneously for both the curbside stream from material recovery facilities (MRFs) and the deposit stream from deposit centers assumed to be located at population weighted state centroids.

Each reclaimer recycles the PET bales into r-PET pellets for sale to manufacturers (Box 6 in Figure A-1), with some loss due to contamination and efficiency. We assume all reclaimers of a specific grade (food-grade or otherwise) have the same yield y_d.

In the deposit supply curve, each bar is the R-PET production quantity of a supplier ($Q^{dc:b:r}_{s,k'}$), where suppliers are model optimized set of state-reclaimer (many-to-many) pairings with baled material $Q^{dc:b}_{s,\ k'}$ being transported from state s to reclaimer k'.

$$Q_{s,k'}^{dc:b:r} = \ y_{dc} * \ Q_{s,\ k'}^{dc:b} \tag{A.3}$$

Deposit collected material is assumed to have a higher yield ($y_{dc} \ = \ 0.9$ 25) compared to curbside collected material ($y_{cc} = 0.73$) following literature estimates (Andreasi Bassi et al., 2022; Heller et al., 2020; Roosen et al., 2020; Smith et al., 2022). To estimate $Recycling\ Cost_{s,\ k'}^{per\ tonne}$, Recycling economics is modeled as described in our previous work (Basuhi et al., 2021), and a food-grade premium of 220 USD/tonne is applied to food-grade end use after supply-demand clearing (described in appendix B) – note that only deposit collected material is eligible for food-grade use in this work.

A.3 Curbside Channel

A.3.1 Curbside Collection

Curbside collection accounts for two-thirds of PET collected in the United States. We use existing estimates of total PET recycled in each state s, $Q_s^{recycled}$ as well as a yield factor ($y^{cc} = 0.73$) accounting for sorting and recycling losses. Adjusting for deposit-based recycling in states that have a deposit return program, we find total curbside collection quantity in a state, Q_s^{cc}:

$$Q_s^{cc} = \frac{Q_s^{recycled} - \ \ \sum_{k'} Q_{s,k'}^{dc:b:r}}{y^{cc}} \tag{A.4}$$

However, curbside collection is a municipal activity, which is decided by collection policies set at the municipal level (such as single stream, automatic access, etc). Few states release data on curbside collection quantities for PET for each locality/municipality. Therefore, we developed a bottom-up curbside collection model, as described below.

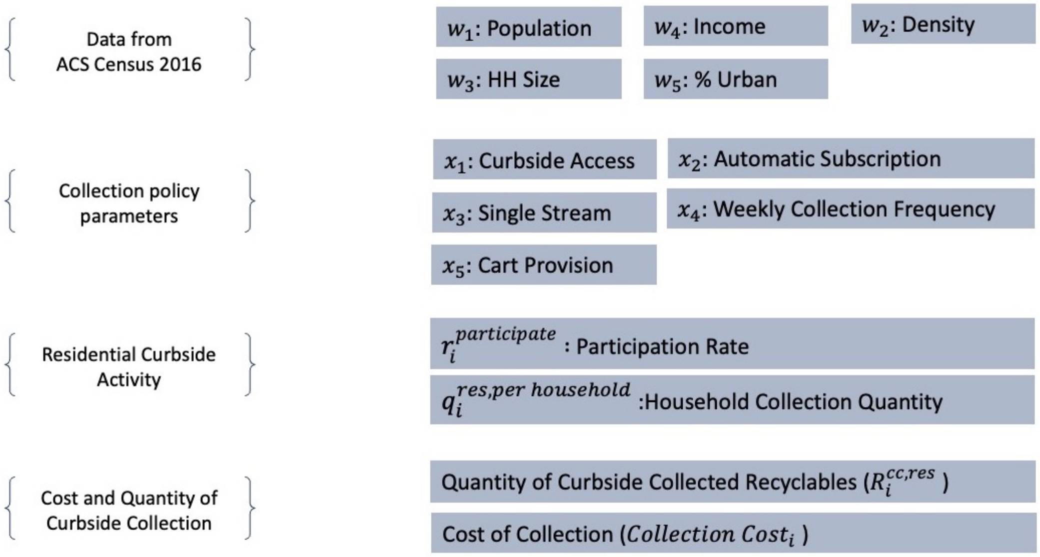

Figure A-2: Schematic describing the four-step bottom-up methodology of predicting curbside collection cost and quantity. Census variables for each county subdivision are used to predict local curbside collection policy variables which are correlated to collection activity in that county subdivision CCSD i. Collection activity is used to estimate the quantity and cost of recyclables in the curbside PET supply curve.

Using census variables to predict curbside policy.

We used regression models to estimate statistical relationships between 5 policy variables, 2 activity variables and the 6 census variables (shown in Figure A-2). Data for this regression is based on curbside collection data from 552 municipalities in North Carolina[2], 352 municipalities in Massachusetts[3] and 368 sampled municipalities in the 2016 State of Curbside Recycling Report[4]. To map to US census terminology, we use census county sub-divisions (CCSDs) as the level of geographical resolution for this curbside model. This choice is made to ensure that the representative geographical unit covers a significant fraction of the population, and its boundaries do not change significantly. Table A.3 describes the statistical relationships used to predict collection policy and activity. All census variables are normalized within a state following the example shown in Equation A.5.

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[2] North Carolina DEQ. FY18-19 North Carolina Solid Waste and Materials Management Annual Report

[3] Massachussetts DEP. Recycling & Solid Waste Data for Massachusetts Cities & Towns

[4] The Recycling Partnership. 2020 State of Curbside Recycling; 2020.

$$w_{1i} = \frac{(Population_i - E[Population_i])}{SD\ [Population_i]} \tag{A.5}$$

Where $E[Population_i]$ and $SD\ \ [Population_i]$ denote the mean and standard deviation of all county subdivisions within a state, which are indexed by i.

Table A.3: Logistic regression and cut-off conditions for collection policy variables.

Collection policy	Logistic regression result	Cut-off condition
Curbside Access	$\ln\left(\frac{p_i}{1-\ p_i}\right) = 3.8 * w_{1i} + 0.5 *\ w_{2i} - 0.23 * w_{3i} + 0.26 * w_{4i} + 0.32 *\ w_{5i}$	Percentage of the population having curbside access in each state.
Automatic opt-in	$\ln\left(\frac{p_i}{1-\ p_i}\right) = 0.36 * w_{3i} + 0.28 *\ w_{4i}$	60% of the population with access to curbside is assumed to have automatic opt-in facility.
Single-stream collection	$\ln\left(\frac{p_i}{1-\ p_i}\right) = 7.56 * w_{1i}$	90% of the population with access to curbside is assumed to have single stream recycling.
Weekly Collection Frequency	$\ln\left(\frac{p_i}{1-\ p_i}\right) = 0.199 * w_{3i} + 0.232 *\ w_{4i}\ -\ 0.51 *\ w_{2i}$	60% of the population with access to curbside is assumed to have weekly collection frequency (rest is every other week).
Cart provision	$\ln\left(\frac{p_i}{1-\ p_i}\right) = 0.64 * w_{4i}$	60% of the population with access to curbside is assumed to have cart provision.

Logistic regression is performed, and regression coefficients are estimated using the ‘glm’ function in R (within the ‘stats’ package); only the statistically significant estimators are retained. For policy variables, logistic regression outputs a probability of a policy existing for a CCSD, which is then interpreted as a binary yes/no using a cutoff probability value.

A population-based cut-off approach is employed - CCSDs are ranked according to their probability of curbside access and cutoff is determined such that the sum of population of top-probability CCSDs is equal to known population having the estimated curbside policy (such as access, opt-in, single stream, etc.) in each state. Therefore, the cut-off value is state-dependent and is derived from other known information[5], detailed in Table A.3.

Using census and policy variables to predict household curbside collection activity.

Collection activity is described by:

- participation rate, which is the number of households that participate in recycling, and
- household collection rate, which is the quantity of recyclables collected from each household.

Multiplying participation rate and household collection rate gives the overall collection rate for CCSD i. The two factors are distinguished because they are affected by different census and policy variables as described below.

Estimating Participation Rate: We used estimated participation rate distributions from the MA and NC datasets and found a natural split between municipalities that have automatic subscription to curbside collection (where the friction to participate is lower) and those that don't. We fit Beta distributions to the two mutually exclusive and exhaustive subsets.

For automatic subscription, we assume participation rate of CCSD i to be, $r_i^{participate} \sim \beta(4, 1)$ such that the expected value of the distribution is 0.8 (in other words, average participation rate for automatically subscribed CCSD is assumed to be 80%).

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[5] American Beverage Association. American Beverage Association. Community Recycling Survey (2008), The Recycling Partnership. 2020 State of Curbside Recycling; 2020

For CCSDs without automatic subscription, we assume $r_i^{participate} \sim \beta(4,8)$ such that the expected value of the distribution is 0.33.

From these distributions, samples are drawn, and an ordinal mapping is employed. We use a likelihood function ($lf_i^{participate}$) that depends on average household size (w_{3i}) and income (w_{4i}) of CCSD i to assign samples to CCSDs.

$$lf_i^{participate} = \ .037 * w_{3i} + .061 * w_{4i} \tag{A.6}$$

The higher the likelihood function value of CCSD i, the higher the rank of that CCSD and the higher value of sampled participation rate is assigned.

Estimating Participant Household Collection Quantity: Each participating household typically places somewhere between $100 - 200$ kgs/year of recyclables[6] in their bins/carts – we define the average quantity collected from each household in a year in CCSD i as $q_i^{res,\ per\ household}$.We find that the logarithm of this quantity is lognormally distributed and is a function of income, availability of single stream services, and whether cart provision is available, and model it accordingly:

$$\ln(q_i^{res,\ per\ household}) = -2 + 0.34 * \ x_{5i} + \ 0.29 * x_{3i} + 0.13 * w_{4i} \tag{A.7}$$

Estimating residential collection quantity in a CCSD

Using predicted values of participation rate and household collection quantity, we estimate baseline collection quantity of all recyclables in each CCSD i, $R_i^{cc,res-baseline}$:

$$R_i^{cc,res-baseline} = q_i^{res,\ per\ household} * r_i^{participate} * \frac{w_{1i}}{w_{3i}} \tag{A.8}$$

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[6] The Recycling Partnership. 2020 State of Curbside Recycling; 2020

In all scenarios, total residential PET collection in a state $Q_s^{cc,\ res}$ (Equation A.9 below) is used to proportionally scale $R_i^{cc,res-baseline}$ to estimate residential PET collected quantity in each CCSD i, $Q_i^{cc,res}$:

$$Q_i^{cc,\ res} = Q_s^{cc,res} * \frac{R_i^{cc,res-baseline}}{\sum_{\boldsymbol{i}\ \in\ \boldsymbol{s}} R_i^{cc,res-baseline}} \tag{A.9}$$

Estimating $\boldsymbol{Q_s^{cc,res}}$: Residential collection quantity in a state $Q_s^{cc,\ res}$ can be estimated by adjusting total curbside quantity (Q_s^{cc} from Equation A.4) with statewide PET collection from commercial sources $Q_s^{cc,\ com}$. The latter is estimated by summing over commercial collection from all CCSDs, $Q_i^{cc,\ com}$:

$$Q_s^{cc,\ res} = Q_s^{cc} - \sum_{i\ \in\ s} Q_i^{cc,com} \tag{A.10}$$

Estimating $\boldsymbol{Q_i^{cc,\ com}}$: Curbside programs are also extended to most small to medium sized commercial establishments in a state but data on waste from commercial sources is even more scarce. For this reason, we extrapolate from waste generation and recycling data in California's commercial recycling survey[7], similar approach can be found in Meyer's work[8]. The survey estimates PET waste generated ($q_b^{gen,\ per\ employee}$) and collected ($q_b^{com,\ per\ employee\ in\ CA}$) per employee by business type. Not all states may have the same level of collection from commercial sources as California. To find per employee estimate in another state, we multiply by the CA estimate by the ratio of total PET recycled in state s and California.

$$q_b^{com,\ per\ employee\ in\ s} = \frac{Q_S^r}{Q_{CA}^r} * q_b^{com,\ per\ employee\ in\ CA} \tag{A.11}$$

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[7] CalRecycle. Generator-Based Characterization of Commercial Sector Disposal and Diversion in California; 2014; p 20.

[8] Meyer, D. E.; Li, M.; Ingwersen, W. W. Analyzing Economy-Scale Solid Waste Generation Using the United States Environmentally-Extended Input-Output Model. Resources, Conservation and Recycling 2020, 157, 104795. https://doi.org/10.1016/j.resconrec.2020.104795.

We then multiply the per employee quantity by $e_{b,i}$, the number of employees in business group b in CCSD i, retrieved from the Annual Census Survey and sum over all businesses in a CCSD to estimate total PET collected in CCSD i, $Q_i^{cc,com}$.

$$Q_i^{cc,\ com} = \sum_b e_{bi} * q_b^{com,\ per\ employee\ in\ s} \tag{A.12}$$

Note: We assume that negligible PET bottles are supplied from drop-off recycling collection in states. Very little data exists on drop-off recycling programs, but since majority of the US population already has access to curbside recycling programs and all of them accept PET bottles, it only contributes a small fraction to the whole (The Recycling Partnership, 2020).

Estimating curbside collection cost

Residential: Similar to Bohm et al.[9], we model (log of) total residential curbside collection cost for a CCSD i as a function of the (log of) collection quantity (all residential waste).

$$\ln\left(Collection\ Cost_i\right) = 5.5025 + 0.95352 * \ln\left(R_i^{cc,res}\frac{Q_i^{cc,\ res}}{\%PET}\right) \tag{A.13}$$

Our estimates of unit collection costs range from 177\$/t (1kt annual) to 142\$/t (100kt annual). EPA reported range[10] is between \$90/t-\$278/t (last updated in 2016).

Commercial: Volumes of waste generated at commercial establishments are larger than residential households. Since collection trucks need to stop fewer times to collect the same quantity, we assume that commercial collection is cheaper. Therefore, we assign all commercially collected PET as having a collection cost equal to the minimum residential collection cost in a state.

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[9] Bohm, R. A.; Folz, D. H.; Kinnaman, T. C.; Podolsky, M. J. The Costs of Municipal Waste and Recycling Programs. Resources, Conservation and Recycling 2010, 54 (11), 864–871. https://doi.org/10.1016/j.resconrec.2010.01.005.

[10] US EPA. Collection Costs, Improving Recycling's Economic Profile: Tools for Local Government Recycling Programs. https://archive.epa.gov/wastes/conserve/tools/localgov/web/html/collection.html (accessed 2022-10-07).

A.3.2 Sorting

Materials Recovery Facilities (MRFs) sort PET from mixed recyclables collected using curbside programs. Therefore, the sorting step is modeled as individual MRF (Materials Recovery Facility) units placed by facility location and transport distance optimization (Box 3 in Figure A-1) in each state, as described below.

In a state, let there be $\{G_i\}$ generating units (CCSDs); set size g. Each G_i has its annual own waste collection amount, Q_i^{cc}. Let there be $\{F_j\}$ potential MRFs (Materials Recovery Facilities) with predetermined locations (county population weighted centroids) each with annual capacity $\{C_{\mathrm{j}}\}$ and binary decision variable y_j indicating the existence of the MRF; set size f. z_{ij} is a binary decision variable that is 1 when Generating Unit G_i has a MRF contract with facility j. Multiple generating units can have a contract with the same MRF but only one MRF may be contracted by a generating unit. Let T_{ij} be the transportation distance of moving the material from when Generating Unit i to facility j. Then, facilities are located at county centroids by the following mixed-integer linear programming solution:

$$minimize \sum_{j=1}^{f}\sum_{i=1}^{g} T_{ij} z_{ij} \tag{A.14}$$

With constraints:

$$u_{min}\ C_j\ y_j < \sum_{i=1}^{g} Q_i^{cc} z_{ij} \leq \ C_j\ y_j \ \ \forall j \tag{A.15}$$

$$\sum_{j=1}^{f} z_{ij} \ \leq 1\ \forall\ i \tag{A.16}$$

$$\sum_{j=1}^{f}\sum_{i=1}^{g} Q_i^{cc} z_{ij} \leq \ s_{min} * \ \sum_{i=1}^{g} Q_i^{cc} \tag{A.17}$$

Over variables:

$$z_{ij} \in \{0,1\} \;\; \forall \; i,j \tag{A.18}$$

$$y_j \in \{0,1\} \;\; \forall \; j \tag{A.19}$$

The constraints in Equations A.15 ensure that capacity minimum ($u_{min} = 60\%$) and maximum (100%) are respected; statewide minimum waste routing factor (s_{min}) in Equation A.17 is used to avoid low-capacity usage while allowing convergence in reasonable time. According to Equation A.16, only one facility is contracted by a generation unit – in reality, multiple facilities may be contracted by a generation unit, specifically in and around large cities but this reasonable simplification makes the optimization tractable. Following MRF location, unit transport cost of recyclables to MRF is calculated by multiplying the average weighted distance travelled to a MRF ($\overline{d_j}$) with a constant road transport cost factor (c^{tra}).

$$\overline{d_j} = \sum_{i=1}^{g} Q_i^{cc} z_{ij} T_{ij} \;\; \forall \; j \mid y_j = 1 \tag{A.20}$$

$$MRF\ Transport\ Cost_{j,\ \ allocated\ to\ PET}^{per\ \ tonne} \;\; = c^{tra} * \; \overline{d_j} \tag{A.21}$$

We assume $c^{tra} = 0.64$ USD/tonne-km, based on known disposal transport cost estimates[11] and density differences[12] in trash and recyclables. With this assumption, it is implied that cost to transport one tonne of PET as part of mixed recyclables is the same as the cost to transport one tonne of mixed recyclables. We use Gurobi Optimization's python extension module (gurobipy) for the distance minimization step; we define the model as specified in Equations A.14-A.21 and use the deterministic concurrent method (setting method = 4) to perform the mixed integer optimization. We set other parameters Optimality Tolerance = 0.001, MIPGap = 0.05 and TimeLimit = 300s after assessing trade-offs between accuracy

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[11] A. Goldsmith Resources. Cost Projections for Transfer, Haul, and Disposal of Municipal Solid Waste; 2014;

[12] US EPA. US EPA. Volume-to-Weight Conversion Factors; 2016

and time. MRF economics (box 4 in Figure A-1) is modeled as described in previous work[13] and the output of this module is the quantity of sorted and baled PET, $Q_j^{cc:b}$ for each MRF located by the distance optimization step and the allocated sorting cost for one tonne of PET, $Sorting\ Cost_{j,\ \ allocated\ to\ PET}^{per\ tonne}$ (Equation A.23).

Net processing fees for MRFs is calculated using output bale prices as shown in Table A.4.

Since a MRF deals with materials other than PET or plastics, sorting cost is allocated to PET on a mass basis following the equation below:

$$Sorting\ Cost_{allocated\ to\ PET}^{facility\ j} = \frac{M_{PET}}{M_{total}} * Cost_{non-plastics}^{facility\ j} + \frac{M_{PET}}{M_{plastics}} Cost_{plastics}^{facility\ j} \tag{A.22}$$

$$Sorting\ Cost_{j,\ \ allocated\ to\ PET}^{per\ tonne} = \frac{Cost_{allocated\ to\ PET}^{facility\ j}}{M_{PET}^{output}} \tag{A.23}$$

Table A.4: MRF bale prices from RecyclingMarkets.net (2021)

Output	Price (USD/tonne)
Plastic Film	14
Corrugated Cardboard	66
Contamination	0
Glass	0
Mixed Paper	32
Ferrous Metal	100
Non-ferrous Metal	984
PET Bottles	300
HDPE Containers (Natural)	1250
HDPE Containers (Colored)	340

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[13] Basuhi, R.; Moore, E.; Gregory, J.; Kirchain, R.; Gesing, A.; Olivetti, E. A. Environmental and Economic Implications of Us Postconsumer Plastic Waste Management. Resources, Conservation and Recycling 2021, 167, 105391.

Table A.5: MRF parameters for the sorting model (details on each column can be found in previous work.)

Equipment	**Throughput (TPH)**	**Electricity (kW)**	**Labour**	**Investment (USD)**	**Footprint (m^2)**	**Labour Rate**	**r**	**q**	**Required Purity**	**Material Sorted**
Vacuum	10	5	2	150000	45	20	0.97	1	0	Plastic Films
Disc Screen	45	8.5	2	175000	15	20	0.8	0.95	0	Corrugated Cardboard
Scalping Screen	7	10	1	280000	15	20	0.97	0.95	0	Fines
Glass Breaking Screen	9	30	1	220000	10	20	0.97	0.95	0	Glass
Disc Screen	21	5.5	6	400000	15	20	0.91	0.95	0	Mixed Paper
Magnet	2	4	1	35000	1	20	0.98	0.98	0	Ferrous
Eddy Current	10	9	1	128000	8	20	0.97	0.98	0	NonFerrous
Receiving and Feed	10	16	1	290000	4.4	20	1	1	0	All
Baler	51	63	1	550000	140	20	1	1	0	None
NIR-PET	10	13	1	350000	12	20	0.95	0.9	0.8	PET
NIR-HDPE	10	40	1	350000	12	20	0.95	0.9	0.8	HDPE

A.3.3 Recycling

PET bales from MRFs and deposit locations are assumed to be sent to known, existing plastic reprocessors or reclaimers across the U.S. (data extracted from map[14] assuming mean capacity from the range in the legend for each size); in the expansion scenarios, a flexible

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[14] Closed Loop Partners. A Data Visualization Tool Identifying Opportunities to Recapture Plastic in the US & Canada. Closed Loop Partners. https://www.closedlooppartners.com/research/us-and-canada-recycling-infrastructure-and-plastic-waste-map/ (accessed 2022-08-23).

scale up parameter is used to modify the capacities of existing recycling facilities (simulating facility expansion) but their locations remain unchanged. A distance optimization model is used to connect MRFs (population-weighted county centroid) and deposit states (population-weighted state centroid) to the recycling facilities of (flexible) capacities (Box 5 in Figure A-1). In the 2018 simulation, the capacities are fixed to reflect the current capacities. Differentiation between types of plastic material is made to allow for matching reclaimers that take in food-grade PET or HDPE and reclaimers are thus sorted by material they accept. Reclaimer matching to plastic material (resin format and grade by source) is done as follows:

Material m (Resin – Format – Source)	Reclaimer Characteristics
PET Bottles Curbside	PET == 1
HDPE Containers Curbside	HDPE == 1
PET Bottles Deposit	PET == 1, Food-Grade == 1

For each material m, a set of eligible reclaimers $\{R_k\}^m$ are identified and the volume of baled and sorted material from specified sources (MRFs or deposit centers) is compiled $\{Q_s^{dc:b},\ Q_j^{cc:b}\}^m$. Admissible set of values for indices s, j and k are dependent on material m – for example PET from deposit sources can be sent to food-grade or non-food grade reclaimers, PET from curbside can only be sent to non-food reclaimers while HDPE can only be sent if the reclaimers process HDPE, etc. Since the method of optimization is independent of type and source of material, index j is used broadly mean any facility or entity that supplies baled and sorted PET bottles (could be MRF j or deposit state s) to simplify notation. Transportation distance (and proportional cost) is denoted by T_{jk}^m along arcs A_{jk}^m – arcs take continuous values between $(0, 1)$ to denote the fraction of material m from facility j going to reclaimer k with the constraint that the sum of arc values from one facility must sum to less than 1 to conserve mass (Equation A.27). A facility can have reprocessing contracts with multiple reclaimers in this optimization setup where transportation distance is minimized. Capacities of each reclaimer is characterized as C_k which can be scaled up in the non-baseline scenarios.

Optimization is performed similar to the sorting model using the Gurobi optimizer[15] but the parameter OptimalityTol is increased to 0.01 to accommodate the larger nationwide model.

$$minimize \sum_{m}\sum_{k=1}^{b[m]}\sum_{l=1}^{r[m]} T_{kl}^{m} * A_{kl}^{m} \tag{A.24}$$

With constraints:

$$u_{min}\ s_{up} C_{l}\ < \sum_{m}\sum_{k=1}^{b[m]} A_{kl}^{m} B_{k} \leq s_{up}\ C_{l}\ \forall\ l \tag{A.25}$$

$$\sum_{k=1}^{b[m]} A_{kl}^{m} B_{k}\ \geq\ M_{min} \sum_{k}^{b[m]} B_{k} \forall\ m \tag{A.26}$$

$$\sum_{l=1}^{r[m]} A_{kl}^{m}\ \leq 1 \forall\ m,\ k \tag{A.27}$$

Over variables:

$$A_{kl}^{m}\ \in [0,1] \forall\ m, k, l \tag{A.28}$$

For u_{min} and s_{up} are minimum utilization factor and flexible scale up factor respectively and Equation A.25 maintains capacity constraints. M_{min}, a minimum total (nationwide) material routing factor is used to avoid low-capacity usage while keeping the optimization tractable in reasonable time (Equation A.26). Once all system-optimized arcs are established, reclaimers or recycling facilities are set up with the optimal connections supplying baled recyclables and the cost of reprocessing is estimated for each recycler. Recycling

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[15] Gurobi Optimization, L. Gurobi Optimization, LLC. Gurobi optimizer reference manual, version 9.0. URL: https://www. gurobi. com/documentation/9.0/refman/index. html.

economics is modeled (box 6 in Figure A-1) as described in previous work[16], and a quality grade of 3 (flakes) is used. Food-grade premium of 220 USD/tonne is estimated and applied to food-grade end use after supply-demand clearing (described in Appendix B).

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[16] Basuhi, R.; Moore, E.; Gregory, J.; Kirchain, R.; Gesing, A.; Olivetti, E. A. Environmental and Economic Implications of Us Postconsumer Plastic Waste Management. Resources, Conservation and Recycling 2021, 167, 105391.

Appendix B

Appendix to Chapter 2 - Demand

B.1 Building Demand Curves

A demand curve statistically models the relationship between r-PET price and demand. At each price point, the demand curve describes how much recycled material all PET product manufacturers collectively would be willing to purchase (Figure B-1).

B.2 Demand for r-PET use in bottles

We model the demand for recycled PET by bottle manufacturers as a derived demand composed of a "substitution curve" and a "product demand curve". Since the willingness of manufacturers to pay for r-PET depends strongly on the price of virgin PET (v-PET), we describe the decision of manufacturers to substitute from r-PET to v-PET via a substitution curve. A substitution curve is the relationship between the price ratio of r-PET and v-PET vs the percentage recycled content used in the product (in this case PET bottles). As the price ratio (r-PET/v-PET) decreases, manufacturers desire a larger proportion of recycled content in their products. Since virgin and recycled material are not perfectly substitutable (due to differences in quality), cheaper recycled material prices are needed to incentivize higher substitution.

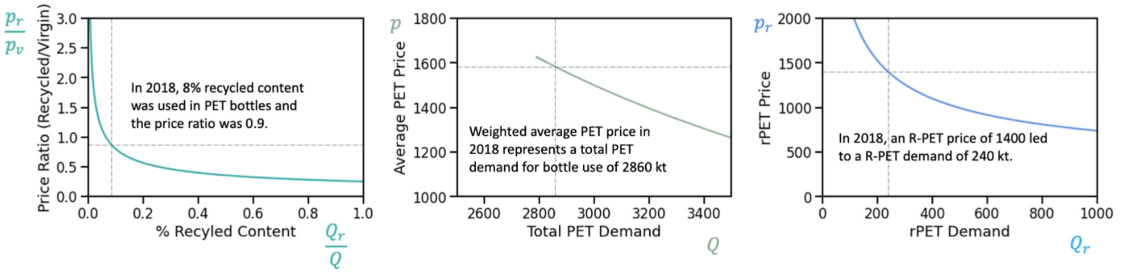

Figure B-1: Modeling demand by (a) substitution curve that establishes the relationship between the proportion of PET content in a bottle that is from secondary sources and the difference in prices between recycled and virgin PET. As the price ratio between recycled and virgin PET increases, manufacturers substitute to using more virgin material. (b) product demand curve that establishes total PET demand (from primary and secondary sources) as a function of weighted average PET price. As PET prices decrease, the demand for PET increases (due to substitution from other materials e.g. aluminum or as a result of increased sales) (c) the recycled PET demand curve is derived from the product and the substitution curve; it measures how recycled PET demand changes as a function of recycled PET price (given a particular level of virgin PET price)

$$\frac{Q_r}{Q_v} = rc * p_{ratio} * \left(\frac{p_r}{p_v}\right)^{\gamma} \tag{B.1}$$

Where rc is the current proportion of recycled content used in bottles, p_{ratio} is the current ratio of recycled PET price to virgin PET price. γ is the price elasticity of substitution, p_r is the price of virgin PET, p_v is the price of recycled PET, Q_r is the recycled PET demand and Q_v is the virgin PET demand.

A product demand curve is the relationship between the demand for PET and the price of PET (average of virgin and recycled PET prices, weighted by the proportion of recycled content in packaging). This curve tells us how much PET (either recycled or virgin) manufacturers want to buy to produce bottles. As prices for virgin and recycled PET increase, manufacturers may produce fewer bottles or switch PET bottles for other packaging such as aluminum cans.

$$Q = \ Q_r + Q_v = \ Q_o * \left(\frac{p}{p_o}\right)^{\varepsilon} \tag{B.2}$$

$$p = \frac{Q_r * p_r + Q_v * p_v}{Q_r + Q_v} \tag{B.3}$$

While the product curve tells us how much PET manufacturers are willing to buy at a given price, the substitution curve tells us the relative amounts of recycled and virgin material they are willing to purchase at a given price difference. We can transform the two curves to calculate a demand curve for recycled PET which tells us how much recycled PET manufacturers are willing to buy at a given r-PET price.

$$Q_r = D^b(p_r) \tag{B.4}$$

Price elasticities for demand are notoriously hard to estimate and infer. Many papers therefore select an elasticity value based on qualitative reasoning and conduct a sensitivity analysis. We use a value of -2 for γ and -0.9 for ε (Taken from Netherlands Bureau for Economic Policy Analysis[1] that calculate the price elasticity for plastics for substitution from virgin to recycled plastics). The substitution elasticity is higher because r-PET and v-PET are assumed to be relatively close substitutes. We also conduct sensitivity analysis by varying these values for elasticity. We follow a similar procedure to build a demand curve for r-PET demand by non-bottle end use sectors $D^{nb}(p_r)$. The following table details all the demand parameters set for the various expansion scenarios considered in this work.

Table B.1: Demand parameters used for scenarios.

Parameter	2018 Bottle	2018 Non-Bottle	2025 Bottle	2025 Non-Bottle	2030 Bottle	2030 Non-Bottle	2035 Bottle	2035 Non-Bottle
q	0.08	0.12	0.25	0	0.5	0	.7	0
Q_o	3010	5300	3070	5300	3240	5300	3400	5300
γ	-2	-1	-2	-1	-2	-1	-2	-1
ε	-0.9	-0.05	-0.9	-0.05	-0.9	-0.05	-0.9	-0.05
p_v	1600	1600	1300	1300	1300	1300	1300	1300
p_{ratio}	0.876	0.741	1	0.741	1	0.741	1	0.741

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[1] Mannaerts, H. J. B. M. STREAM: Substance Throughput Related to Economic Activity Model: A Partial Equilibrium Model for Material Flows in the Economy; Onderzoeksmemorandum / Centraal Planbureau; Netherlands Bureau for Economic Policy Analysis: The Hague, 2000.

In 2018, average virgin PET price was ~\$1565/tonne (1385\$/t in January; 1650\$/t in July; 1660\$/t in December) while average recycled PET price was \$1355/tonne (\$1125/t in January; \$1450/t in July and \$1492/t in December). We rounded the average prices given the high range of variation in prices. Data on price is from ICI[2] and data on PET quantities and recycled content are from EPA[3] and NAPCOR[4].

B.3 Supply-Demand Interaction

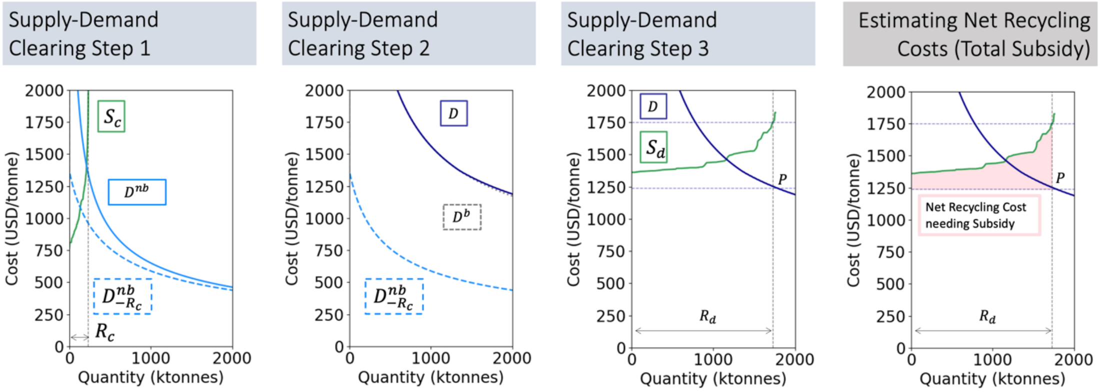

Figure B-2: Visualizing supply demand interaction steps. In step 1, non-bottle demand D^{nb} is matched against curbside supply S_c, and in step 2 excess demand not satisfied by curbside collection $D^{nb}_{(-R_c)}$ is added to bottle demand D^b to created combined demand D which is matched against deposit supply S_d in step 3. Right-most figure illustrates how net recycling cost needing subsidy can be evaluated as the area under the curve above the demand price P.

Demand for r-PET comes from two sources: bottle manufacturers who consume only higher-grade material as pellets and non-bottle end uses (that can use lower quality r-PET). The two sources of supply we consider is deposit material (higher quality) and curbside collected material (lower quality due to greater contamination). We assume bottle manufacturers can only consume deposit material while non-bottle manufacturers are willing to purchase both deposit and curbside collected material. While there is some curbside collected r-PET that

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[2] ICIS Chemical Business. United States Chemical Profile: PET; 2019

[3] US EPA. Advancing Sustainable Materials Management: 2018 Tables and Figures; 2020; p 84.

[4] NAPCOR. Postconsumer PET Recycling Activity in 2018; 2019. https://napcor.com/wp-content/uploads/2021/07/Postconsumer-PET-Recycling-Activity-in-2018.pdf (accessed 2022-07-17).

meets FDA approval for use in food grade bottles, these are typically lower quantities.

We assume that all the curbside recycled material is consumed by non-bottle end uses and this partially reduces their demand. This shifts the demand curve for non-bottle end uses by R_c (Step 1, Figure B-2). The demand curve for food-grade RPET (satisfied by deposit supply) is the sum of the demand by bottle manufacturers ($D^b\left(p_r\right)$) and demand by fiber manufacturers that is not satisfied by curbside material (Step 2, Figure B-2).

$$D(p_r) = D^b\left(p_r\right) + \max(0,\ D^{nb}\left(p_r\right) - R_c \tag{B.5}$$

Here R_c is the total amount of recycled PET that was collected curbside. The price at which the market for food grade r-PET clears (i.e., the price at which r-PET is sold) is the price the end-users are willing to pay when the quantity supplied is equal to 95% of the total deposit material recycled for each expansion scenario (Equation B.5; Step 3, Figure B-2). In other words, the price is determined by the price at which 95% of the recycled supply is consumed (95% excludes the small number of unusually high-cost suppliers determined by the model that are unlikely to exist in real life).

$$P = D^{-1}(0.95 *\ R_d) \tag{B.6}$$

We do not calculate price based on demand-supply curve matching (finding the value for which demand equals supply). This is because there is typically very low (approximately 0) supply elasticity in secondary markets. Waste collection and sorting occurs for many materials together and the decision to collect PET bottles does not depend on the price of r-PET. Moreover, recycling is often mandated and supported by the government. In this scenario, the sellers are willing to sell for any price because the cost has been incurred already and the loss is subsidized. The price is, therefore, determined by how much the buyers are willing to pay while the quantity is determined by how much suppliers can collect and recycle.

B.4 Estimating net recycling costs (total subsidy)

Finally, we use information on prices and quantities of recycled material to calculate net recycling costs of the full system, or the total subsidy needed to keep it operational. In the case that the price of PET (P) is lower than the cost of recycling (height of the bar on the supply curve, $C_{s,k}^{dc:b:r}$ for deposit supply and $C_{j,k}^{cc:b:r}$ for curbside supply) for a particular entity on the supply curve, the municipality or state will need to subsidize collection and recycling for those entities The efficient subsidy needed is the sum of the subsidy needed for each entity to ensure they run without loss, and is defined as net recycling cost for deposit ($NRC^{deposit}$) and curbside ($NRC^{curbside}$):

$$NRC^{deposit} = \sum_{(s,\ k)} \max\left(0, \left(C_{s,k}^{dc:b:r} - P\right) *\ Q_{s,k}^{dc:b:r}\ \right) \tag{B.7}$$

$$NRC^{curbside} = \sum_{(j,k)} \max\left(0, \left(C_{j,k}^{cc:b:r} - P\right) *\ Q_{j,k}^{cc:b:r}\ \right) \tag{B.8}$$

We construct a unit cost metric per tonne of total R-PET recycled and used,

$$NRC_{per\ tonne}^{PET\ recycling} = \frac{Net\ Recycling\ Cost^{Deposit}\ +\ Net\ Recycling\ Cost^{Curbside}}{R_d +\ R_c} \tag{B.9}$$

Subsidies are common in deposit return schemes and typically take the form of handling fees that cover the costs of redemption centers that are not met by the sale of recycled material. Processing fees and municipal responsibility of waste collection for recycling is also an indirect subsidy for curbside streams.

B.5 Estimating extended producer responsibility (EPR) fees

New packaging EPR laws across the country[5] are expected to apply EPR fees to PET bottles to transfer the cost of recycling to the producer (and indirectly, in partial or whole, to the consumer). To put the number in perspective, we also calculate the EPR fees in \$ per unit packaging:

$$EPR\ Fees^{PET\ Bottles} = NRC_{per\ tonne}^{PET\ recycling} * \ w_{av} * \ 10^{-64} \tag{B.10}$$

To convert per unit bottle costs to cost per ton of PET, we use an estimated average weight of the bottle, $w_{av} = 31.7$g. According to Becerril-Arreola and Bucklin[6], the average weight of a water container is 27.56g, the average weight of a soda container is 27.38g and the average weight of a juice container is 53g. A report analyzing a bottle bill proposal for Minnesota used a bottle weight of 31.7g[7], which is the value we picked.

For the 2035 Scenario case A (70% RC, 1300 USD/tonne V-PET price) we estimate that $NRC_{per\ tonne}^{PET\ recycling}$ is \$290/t, which leads to an EPR fees of 0.9 cents/bottle.

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[5] Sustainable Packaging Coalition. EPR Guide to All Policies https://epr.sustainablepackaging.org (accessed 2022-12-18)

[6] Becerril-Arreola, R.; Bucklin, R. E. Beverage Bottle Capacity, Packaging Efficiency, and the Potential for Plastic Waste Reduction. Sci Rep 2021, 11 (1), 3542. https://doi.org/10.1038/s41598-021-82983-x.

[7] Reclay StewardEdge. Reclay StewardEdge. Recycling Refund System Cost-Benefit Analysis; 2014

B.6 Additional Results

B.6.1 Mass flows for simulated future scenarios

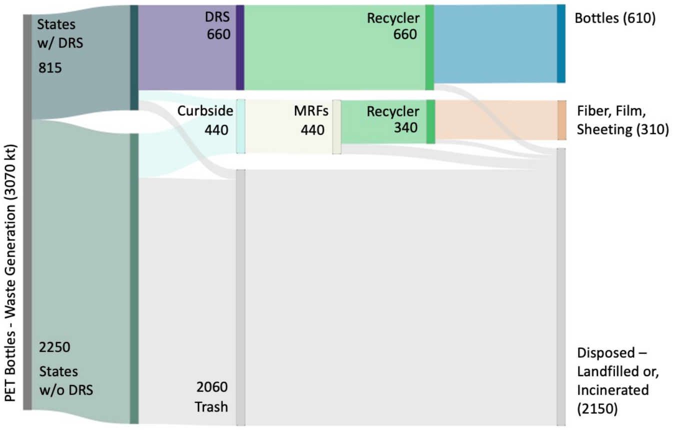

Figure B-3: Sankey diagram showing mass flows for future scenario in 2025 when deposit return system in existing states is maximized by improving deposit fees to 10c and increasing coverage rate.

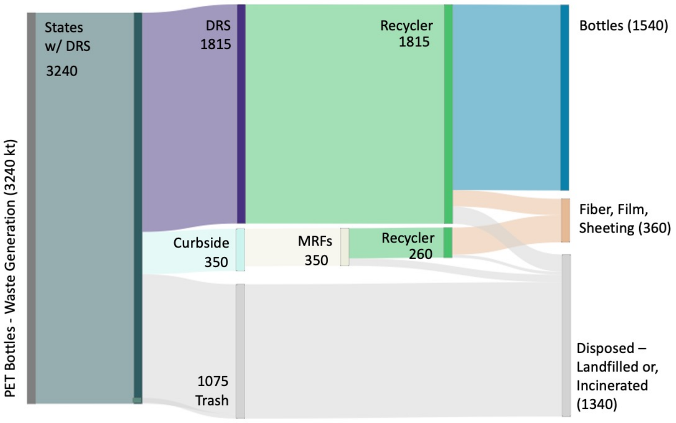

Figure B-4: Sankey diagram showing mass flows for future scenario in 2030 when deposit return system is expanded beyond the existing states to all 48 contiguous states at 5c/unit deposit fee.

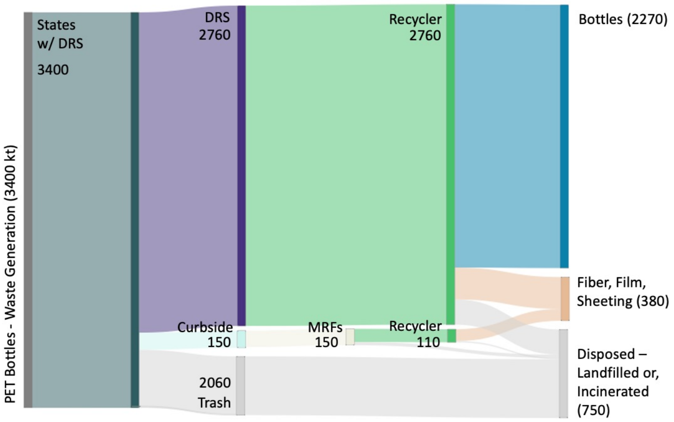

Figure B-5: Sankey diagram showing mass flows for future scenario in 2035 “national deposit system” when deposit return system is expanded to all 48 states at 10c deposit fee with maximum coverage rates.

B.6.2 Variation of R-PET demand curve with V-PET prices

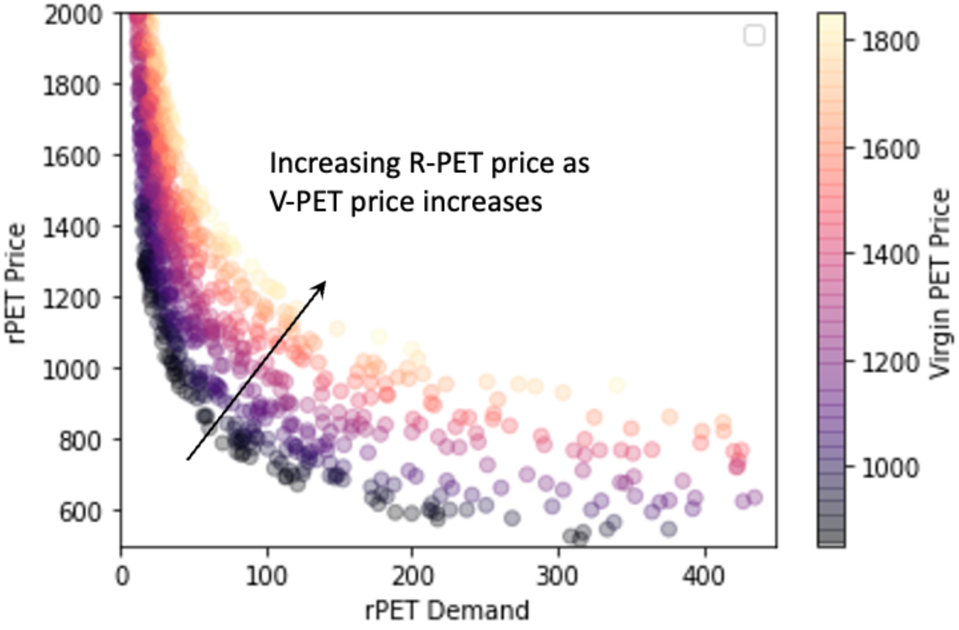

Figure B-6: R-PET demand curves visualized as a function of V-PET price.

B.6.3 Impact of high non-bottle demand on net system costs

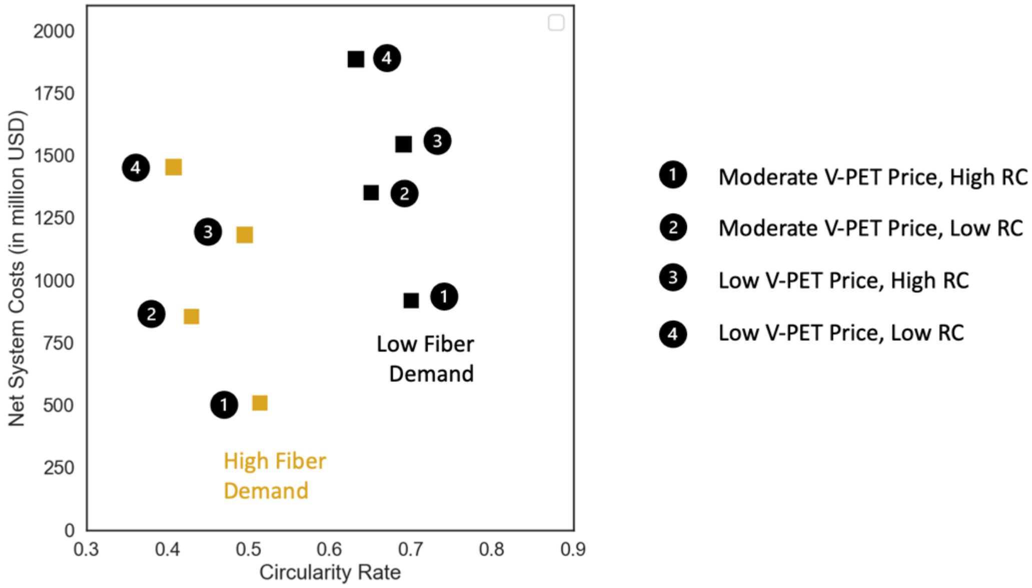

Figure B-7: Non-bottle demand can lower system costs but decrease circularity rates. Cases 1,2,3 and 4 same as main Figure 5.

B.6.4 mpact of high non-bottle demand on R-PET price and net recycling fees

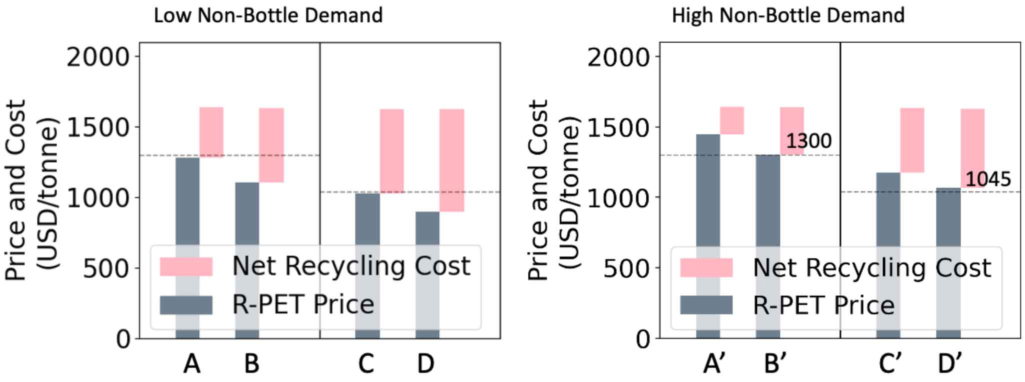

Figure B-8: Comparing net recycling costs across cases A, B, C, D (low non-bottle demand) and A', B', C' and D' (high non-bottle demand) – cases are illustrated in Figure 3c of the main article.

B.6.5 Unredeemed deposits for 2018, 2025, 2030, and 2035 scenarios

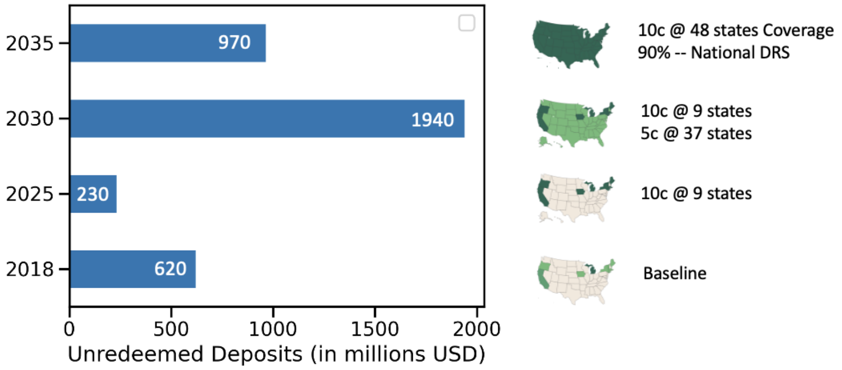

Figure B-9: Unredeemed deposits for scenarios 2018, 2025, 2030 and 2035

Unredeemed deposits are fees collected from consumers who failed to return and redeem the deposit fees – and handling and use of unredeemed deposits varies widely by state. However, since the unredeemed amount decreases as the program becomes more effective (as 10c scenarios have lower unredeemed revenue than 5c, also see Figure B-9), it is a strange revenue incentive that can hinder improving return rates by stifling convenience or expansion of return center networks. Policy design should appropriately link these funds to the externalities of limited recycling to ensure that different actors in the value chain do not have misaligned objectives.

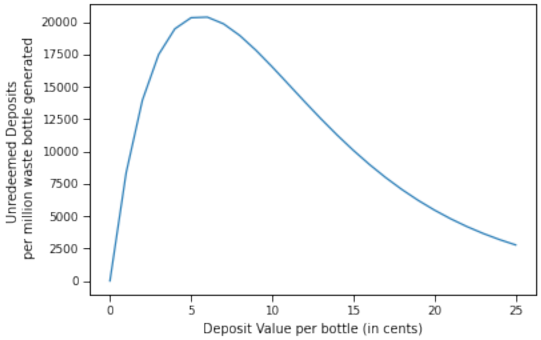

Figure B-10: Unredeemed deposits per million waste bottles generated as a function of deposit value of each bottle in cents

Appendix C

Appendix to Chapter 3 - MFA

C.1 US Flexible Plastic Packaging Waste Flows

C.1.1 Scope and Terminology

In literature and this material flow mapping exercise, “plastic film”, “flexible plastic packaging” and “flexible film” are used interchangeably. The focus is only on thermoplastics, even when plastics are not explicitly mentioned. Within flexible plastic film uses, a differentiation between packaging and non-packaging is made. Even though single-use packaging flows comprise most of the flexible film waste and will be the focus for mitigation strategies and policy impacts, applications like agriculture are also included in the material flow analysis as many recycling estimates include them. Within packaging, both primary packaging (packaging a product) as well as secondary (unitizing separatable products) and tertiary packaging (grouping secondary or primary packaging for easy transport) is included. We do not include pre-consumer scrap from industrial sources as they are usually internally recycled. Geographically, the scope is once again limited to the United States. However, due to the lack of granular data relating to US film waste volumes, datapoints from Canada, UK, and EU states are sometimes used to extrapolate (with uncertainty) to the US.

Flexible packaging is common industry and everyday terminology, defined[1] as "capable of being flexed or bent, such that they are pliant…" and plastic films (including those used as packaging) are also typically characterized by their thin-ness and flexibility. The criteria for thin-ness may differ but are typically in um to mm range, and I also include flexible "sheets" that have a similar form factor. Rigid sheets, on the other hand, are excluded. Film manufacturing can employ any of the following processes: blown-film molding and casting (monolayer films), co-extrusion and lamination (multi-layer films). From a material standpoint, I limit the discussion to films that contain a majority fraction of PE, PP or PET.

C.1.2 Data Collection

Data for waste generation and recovery is estimated and triangulated from several sources. Broadly, reports and estimates are along generator/source of waste, type/application, resin types. Below, I list out existing estimates with tabulated data, describe data sources used, and discuss how the data will be used in our material flow mapping.

Estimates by generator source:

Commercial Generators: The state of California surveys commercial waste (CalRecycle, 2015[2]) and reports plastic film volumes generated by commercial, institutional, and industrial waste generators as well as 5 product types. The product types do not match our designated product types but are nonetheless useful in informing some of the flows. I employ spatial scale-up and extrapolate from waste generation data in the survey to estimate national waste volumes. The CalRecycle survey estimates waste generated per employee by business type at the 3-digit NAICS level arranged into 17 business groups, with 66 materials studied. The waste generated per employee is multiplied by the number of employees in each business group (data from US Census County Business Patterns 2016[3]). A summary

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[1] Niaounakis, M. Recycling of Flexible Plastic Packaging; 2019.

[2] CalRecycle. Generator-Based Characterization of Commercial Sector Disposal and Diversion in California; 2014

[3] https://www.census.gov/data/datasets/2016/econ/cbp/2016-cbp.html

Table C.1: Commercial Film Generated in the US (in kt), estimated by using data from CalRecycle 2015

Commercial Generator Category	**Volume (kt)**
C&I Retail	672
C&I Services	3010
C&I Hospitality	2487
C&I Manufacturing	2023

of our national estimate for 2016 is given below. The California survey is unique in that it includes estimates for trash (and recycling) bags generated as well.

RSE's film recycling investment report[4] estimates waste generation of PE and Other film (multilayer/non-PE films) from commercial sources as shown in Table C.2. However, no methodology or sources for generation details are noted. The report refers to ACC's national film and bag recycling 2014 reports for estimates of recovery (not shown below) for which updated, recent accounts also exist.

Table C.2: Commercial Film Generated in the US (in kt), estimated by RSE (2012)

Resin	Amount (kt)
PE film	1881
Other film (non-PE/multimaterial)	566

Residential Generators - Households: There is no clear track-able statistic for how much flexible film waste is generated by households nationally as it varies significantly across locations and over time. The Recycling Partnership's State of Curbside Recycling that estimated national residential waste quantities for several recyclables does not include flexible films as they are not "recyclable"; therefore, we use several sources to corroborate per household generation rates, and scale it up to all of US. Several city/community/statewide waste audits can be used to estimate household generation, with further breakdown by product categories summarized below in Table C.3.

According to The Recycling Partnership[5], only 50 local governments consider flexible films suitable for curbside recycling, covering approximately 1% of US households. Since most

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[4] RSE. Film Recycling Investment Report; Closed Loop Partners, 2014

[5] The Recycling Partnership. Addressing the Challenge of Film and Flexible Packaging Data; The Recycling Partnership, 2021. https://recyclingpartnership.org/wp-content/uploads/dlm$_{u}ploads$/2021/04/$FF_{W}hitepaper_{F}inal.pdf$

Table C.3: Residential Flexible Waste Generation Audit, multiple cities/states – averaged to per household (hh)

Waste Audit	**Generation (kg/hh)**	**Categories**
Portland Metro Household, 2019	34 kg/hh, 25 kg/hh (without trash bags)	PE Shopping Bags PE Other Household Bags PE Product Overwrap PE Shipping/Transport Film PE Other Film & Flexibles Plastic Pouches PE Garbage Bags Contaminated PE Film & Flexibles Other Film & Flexibles
Red Wing, Minnesota, 2019	19 kg/hh, 13 kg/hh (without trash bags)	Trash Bags Retail Carry Out Bags E-commerce Film Mailers Other PE Bags & Wraps Non-PE Film & Flexibles
Solid Waste Authority of Central Ohio Household Material Generation, 2019	31 kg/hh	Plastic Bags Other Film - Recyclable
Cary, North Carolina, 2019	50 kg/hh	Recyclable film Non-recyclable film
California Statewide, 2018	44.5 kg/hh, 31 kg/hh (without trash bags)	Grocery & Other Merchandise Bags Non-Bag Film Film Products Flexible Plastic Pouches Trash Bags Other Film

communities do not consider films appropriate for curbside recycling, it is difficult to estimate how much of the generated films end up in recycling streams. Moreover, as films are considered contamination by MRFs serving such communities, it is likely that a lot of films put into recycling streams by residents are sent to MRFs to only be sorted into the residual stream sent to trash or incineration. The survey in Cary, North Carolina reports that residents separate out 1.1 kg/hh of recyclable and 0.7 kg/hh of non-recyclable flexible films for recycling. Using retail center audits, The Recycling Partnership estimates that 83 kt of flexible packaging is returned to retail sites (store drop-off) across the US, averaging to approximately 0.7 kg/hh/year. More Recycling[6] estimate the quantity of MRF curbside

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[6] More Recycling. 2018 National Post-Consumer Plastic Bag and Film Report; 2020

film sorted for value to be 1.7 kt in 2018 but more recent reports lump MRF curbside films with “Other films” and estimates are unclear.

RSE’s film recycling investment report also estimates residential generation of flexible for 2012 but it is unclear what method is used and no per household numbers are reported.

Table C.4: Residential Film Generated in the US (in kt), estimated by RSE (2012)

Resin	**Amount (kt)**	
PE film	1515	
Other film (non-PE/multimaterial)	1508	

Estimates by application type, product type and resin type:

First, the EPA reports the amount of film waste as “bags, sacks and wraps” under “containers and packaging” in municipal solid waste along with resin type breakdown. EPA is a secondary (public) data source that synthesizes information from primary sources - American Chemistry Council’s Resin Review and NAPCOR. According to the EPA, the reported amount of municipal solid waste encompasses residential, commercial, institutional, and industrial (not including process waste) generation sources. The EPA does not include trash bags in its estimate.

Table C.5: ”Bags, Sacks and Wraps” waste (in kt) estimated by EPA (2018)

Resin	Generated	Recycled
HDPE	580	54
L(L)DPE	2520	336
PP	517	–

RSE’s film recycling investment report more thoroughly expands on product type/resin type breakdown of flexible film consumption and waste generation. The report mentions that “the data shown above has been derived from multiple sources, but principally from the Flexible Packaging Association” who only provide resources to members (that must be in the flexible packaging business[7]). RSE’s accounting also excludes trash bags. The report also projects changes in flexible film use in a number of product categories by using 2013-2018 compound annual growth rate values gleaned from numerous industry sources

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[7] https://www.flexpack.org

not referenced explicitly.

Table C.6: Estimates of film waste generaion (in kt) and composition for 2012, compiled from RSE (2014)

Product	Quantity (in kt)		Composition				
	2012	2022	LDPE	HDPE	PP	PET	Others
Retail Bags	315	347	0.33	0.67	0	0	0
Heavy Duty Shipping Bags	774	944	1	0	0	0	0
Box Liners	375	426	0.1	0.9	0	0	0
Storage Bags	277	309	1	0	0	0	0
Roll Wrap	516	543	1	0	0	0	0
Shrink Bundling	392	565	1	0	0	0	0
Stretch Films	425	544	1	0	0	0	0
Pillow Pouches	1015	1178	0.47	0	0.39	0.14	0
Woven bags	76	93	0	0	1	0	0
Wraps	25	30	0	0	0.98	0	0.02
Other bags	246	269	0.87	0	0	0	0.13
Lay Flat Pouches	489	608	0.23	0	0.07	0.62	0.08
Stand-up Pouches	436	646	0.32	0	0.03	0.58	0.07

I could not find national or sub-national estimates of non-packaging flexible film use in the agricultural and construction sector for the US. So, I turned to data from non-US developed nation sources. I only extract information pertaining to flexible film applications and waste generation from these sources since recovery and recycling depend more critically on local attitudes, practices, and regulations and cannot be as easily transferred across contexts.

A report detailing Europe's flexible film industry[8] lists percentage breakdown of flexible film use by applications/products (see Table C.7. While these are different from my chosen product types, the data can be additionally used to inform flows in the MFA by appropriately mapping to the relevant group/fraction of flows. As a percentage value the data is directly used without any scaling.

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[8] Plastics Recyclers Europe. Flexible Films Market in Europe State of Play - Production, Collection and Recycling Data. Eunomia 2022

Table C.7: Percentage breakdown by application/products from Flexible Films in Europe: State of Play Report (2021)

Application/Product	Percentage of Total Use
Food Packaging	23%
Non-food Packaging	41%
Film on Reel	9%
Stretch Film	18%
Shrink Film	14%
Bags & Sacks	22%
Refuse Sacks	5%
Carrier Bags	4%
Heavy Duty Bags	7%
Other Bags & Sacks	6%
Agricultural Films	7%
Buildings & Construction	2%
Others	5%

Data from UK[9] breaks down information on flexible film flows and composition by generator source. Table C.8 below shows volume of film packaging for the UK along with composition for some of the categories. I scale this to the US by using the GDP ratios.

Table C.8: UK PlasticFlow Flexible Plastic Films Placed on Market (in kt), WRAP (2018) [Note these values are for the UK.]

Categories, SubCategories	All packaging	Error	Film	Composition* (in %)		
	(in kt)	%	(in kt)	LDPE**	HDPE	PP
Consumer/Retail Packaging	1532	7	395	35.5	22.5	28
– Grocery Packaging	948	6				
– NonGrocery Packaging	548	16				
C&I Packaging	731	13				
– Retail Back of Store	126	15	123	90	5	5
– Hospitality	196	15	36	63.5	2.5	20
– Manufacturing & Other	409	117				
Agriculture	37	21	31	68		32
Construction & Demolition	62	21	54	100		

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[9] PlasticFlow 2025 - Plastic packaging flow data report. 2018. https://wrap.org.uk/resources/report/plasticflow-2025-plastic-packaging-flow-data-report

Estimates by end-use/end-of-life fate:

Only one source was available to estimate quantity of recycled plastics: Stina's U.S. Post-Consumer Plastic Recycling Data Report[10] (previously National Post-Consumer Plastic Bag & Film Recycling Report by More Recycling, till 2020). This report breaks it down not by resin but film quality characteristics (such color, mixing from source).

Table C.9: US Flexible Plastic Film Waste Recovered (2021) reported by StinaInc, APR

Bale Type	Amount (in kt)
PE Clear Film	199
PE Mixed Color Film	86
PE Retail Bag and Film	70
PE Agricultural Film	120
Other Film*	27
Export	85

Recent reports do not have explicit information on end-uses so older 2018 account is summarized below to identify destinations for recycled film.

Table C.10: US Flexible Plastic Film Waste Recycled by End-Use (2018) reported by ACC, More Recycling

End-Use	Share (%)
Lumber/Decking	46
Film/Sheet	34
Injection Molding	12
Other	8

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[10] https://circularityinaction.com/2020PlasticRecyclingData

C.2 Mapping material flows

Data for flexible film material flow Sankey diagram in 3-6 is listed below.

From	**To**	**Volume**
C&I Services	Trash Bags	1195
C&I Retail	Trash Bags	572
C&I Hospitality	Trash Bags	1043
C&I Manufacturing	Trash Bags	143
Agricultural	Trash Bags	51
Residential	Trash Bags	1220
Trash Bags	Disposal	4224
C&I Services	Retail Bags	128
C&I Retail	Retail Bags	119
C&I Hospitality	Retail Bags	501
C&I Manufacturing	Retail Bags	16
Residential	Retail Bags	735
Retail Bags	Return to Retail	112
Retail Bags	Curbside Residential MRF	135
Retail Bags	Disposal	1252
C&I Services	Heavy Duty Shipping	412
C&I Retail	Heavy Duty Shipping	245
C&I Hospitality	Heavy Duty Shipping	104
C&I Manufacturing	Heavy Duty Shipping	239
Heavy Duty Shipping	Commercial MRF	180
Heavy Duty Shipping	Disposal	820
C&I Services	Box Liners	127
C&I Hospitality	Box Liners	127
Residential	Box Liners	169
C&I Manufacturing	Box Liners	60

Box Liners	Return to Retail	3.8
Box Liners	Curbside Residential MRF	2.9
Box Liners	Disposal	476
Residential	Storage Bags	300
Storage Bags	Return to Retail	9
Storage Bags	Curbside Residential MRF	3
Storage Bags	Disposal	288
C&I Services	Roll Wrap	180
C&I Retail	Roll Wrap	60
C&I Hospitality	Roll Wrap	60
C&I Manufacturing	Roll Wrap	180
Residential	Roll Wrap	145
Roll Wrap	Commercial MRF	86
Roll Wrap	Return to Retail	4.3
Roll Wrap	Curbside Residential MRF	1.5
Roll Wrap	Disposal	533
C&I Retail	Shrink Bundling	116
C&I Manufacturing	Shrink Bundling	116
Residential	Shrink Bundling	243
Shrink Bundling	Commercial MRF	42
Shrink Bundling	Return to Retail	7
Shrink Bundling	Curbside Residential MRF	2.5
Shrink Bundling	Disposal	423
C&I Services	Stretch Film	182
C&I Retail	Stretch Film	110
C&I Hospitality	Stretch Film	170
C&I Manufacturing	Stretch Film	78
Stretch Film	Commercial MRF	180
Stretch Film	Disposal	360

C&I Hospitality	Pillow Pouches	292
Residential	Pillow Pouches	877
Pillow Pouches	Curbside Residential MRF	19
Pillow Pouches	Disposal	1150
C&I Hospitality	Woven Bags	32
Residential	Woven Bags	60
Woven Bags	Curbside Residential MRF	4
Woven Bags	Disposal	88
Residential	Wraps & Others	330
Wraps & Others	Curbside Residential MRF	11
Wraps & Others	Disposal	319
C&I Hospitality	Lay Flat Pouches	61
Residential	Lay Flat Pouches	544
Lay Flat Pouches	Curbside Residential MRF	11.5
Lay Flat Pouches	Disposal	593
C&I Hospitality	Standup Pouches	66
Residential	Standup Pouches	586
Standup Pouches	Curbside Residential MRF	13
Standup Pouches	Disposal	639
Agricultural	Film Products	217
Film Products	Other Collection	60
Film Products	Disposal	157

Appendix D

Appendix to Chapter 3 - TEA

D.1 Advanced Recycling TEA Data Tables

In this appendix, I present data tables that contain information extracted from academic and practitioner publications to survey advanced recycling technology capital and operating costs, readiness, and scalability. Column information for these datatables is clarified below.

Table D.1: Main Worksheet Guide

Column Name	**Description and assumptions**
Index	Reference number as in datatables below.
Link	Link to where reference can be found
Author(s)	Authors of reference
Article	Type of literature reference falls under RESEARCH - unique research article REVIEW - compilation of other articles PERSPECTIVE - compilation of other articles REPORT - gray literature
Year	Year reference was published

Table D.1 continued from previous page

Location	Country in which data originates, if given
LCA	Yes/No if a life cycle assessment is reported
Model	What, if any, software was used for process modeling
Currency	Currency used for cost reporting
Ref. Year Given?	Yes/No if reference year for costs was reported
Ref. Year	Reported year, or assumed as year before publication
Opex in USD$2023	Operating expenses in USD$(July 2023) per metric ton feed, using currency conversion and inflation by CPI (U.S. city average, All items) data from U.S. Bureau of Labor Statistics
Capex in USD$2023	Capital cost estimates in USD$(July 2023) using currency conversion and Chemical Engineering Plant Cost Index from Chemical Engineering Magazine
Feedstock	Reported feedstock as resin (PET, HDPE, LDPE, PP, PS, PVC, Other) if given, and mixed plastic waste (MPW) if not
FPrice	Feed price if reported
L-MSP	Minimum selling price of main product; lower threshold if range given
U-MSP	Upper threshold of main product minimum selling price

Table D.1 continued from previous page

Catalyst	Yes/No if catalysts used in cracking process
L-Capacity	Capacity to process feed in tonnes per day as reported or calculated; lower threshold if range given
U-Capacity	Upper threshold of feed input capacity
Product	Reported products from process
BTX	Yes/No if benzene, toluene, xylene (/styrene) reported as products
Ethylene	Yes/No if ethylene reported as product
Main Prod.	Main product from process either as reported or chosen by yield if no distinction between main and co-products is given
Main Prod. Yield	Main product yield from feed, by mass

Table D.2: OPEX Worksheet Guide

Column Name	**Description and assumptions**
Rx Temp	Cracking reactor operating temperature
Rx Pres.	Cracking reactor operating pressure
Oper. time	Time on stream (when plant is operational) in hours per year
Overall	Reported operating expenses if given, or summation of reported components if not; removed credits or savings accounted to co-products where given and specified in text

Table D.2 continued from previous page

Materials	Cost of materials including feed, catalysts, and reagents
Utilities	Cost of utilities including electricity, energy for heating, water
Labor	Cost of labor as reported, either by operators alone or operators and supervisors
Waste	Cost of waste disposal
Elec. Price	Cost of electricity
Power cons.	Power required for either full plant operations or cracking process alone; both electrical power and energy requirements together, converted using TPY and operating time where necessary
Depreciation	Depreciation payments per tonne feed
Rent	Cost to rent space for facilities, or reported assumptions on how reference estimated it
Overhead	Cost of general overhead, or reported assumptions on how reference estimated it
Maintenance	Cost to maintain process equipment, or reported assumptions on how reference estimated it
Insurance	Cost of insurance coverage for site, or reported assumptions on how reference estimated it; sometimes reported with property tax

Table D.3: CAPEX Worksheet Guide

Column Name	**Description and assumptions**
Size	Actual operating throughput of feed in metric tons per year, either as given or calculated from operating time
Cost Source	Investment cost data primarily obtained from ASP - Aspen software LIT-J - Journal articles LIT-T - Textbooks and handbooks (reference books) LIT-O - Other literature, incl. technical reports from institutions FIRM - Companies, incl. quotes, feasibility studies, webpages NOT GIVEN - No source given
Cost Jargon	The term used in reference to refer to capex estimates
Estimate	Capital expenditure, or capex, required for plant in millions per year, as reported or calculated
Per feed	Capex per tonne feed, either as given or calculated
FCI	Fixed capital investment if reported
Working	Working capital as percentage of FCI or other costs
Lifetime	Plant lifetime
Loan	Loan lifetime, if given
Startup	Time to startup operations, if given
Construction	Time to construct facilities, if given
Depreciation	Length of depreciation period
Depreciation	Type of methodology reported for depreciation
Salvage	Salvage value as reported
Land	Land required in square meters
Land cost	Cost of land, or reported assumptions on how reference estimated it
Proj. contin.	Reported project contingency in costs
Discount rate	Reported rate used for discounted cash flow analysis

Table D.3 continued from previous page

Interest rate	Reported rate used for discounted cash flow analysis
Inflation	Inflation considered, if given
Tax rate	Income tax rate, if given
Proc. Cat.	Process categories of main process sections in plant, as PRE - feed pretreatment, including sorting, washing, grinding CRA - cracking reaction to break C-C bonds DIS - distillation, by towers, including fractionation CUP - chemical upgrading by additional reactions to yield higher-value products PUR - physical separations to increase purity, e.g. by solvent extraction
Commercial Refs.	Commercial plants reported

Table D.4: Products Worksheet Guide

Column Name	**Description and assumptions**
Price	Price for per metric ton reported product
Yield	Yield of reported product from feed

Table D.5: Profitability Worksheet Guide

Column Name	**Description and assumptions**
ROI	Return on investment as reported; or calculated as (capex - net revenue) / capex
IRR	Internal rate of revenue as reported; the discount rate at which net present value becomes 0 (positive) in discounted cash flow analysis

Table D.5 continued from previous page

PBP	Payback period as reported years to recoup investment
Net Revenue	Profits as opex minus revenue from total product sales
Co-products	Co-products from cracking process
Co-products revenue	Additional revenue/credits obtained from products
Other products	Other products for energy recovery
Other products revenue	Additional revenue/credits obtained from products

Table D.6: Impact Worksheet Guide

Column Name	**Description and assumptions**
Energy use	Required energy consumption
Water use	Required water consumption, if given
GHG emissions	Global warming potential as CO_2-equivalents per feed

Reference								in USD$2023			ALL ESTIMATES PER TONNE FEED UNLESS OTHERWISE SPECIFIED										
								$/tonne	$/TPY		$/tonne	$/tonne	$/tonne		TPD	TPD	TPY	hr/yr			tonne prod/tonne
ID	Link	Author(s)	Year	Article Type	Location	Currency	Ref. Year	Opex	Capex	Feedstock	FPrice	L-MSP	U-MSP	Catalyst	L-Capacity	U-Capacity	Size	Oper. time	Product	Main Prod.	Main Prod. Yield
1	DOI	Li, H. et al	2023	Research	USA	USD	2022	697	2,763	HDPE, LDPE, PP		508	1950	☑	13.1	285.7	36,500	8,400	Paraffins, aromatics (BTX), monoalcohols, dialcohols	Monoalcohols	0.49
2-A	DOI	Yadav, G. et al	2023	Research	USA	USD	2016	1,112	1,451	HDPE, LDPE, PET, PP, PS, PVC	600	2180		☑	240.0	500.0	78,840	7,884	Naphtha, aromatics (BTX), olefins, coke/char	Naphtha	0.40
2-B	DOI	Yadav, G. et al	2023	Research	USA	USD	2016	1,738	2,000	HDPE, LDPE, PET, PP, PS, PVC	600	1070		☑	240.0	500.0	78,840	7,884	Naphtha, aromatics (BTX), olefins, coke/char	BTX	0.22
2-C	DOI	Yadav, G. et al	2023	Research	USA	USD	2016	1,544	1,972	HDPE, LDPE, PET, PP, PS, PVC	600	950		☑	240.0	500.0	78,840	7,884	Naphtha, aromatics (BTX), olefins, coke/char	BTX	0.30
2-D	DOI	Yadav, G. et al	2023	Research	USA	USD	2016	1,516	2,186	HDPE, LDPE, PET, PP, PS, PVC	600	850		☑	240.0	500.0	78,840	7,884	Naphtha, aromatics (BTX), olefins, coke/char	Ethylene	0.12
3	DOI	Fivga, A. and Dimitriou, I.	2018	Research	UK	GBP	2013	1,093	2,421	PE, PP, PS		870		☐	2.4	2400.0	701	7,012	Pyrolysis fuel: paraffins, olefins, aromatics, char	Heavy fuel oil	
4	DOI	Kulas, D.G. et al	2023	Research	USA	USD	2019	373	702	HDPE, PP	240	592		☐	240.0	480.0	84,000	8,400	Pyrolysis oil, pyrolysis gas, pyrolysis wax	Pyrolysis oil (naphtha)	0.22
5	DOI	Tan, K. Q. et al	2023	Review										☐							
6	DOI	Sahu, J. N. et al	2013	Research	Malaysia	USD	2011	742	2,027	PE, PP, PS				☑	342.9		10,000	8,400	Gasses, light fuel oil ("gasoline"), heavy fuel oil ("diesel")	Heavy fuel oil (diesel)	0.92
7	DOI	Bora, R.R. et al	2020	Research	USA	USD	2012	408	5,931	PP	200			☐	250.0	288.0	100,000	8,400	Gasses, light & heavy hydrocarbons (olefins & alkanes), aromatics (BTX)	Light HC fuels	0.90
8	DOI	Cappello, V. et al	2022	Research	USA	USD	2016	568	1,262	HDPE	323	667	2200	☑	250.0		91,250	8,760	HQL = "PAO lubricants", naphtha, flue gas	HQL ("PAO lubricant")	0.90
9	DOI	Selvam, E. et al	2023	Research						LDPE				☑			32,000	8,000	Olefins, alkanes C1-C6	API Grade I lubricants	0.88
10	DOI	Almohamadi, H. et al	2021	Research		USD	2020	83	481	PE, PET, PP, PS	0	23	52	☐	1000.0	2000.0	328,500	7,884	Hydrocarbon fuel (olefins, paraffins, aromatics)	Hydrocarbon fuel	0.40
11	DOI	Riedewald, F. et al	2021	Research	Belgium	EUR	2020	106	914	MPW	0			☑	109.6		40,000	7,500	Pyrolysis oil, solids (char), gas	P-oil as Fuel Oil No.6 or Heavy Bunker Oil	0.65
12	DOI	Gracida-Alvarez, U.R. et al	2019	Research	USA	USD	2017	295	1,095	HDPE	22			☐	500.0		175,000	8,400	Olefins (ethylene, propylene), aromatics mixture (BTX, styrene), paraffins (low MWHC mixture = "gasoline", high MWHC mixture = "diesel")	Gasoline (low molecular weight hydrocarbon mixture)	0.54
13	DOI	Gracida-Alvarez, U.R. et al	2019	Research	USA	USD	2017	295	1,114	HDPE	22			☐	500.0		175,000	8,400	Olefins (ethylene, propylene), aromatics mixture (BTX, styrene), paraffins (low MWHC mixture = "gasoline", high MWHC mixture = "diesel")	Gasoline (low molecular weight hydrocarbon mixture)	0.54
14-O	DOI	Larrain, M. et al	2020	Research	Belgium	EUR	2019	446	1,409	LDPE, PP	50			☐	210.0	360.0	120,000	8,000	Gasses, naphtha, wax ("slack wax")	Naphtha	0.40
14-C	DOI	Larrain, M. et al	2020	Research	Belgium	EUR	2019	457	1,174	LDPE, PP	50			☐	210.0	360.0	120,000	8,000	Gasses, naphtha	Naphtha	0.69
15	DOI	Jiang, G. et al	2020	Research	UK	USD	2018	362	893	HDPE, LDPE, PET, PP, PS, PVC	65			☐	12.0	48.0	8,000	8,000	oil/wax (oddly reported from individual plastics instead of MPW)	Oil/wax	0.79
16	DOI	Solis, M. and Silveira, S.	2020	Review										☐							
17-A1	Link	Ocean Recovery Alliance	2015	Report	USA	USD	2014	0	1,067					☐	50.0		15,584	8,059		Synthetic crude oil	0.69
17-A2	Link	Ocean Recovery Alliance	2015	Report	USA	USD	2014	0	989					☐	50.0		18,207	8,059		Synthetic crude oil	0.72
17-MKA	Link	Ocean Recovery Alliance	2015	Report	India	USD	2014	205	1,466		0			☑	10.0		3,308	7,183		Synthetic crude oil	0.63
17-PS1	Link	Ocean Recovery Alliance	2015	Report	India	USD	2014	0	427					☐	3.0		992	7,183		Synthetic crude oil	0.52
17-PS2	Link	Ocean Recovery Alliance	2015	Report	India	USD	2014	0	337					☐	6.0		1,796	7,183		Synthetic crude oil	0.52
17-PS3	Link	Ocean Recovery Alliance	2015	Report	India	USD	2014	0	226					☐	12.0		3,977	7,183		Synthetic crude oil	0.52
17-GR1	Link	Ocean Recovery Alliance	2015	Report	USA	USD	2014	0	1,580					☐	24.0		4,386	7,884		Distillate fuel oiil	0.62
17-GR2	Link	Ocean Recovery Alliance	2015	Report	USA	USD	2014	0	1,316					☐	24.0		6,316	7,884		Distillate fuel oiil	0.62
17-JBI1	Link	Ocean Recovery Alliance	2015	Report	USA	USD	2014	0	2,023					☑	20.0	30.0	3,425	6,570		Distillate fuel oiil	0.86
17-JBI2	Link	Ocean Recovery Alliance	2015	Report	USA	USD	2014	0	1,264					☑	20.0	30.0	8,772	6,570		Distillate fuel oiil	0.86
17-NF1	Link	Ocean Recovery Alliance	2015	Report	USA	USD	2014	0	872					☐	50.0		14,308	8,410		Distillate fuel oiil	0.71
17-NF2	Link	Ocean Recovery Alliance	2015	Report	USA	USD	2014	0	712					☐	50.0		23,346	8,410		Distillate fuel oiil	0.91
17-V1	Link	Ocean Recovery Alliance	2015	Report	USA	USD	2014	0	1,260					☐	60.0		18,702	7,884		Distillate fuel oiil	0.68
17-V2	Link	Ocean Recovery Alliance	2015	Report	USA	USD	2014	0	1,179					☐	60.0		21,152	7,884		Distillate fuel oiil	0.68
18-PE	DOI	Xu, Z. et al	2023	Research	USA	USD	2022	1,193	306	PE	600			☑	30		10,000	8,000	Waxes, fatty acids	Fatty acids for soap	0.86
18-PP	DOI	Xu, Z. et al	2023	Research	USA	USD	2022	1,193	306	PP	600			☑	30		10,000	8,000	Waxes, fatty acids	Fatty acids for soap	0.90
19-PY	DOI	Hernández, B. et al	2023	Research	USA	USD	2021	659	1,770	LDPE	130	660		☑	16.4	164.4	25,000	8,760	Lubricants, jet oil, diesel, gasoline	Lubricants	0.49
19-HC	DOI	Hernández, B. et al	2023	Research	USA	USD	2021	889	2,018	LDPE	130	740		☑	16.4	164.4	25,000	8,760	Diesel, gasoline, electricity	Gasoline	0.73
19-HTL	DOI	Hernández, B. et al	2023	Research	USA	USD	2021	505	716	LDPE	130	880		☐	16.4	164.4	25,000	8,760	Jet oil, diesel, gasoline	Gasoline	0.30
19-HO	DOI	Hernández, B. et al	2023	Research	USA	USD	2021	546	2,289	LDPE	130	1670		☑	16.4	164.4	25,000	8,760	Lubricants, jet oil, diesel, gasoline, LPG	Lubricants	0.19

					CPI to 2023	
		xe.com			U.S. BLS data	Chem engr
		Currency conversion	USD		**Inflation (CPI)**	**CEPCI**
	AUD	4 Oct 2023	0.629	1996	156.90	381.7
	EUR	4 Oct 2023	1.046	2007	207.34	525.4
	GBP	4 Oct 2023	1.207	2011	224.94	585.7
	NLG	4 Oct 2023	0.475	2012	229.59	584.6
				2013	232.96	567.3
				2014	236.74	576.1
				2015	237.02	556.8
		Other conversions		2016	240.01	541.7
		U.S. gal per U.S. barrel	55	2017	245.12	567.5
(crude oil)		U.S. gal per metric ton	307.8	2018	251.11	603.1
		Liter per U.S. barrel	200	2019	255.66	607.5
		Liter per U.S. gallon	3.785	2020	258.81	596.2
		Liter per metric ton	1165	2021	270.97	708.8
				2022	292.66	816
				2023	305.69	798.3
				(July 2023^)		

Reference											pyrolysis				Opex											
							TPD	TPD			°C	atm	hr/yr	TPY	$/tonne	$/tonne	$/tonne	$/tonne	$/tonne	$/kWh	kW	$/tonne	$/yr	$/yr	$/yr	$/yr
ID	Author(s)	Year	Article Type	Location	Currency	Ref. Year	L-Capacity	U-Capacity	Main Prod.	Main PYield	Rx Temp	Rx Pres.	Oper. time	Size	Overall	Materials	Utilities	Labor	Waste	Elec. Price	Power cons.	Depreciation	Rent	Overhead	Maintenance	Insurance
1	Li, H. et al	2023	Research	USA	USD	2022	13.1	285.7	Monoalcohols	0.5	500.0		8,400	36,500	666.8			11							3% of capex	0.7% of capex
2-A	Yadav, G. et al	2023	Research	USA	USD	2016	240.0	500.0	Naphtha	0.4	550.0	3.0	7,884	78,840	873.2	632	39	16	3	0.068				2,363,000	3% of fixed capital investment	0.7% of fixed capital investment
2-B	Yadav, G. et al	2023	Research	USA	USD	2016	240.0	500.0	BTX	0.2	670.0	3.0	7,884	78,840	1364.4	914	130	16	6	0.068				2,363,000	3% of FCI	0.7% of FCI
2-C	Yadav, G. et al	2023	Research	USA	USD	2016	240.0	500.0	BTX	0.3	670.0	3.0	7,884	78,840	1212.3	787	147	16	24	0.068				2,363,000	3% of FCI	0.7% of FCI
2-D	Yadav, G. et al	2023	Research	USA	USD	2016	240.0	500.0	Ethylene	0.1	670.0	3.0	7,884	78,840	1190.5	778	137	16	6	0.068				2,363,000	3% of FCI	0.7% of FCI
3	Fivga, A. and Dimitriou, I.	2018	Research	UK	GBP	2013	2.4	2400.0	Heavy fuel oil		530.0		7,012	701	690.0		21			0.058	77	57				
4	Kulas, D.G. et al	2023	Research	USA	USD	2019	240.0	480.0	Pyrolysis oil (naphtha)	0.2	600.0		8,400	84,000	312.2	240	6		61	0.073			1% of FCI	65% of labor & mainten.	3% of ISBL	1% of FCI
5	Tan, K. Q. et al	2023	Review																							
6	Sahu, J. N. et al	2013	Research	Malaysia	USD	2011	342.9		Heavy fuel oil (diesel)	0.9	500.0	1.0	8,400	10,000	545.8				33		1,183					
7	Bora, R.R. et al	2020	Research	USA	USD	2012	250.0	288.0	Light HC fuels	0.9	500.0	1.0	8,400	100,000	306.1		82			0.052						
8	Cappello, V. et al	2022	Research	USA	USD	2016	250.0		HQL ("PAO lubricant")	0.9	300.0	11.5	8,760	91,250	445.9	365	29	20		0.070	4,290			20% of labor		2% of TCI
9	Selvam, E. et al	2023	Research						API Grade I lubricants	0.9	375.0		8,000	32,000												
10	Almohamadi, H. et al	2021	Research		USD	2020	1000.0	2000.0	Hydrocarbon fuel	0.4	450.0	1.0	7,884	328,500	70.0	9	6	9				6			3,000,000	
11	Riedewald, F. et al	2021	Research	Belgium	EUR	2020	109.6		P-oil as Fuel Oil No.6 or Heavy Bunker Oil	0.7	450.0		7,500	40,000	85.6	0	1	39	18	0.130			360,000	318,000	244,700	1% of fixed capex
12	Gracida-Alvarez, U.R. et al	2019	Research	USA	USD	2017	500.0		Gasoline (low molecular weight hydrocarbon mixture)	0.5	650.0	1.0	8,400	175,000	236.6	80	113	24		0.069	61,944				3,550,000	
13	Gracida-Alvarez, U.R. et al	2019	Research	USA	USD	2017	500.0		Gasoline (low molecular weight hydrocarbon mixture)	0.5	650.0	1.0	8,400	175,000	236.6	80	85	24		0.069	44,722				3,610,000	
14-O	Larrain, M. et al	2020	Research	Belgium	EUR	2019	210.0	360.0	Naphtha	0.4	460.0	1.0	8,000	120,000	356.3	89	30	3	132	0.074	7,153			65% of labor & mainten.	4% of (ISBL + OSBL)	1.5% of equip. cost
14-C	Larrain, M. et al	2020	Research	Belgium	EUR	2019	210.0	360.0	Naphtha	0.7	460.0		8,000	120,000	365.8	98	45	3	132	0.074	16,888			65% of labor & mainten.	4% of (ISBL + OSBL)	1.5% of equip. cost
15	Jiang, G. et al	2020	Research	UK	USD	2018	12.0	48.0	Oil/wax	0.8	500.0		8,000	8,000	297.5	90				0.100	662			10% of prod. cost	5% (ISBL + OSBL)	
16	Solis, M. and Silveira, S.	2020	Review																							
17-A1	Ocean Recovery Alliance	2015	Report	USA	USD	2014	50.0		Synthetic crude oil	0.7			8,059	15,584												
17-A2	Ocean Recovery Alliance	2015	Report	USA	USD	2014	50.0		Synthetic crude oil	0.7			8,059	18,207												
17-MKA	Ocean Recovery Alliance	2015	Report	India	USD	2014	10.0		Synthetic crude oil	0.6			7,183	3,308	158.4											
17-PS1	Ocean Recovery Alliance	2015	Report	India	USD	2014	3.0		Synthetic crude oil	0.5			7,183	992												
17-PS2	Ocean Recovery Alliance	2015	Report	India	USD	2014	6.0		Synthetic crude oil	0.5			7,183	1,796												
17-PS3	Ocean Recovery Alliance	2015	Report	India	USD	2014	12.0		Synthetic crude oil	0.5			7,183	3,977												
17-GR1	Ocean Recovery Alliance	2015	Report	USA	USD	2014	24.0		Distillate fuel oiil	0.6			7,884	4,386												
17-GR2	Ocean Recovery Alliance	2015	Report	USA	USD	2014	24.0		Distillate fuel oiil	0.6			7,884	6,316												
17-JBI1	Ocean Recovery Alliance	2015	Report	USA	USD	2014	20.0	30.0	Distillate fuel oiil	0.9			6,570	3,425												
17-JBI2	Ocean Recovery Alliance	2015	Report	USA	USD	2014	20.0	30.0	Distillate fuel oiil	0.9			6,570	8,772												
17-NF1	Ocean Recovery Alliance	2015	Report	USA	USD	2014	50.0		Distillate fuel oiil	0.7			8,410	14,308												
17-NF2	Ocean Recovery Alliance	2015	Report	USA	USD	2014	50.0		Distillate fuel oiil	0.9			8,410	23,346												
17-V1	Ocean Recovery Alliance	2015	Report	USA	USD	2014	60.0		Distillate fuel oiil	0.7			7,884	18,702												
17-V2	Ocean Recovery Alliance	2015	Report	USA	USD	2014	60.0		Distillate fuel oiil	0.7			7,884	21,152												
18-PE	Xu, Z. et al	2023	Research	USA	USD	2022	30.0		Fatty acids for soap	0.9	360.0	1.0	8,000	10,000	1142.2	628	93				2,222					
18-PP	Xu, Z. et al	2023	Research	USA	USD	2022	30.0		Fatty acids for soap	0.9	360.0	1.0	8,000	10,000	1142.2	628	93				2,222					
19-PY	Hernández, B. et al	2023	Research	USA	USD	2021	16.4	164.4	Lubricants	0.5	550.0	1.0	8,760	25,000	584.0	210	2			0.072					1,570,000	
19-HC	Hernández, B. et al	2023	Research	USA	USD	2021	16.4	164.4	Gasoline	0.7	250.0	29.6	8,760	25,000	788.0	357	0			0.072					1,790,000	
19-HTL	Hernández, B. et al	2023	Research	USA	USD	2021	16.4	164.4	Gasoline	0.3	400.0	6.9	8,760	25,000	448.0	201	67			0.072					238,000	
19-HO	Hernández, B. et al	2023	Research	USA	USD	2021	16.4	164.4	Lubricants	0.2	250.0	29.6	8,760	25,000	484.0	138	0			0.072					2,030,000	

Utilities = energy + water

kg/barrel density	0.005
L per U.S. barrel	200.000
tonne/L density	0.026

Reference											Capex																						
							TPD	TPD	hr/yr	TPY	$ mil	$/tonne	$ mil		yr	yr	yr	yr	yr	type	%	m2	$/m2	%	%	%	%	%	PRE	CRA	DIS	CUP	PUR
ID	Author(s)	Year	Article Type	Location	Currency	Ref. Year	L-Capacity	U-Capacity	Oper. time	Size	Estimate	per feed	FCI	Working capital	Lifetime	Loan	Startup	Construction	Deprec.		Salvage	Land	Land cost	Proj. contin.	Discount	Interest	Inflation	Tax rate	Process Categories				
1	Li, H. et al	2023	Research	USA	USD	2022	13.1	285.7	8,400	36,500	103.1	2,825			20				10	straight line						10%		39%	☐	☑	☑	☑	☑
2-A	Yadav, G. et al	2023	Research	USA	USD	2016	240.0	500.0	7,884	78,840	77.6			5%	30	10	0.5	3	7	straight line		40468.6	3.5	10%	10%	8%		21%	☑	☑	☑	☑	☑
2-B	Yadav, G. et al	2023	Research	USA	USD	2016	240.0	500.0	7,884	78,840	107.0		101.8	5%	30	10	0.5	3	7	straight line		40468.6	3.5	10%	10%	8%		21%	☑	☑	☑	☑	☑
2-C	Yadav, G. et al	2023	Research	USA	USD	2016	240.0	500.0	7,884	78,840	105.5			5%	30	10	0.5	3	7	straight line		40468.6	3.5	10%	10%	8%		21%	☑	☑	☑	☑	☑
2-D	Yadav, G. et al	2023	Research	USA	USD	2016	240.0	500.0	7,884	78,840	117.0			5%	30	10	0.5	3	7	straight line		40468.6	3.5	10%	10%	8%		21%	☑	☑	☑	☑	☑
3	Fivga, A. and Dimitriou, I.	2018	Research	UK	GBP	2013	2.4	2400.0	7,012	701	1.0	170		5%	20					straight line	20% of capital cost				20%	10%	5%	40%	☐	☑	☐	☐	☐
4	Kulas, D.G. et al	2023	Research	USA	USD	2019	240.0	480.0	8,400	84,000	44.9	464	39.0	15%	20		0.5	1	7	MACRS					20%		2%	21%	☑	☑	☐	☐	☐
5	Tan, K. Q. et al	2023	Review																										☐	☐	☐	☐	☐
6	Sahu, J. N. et al	2013	Research	Malaysia	USD	2011	342.9		8,400	10,000	14.9		14.9											10%					☑	☑	☑	☑	☐
7	Bora, R.R. et al	2020	Research	USA	USD	2012	250.0	288.0	8,400	100,000	434.4			5%	20		0.5	2					6% of purch. equip. cost	34% of purch. equip. cost	10%			20%	☐	☑	☐	☑	☐
8	Cappello, V. et al	2022	Research	USA	USD	2016	250.0		8,760	91,250	78.1			15%	40	43	1	3	20	MACRS	10% of TCI		1.5% of TDC	15% of TDC	10.1%	3.70%	1.90%	26%	☑	☑	☑	☑	☐
9	Selvam, E. et al	2023	Research						8,000	32,000																			☐	☑	☑	☐	☐
10	Almohamadi, H. et al	2021	Research		USD	2020	1000.0	2000.0	7,884	328,500	118.0		112.2	5%	20			2	7	MACRS	10% of capex		3% of purchased equip. cost			10%		40%	☑	☑	☐	☑	☐
11	Riedewald, F. et al	2021	Research	Belgium	EUR	2020	109.6		7,500	40,000	26.1				20	10		1	5, 10, 20	straight line				8.0%	10.0%	5%	3%		☐	☑	☐	☑	☐
12	Gracida-Alvarez, U.R. et al	2019	Research	USA	USD	2017	500.0		8,400	175,000	136.3	677	118.5	15%	20				7	MACRS			6% of purchased equip cost	34.0%	10.0%			21%	☑	☑	☑	☑	☑
13	Gracida-Alvarez, U.R. et al	2019	Research	USA	USD	2017	500.0		8,400	175,000	138.6		120.5	15%	20				7	MACRS			6% of purchased equip cost	34.0%	10.0%			21%	☑	☑	☑	☑	☑
14-O	Larrain, M. et al	2020	Research	Belgium	EUR	2019	210.0	360.0	8,000	120,000	123.0			15% of (ISBL + OSBL)	15				10	straight line		30000.0	1400.0	15% of (ISBL + OSBL)	15.0%			25%	☐	☑	☑	☑	☑
14-C	Larrain, M. et al	2020	Research	Belgium	EUR	2019	210.0	360.0	8,000	120,000	102.5			15% of (ISBL + OSBL)	15				10	straight line		30000.0	1400.0	15% of (ISBL + OSBL)	15.0%			25%	☐	☑	☐	☑	☑
15	Jiang, G. et al	2020	Research	UK	USD	2018	12.0	48.0	8,000	8,000	5.4			20%	20			2	10					10% of (ISBL + OSBL)				40%	☐	☑	☑	☐	☐
16	Solis, M. and Silveira, S.	2020	Review																										☐	☐	☐	☐	☐
17-A1	Ocean Recovery Alliance	2015	Report	USA	USD	2014	50.0		8,059	15,584	12.0																		☐	☐	☐	☐	☐
17-A2	Ocean Recovery Alliance	2015	Report	USA	USD	2014	50.0		8,059	18,207	13.0																		☐	☐	☐	☐	☐
17-MKA	Ocean Recovery Alliance	2015	Report	India	USD	2014	10.0		7,183	3,308	3.5																		☑	☐	☐	☐	☐
17-PS1	Ocean Recovery Alliance	2015	Report	India	USD	2014	3.0		7,183	992	0.3											600.0							☐	☐	☐	☐	☐
17-PS2	Ocean Recovery Alliance	2015	Report	India	USD	2014	6.0		7,183	1,796	0.4											1000.0							☐	☐	☐	☐	☐
17-PS3	Ocean Recovery Alliance	2015	Report	India	USD	2014	12.0		7,183	3,977	0.6											1800.0							☐	☐	☐	☐	☐
17-GR1	Ocean Recovery Alliance	2015	Report	USA	USD	2014	24.0		7,884	4,386	5.0																		☑	☐	☐	☐	☐
17-GR2	Ocean Recovery Alliance	2015	Report	USA	USD	2014	24.0		7,884	6,316	6.0																		☑	☐	☐	☐	☐
17-JBI1	Ocean Recovery Alliance	2015	Report	USA	USD	2014	20.0	30.0	6,570	3,425	5.0																		☐	☐	☐	☐	☐
17-JBI2	Ocean Recovery Alliance	2015	Report	USA	USD	2014	20.0	30.0	6,570	8,772	8.0																		☐	☐	☐	☐	☐
17-NF1	Ocean Recovery Alliance	2015	Report	USA	USD	2014	50.0		8,410	14,308	9.0																		☑	☐	☐	☐	☐
17-NF2	Ocean Recovery Alliance	2015	Report	USA	USD	2014	50.0		8,410	23,346	12.0																		☑	☐	☐	☐	☐
17-V1	Ocean Recovery Alliance	2015	Report	USA	USD	2014	60.0		7,884	18,702	17.0																		☐	☐	☐	☐	☐
17-V2	Ocean Recovery Alliance	2015	Report	USA	USD	2014	60.0		7,884	21,152	18.0																		☐	☐	☐	☐	☐
18-PE	Xu, Z. et al	2023	Research	USA	USD	2022	30.0		8,000	10,000	3.1		2.7	15%					10	straight line	0		5% of equip cost	10.0%		3%			☐	☑	☐	☑	☐
18-PP	Xu, Z. et al	2023	Research	USA	USD	2022	30.0		8,000	10,000	3.1		2.7	15%					10	straight line	0		5% of equip cost	10.0%		3%			☐	☑	☐	☑	☐
19-PY	Hernández, B. et al	2023	Research	USA	USD	2021	16.4	164.4	8,760	25,000	39.3				20													21%	☐	☑	☑	☑	☑
19-HC	Hernández, B. et al	2023	Research	USA	USD	2021	16.4	164.4	8,760	25,000	44.8				20													21%	☐	☑	☑	☐	☐
19-HTL	Hernández, B. et al	2023	Research	USA	USD	2021	16.4	164.4	8,760	25,000	15.9				20													21%	☐	☑	☑	☑	☑
19-HO	Hernández, B. et al	2023	Research	USA	USD	2021	16.4	164.4	8,760	25,000	50.8				20													21%	☐	☑	☑	☐	☐

FCI	total/fixed capital investment	TCI	TPI
TDC	total depreciable capital		
FCOP	fixed cost of production		
VCOP	variable cost of production		
G&A	general & administrative	overhead	
PRE	Pretreatment	sorting, washing, grinding	
CRA	Cracking	breaking C-C bonds	
DIS	Distillation	fractionation	
CUP	Chemical Upgrading	more rxns to yield higher-value products	
PUR	Purification	further separations to increase purity	

Reference					Prices in per tonne product		BTX		Benzene		Butene		Diesel/"heavy"		Diols		Ethylene		Fatty acids		Flue gas		Gasoline/"light"		Jet oil		LPG		Lubricants		Monoalcohols		Naphtha		NGLs		Oil/wax		Olefins		Propylene		Pyro. fuel/oil		Pyro. gas		Toluene	
					Yields in tonne per tonne feed		$/tonne		$/tonne		$/tonne		$/tonne		$/tonne		$/tonne		$/tonne		$/tonne		$/tonne		$/tonne		$/tonne		$/tonne		$/tonne		$/tonne		$/tonne		$/tonne		$/tonne		$/tonne		$/tonne		$/tonne		$/tonne	
ID	Author(s)	Year	Article Type	Location	Currency	Main Prod.	Price	Yield	Price	Yield	Price	Yield	Price	Yield	Price	Yield	Price	Yield	Price	Yield	Price	Yield	Price	Yield	Price	Yield	Price	Yield	Price	Yield	Price	Yield	Price	Yield	Price	Yield	Price	Yield	Price	Yield	Price	Yield	Price	Yield	Price	Yield	Price	Yield
1	Li, H. et al	2023	Research	USA	USD	Monoalcohols	780	0.12					637	0.11	3000	0.13							600								2000	0.49																
2-A	Yadav, G. et al	2023	Research	USA	USD	Naphtha	680				1270						580																380	0.40							830							
2-B	Yadav, G. et al	2023	Research	USA	USD	BTX	680	0.20			1270						580																380	0.21		0.07				0.32	830	0.16						
2-C	Yadav, G. et al	2023	Research	USA	USD	BTX	680	0.30			1270						580																380	0.26		0.05				0.20	830							
2-D	Yadav, G. et al	2023	Research	USA	USD	Ethylene	680				1270						580	0.12															380	0.19		0.15				0.40	830							
3	Fivga, A. and Dimitriou, I.	2018	Research	UK	GBP	Heavy fuel oil																																					550.00					
4	Kulas, D.G. et al	2023	Research	USA	USD	Pyrolysis oil (naphtha)																											561	0.22											0.01	0.28		
5	Tan, K. Q. et al	2023	Review																																													
6	Sahu, J. N. et al	2013	Research	Malaysia	USD	Heavy fuel oil (diesel)								0.92										0.03																								
7	Bora, R.R. et al	2020	Research	USA	USD	Light HC fuels		0.05	1020					0.06			610							0.90																	970						1970	
8	Cappello, V. et al	2022	Research	USA	USD	HQL ("PAO lubricant")																0.00							700	0.90			330	0.03														
9	Selvam, E. et al	2023	Research			API Grade I lubricants																							440	0.88																		
10	Almohamadi, H. et al	2021	Research		USD	Hydrocarbon fuel																																						0.36				
11	Riedewald, F. et al	2021	Research	Belgium	EUR	P-oil as Fuel Oil No.6 or Heavy Bunker Oil																																					210.00	0.65				
12	Gracida-Alvarez, U.R. et al	2019	Research	USA	USD	Gasoline (low molecular weight hydrocarbon mixture)	1020	0.04					840	0.05			610	0.19					860	0.54																	970	0.13						
13	Gracida-Alvarez, U.R. et al	2019	Research	USA	USD	Gasoline (low molecular weight hydrocarbon mixture)	1020	0.04					840	0.05			610	0.19					860	0.54																	970	0.13						
14-O	Larrain, M. et al	2020	Research	Belgium	EUR	Naphtha																												0.40			680	0.34						0.11				
14-C	Larrain, M. et al	2020	Research	Belgium	EUR	Naphtha																												0.69										0.15				
15	Jiang, G. et al	2020	Research	UK	USD	Oil/wax																															715	0.79										
16	Solis, M. and Silveira, S.	2020	Review																																													
17-A1	Ocean Recovery Alliance	2015	Report	USA	USD	Synthetic crude oil																																										
17-A2	Ocean Recovery Alliance	2015	Report	USA	USD	Synthetic crude oil																																										
17-MKA	Ocean Recovery Alliance	2015	Report	India	USD	Synthetic crude oil																																										
17-PS1	Ocean Recovery Alliance	2015	Report	India	USD	Synthetic crude oil																																										
17-PS2	Ocean Recovery Alliance	2015	Report	India	USD	Synthetic crude oil																																										
17-PS3	Ocean Recovery Alliance	2015	Report	India	USD	Synthetic crude oil																																										
17-GR1	Ocean Recovery Alliance	2015	Report	USA	USD	Distillate fuel oiil																																										
17-GR2	Ocean Recovery Alliance	2015	Report	USA	USD	Distillate fuel oiil																																										
17-JBI1	Ocean Recovery Alliance	2015	Report	USA	USD	Distillate fuel oiil																																										
17-JBI2	Ocean Recovery Alliance	2015	Report	USA	USD	Distillate fuel oiil																																										
17-NF1	Ocean Recovery Alliance	2015	Report	USA	USD	Distillate fuel oiil																																										
17-NF2	Ocean Recovery Alliance	2015	Report	USA	USD	Distillate fuel oiil																																										
17-V1	Ocean Recovery Alliance	2015	Report	USA	USD	Distillate fuel oiil																																										
17-V2	Ocean Recovery Alliance	2015	Report	USA	USD	Distillate fuel oiil																																										
18-PE	Xu, Z. et al	2023	Research	USA	USD	Fatty acids for soap														0.86																												
18-PP	Xu, Z. et al	2023	Research	USA	USD	Fatty acids for soap														0.90																												
19-PY	Hernández, B. et al	2023	Research	USA	USD	Lubricants								0.13										0.14		0.07				0.49																		
19-HC	Hernández, B. et al	2023	Research	USA	USD	Gasoline								0.03										0.73																								
19-HTL	Hernández, B. et al	2023	Research	USA	USD	Gasoline								0.33										0.30		0.22																						
19-HO	Hernández, B. et al	2023	Research	USA	USD	Lubricants								0.12										0.19		0.07		0.08		0.19																		

Reference										Other Revenue & Profitability							
								hr/yr	TPY	%	%	yr	$/tonne		$/tonne	Energy Recovery	
ID	Author(s)	Year	Article Type	Location	Ref. Year	Main Prod.	Main PYield	Oper. time	Size	ROI	IRR	PBP	Net Revenue	Co-products		Other Product	$/tonne
1	Li, H. et al	2023	Research	USA	2022	Monoalcohols	0.5	8,400	36,500								
2-A	Yadav, G. et al	2023	Research	USA	2016	Naphtha	0.4	7,884	78,840	9.6%			42.28				
2-B	Yadav, G. et al	2023	Research	USA	2016	BTX	0.2	7,884	78,840	8.8%			60.40		1129		
2-C	Yadav, G. et al	2023	Research	USA	2016	BTX	0.3	7,884	78,840	8.9%			60.40		928		
2-D	Yadav, G. et al	2023	Research	USA	2016	Ethylene	0.1	7,884	78,840	8.8%			66.44		1089		
3	Fivga, A. and Dimitriou, I.	2018	Research	UK	2013	Heavy fuel oil		7,012	701								
4	Kulas, D.G. et al	2023	Research	USA	2019	Pyrolysis oil (naphtha)	0.2	8,400	84,000				133.21	Pyrolysis gas	223		
5	Tan, K. Q. et al	2023	Review														
6	Sahu, J. N. et al	2013	Research	Malaysia	2011	Heavy fuel oil (diesel)	0.9	8,400	10,000	4.2%		24.0	62.00	Light fuel oil (gasoline)			
7	Bora, R.R. et al	2020	Research	USA	2012	Light HC fuels	0.9	8,400	100,000		22%			Light & heavy HC fuels, gasses, aromatics (BTX)			
8	Cappello, V. et al	2022	Research	USA	2016	HQL ("PAO lubricant")	0.9	8,760	91,250		8%			Naphtha	32		
9	Selvam, E. et al	2023	Research			API Grade I lubricants	0.9	8,000	32,000								
10	Almohamadi, H. et al	2021	Research		2020	Hydrocarbon fuel	0.4	7,884	328,500	2.5%			15.22				
11	Riedewald, F. et al	2021	Research	Belgium	2020	P-oil as Fuel Oil No.6 or Heavy Bunker Oil	0.7	7,500	40,000		20%		19.18	Tipping fees for accepting MPW as feed			
12	Gracida-Alvarez, U.R. et al	2019	Research	USA	2017	Gasoline (low molecular weight hydrocarbon mixture)	0.5	8,400	175,000	52.7%	31%	1.8	535.77	Low MWHCs, PP, ethylene		LP steam	
13	Gracida-Alvarez, U.R. et al	2019	Research	USA	2017	Gasoline (low molecular weight hydrocarbon mixture)	0.5	8,400	175,000		32%		556.03	Low MWHCs, PP, ethylene		LP steam	
14-O	Larrain, M. et al	2020	Research	Belgium	2019	Naphtha	0.4	8,000	120,000		21%			Pyrolysis gas			
14-C	Larrain, M. et al	2020	Research	Belgium	2019	Naphtha	0.7	8,000	120,000		16%			Pyrolysis gas			
15	Jiang, G. et al	2020	Research	UK	2018	Oil/wax	0.8	8,000	8,000	24.1%	24%		275.00			MHP (15 bar) steam	177
16	Solis, M. and Silveira, S.	2020	Review														
17-A1	Ocean Recovery Alliance	2015	Report	USA	2014	Synthetic crude oil	0.7	8,059	15,584								
17-A2	Ocean Recovery Alliance	2015	Report	USA	2014	Synthetic crude oil	0.7	8,059	18,207								
17-MKA	Ocean Recovery Alliance	2015	Report	India	2014	Synthetic crude oil	0.6	7,183	3,308								
17-PS1	Ocean Recovery Alliance	2015	Report	India	2014	Synthetic crude oil	0.5	7,183	992								
17-PS2	Ocean Recovery Alliance	2015	Report	India	2014	Synthetic crude oil	0.5	7,183	1,796								
17-PS3	Ocean Recovery Alliance	2015	Report	India	2014	Synthetic crude oil	0.5	7,183	3,977								
17-GR1	Ocean Recovery Alliance	2015	Report	USA	2014	Distillate fuel oiil	0.6	7,884	4,386								
17-GR2	Ocean Recovery Alliance	2015	Report	USA	2014	Distillate fuel oiil	0.6	7,884	6,316								
17-JBI1	Ocean Recovery Alliance	2015	Report	USA	2014	Distillate fuel oiil	0.9	6,570	3,425								
17-JBI2	Ocean Recovery Alliance	2015	Report	USA	2014	Distillate fuel oiil	0.9	6,570	8,772								
17-NF1	Ocean Recovery Alliance	2015	Report	USA	2014	Distillate fuel oiil	0.7	8,410	14,308								
17-NF2	Ocean Recovery Alliance	2015	Report	USA	2014	Distillate fuel oiil	0.9	8,410	23,346								
17-V1	Ocean Recovery Alliance	2015	Report	USA	2014	Distillate fuel oiil	0.7	7,884	18,702								
17-V2	Ocean Recovery Alliance	2015	Report	USA	2014	Distillate fuel oiil	0.7	7,884	21,152								
18-PE	Xu, Z. et al	2023	Research	USA	2022	Fatty acids for soap	0.9	8,000	10,000	33.0%	39%	2.6	1,284.00				
18-PP	Xu, Z. et al	2023	Research	USA	2022	Fatty acids for soap	0.9	8,000	10,000	33.0%	39%	2.6	1,284.00				
19-PY	Hernández, B. et al	2023	Research	USA	2021	Lubricants	0.5	8,760	25,000	413.0%	25%						
19-HC	Hernández, B. et al	2023	Research	USA	2021	Gasoline	0.7	8,760	25,000	-107.0%						Electricity	
19-HTL	Hernández, B. et al	2023	Research	USA	2021	Gasoline	0.3	8,760	25,000	-237.0%							
19-HO	Hernández, B. et al	2023	Research	USA	2021	Lubricants	0.2	8,760	25,000	-19.0%	-2%						

Reference										Environmental Impact			
								hr/yr	TPY	MJ/tonne	m3/tonne	kgCO2eq/kg	
ID	Author(s)	Year	Article Type	Location	Ref. Year	Main Prod.	Main PYield	Oper. time	Size	Energy use	Water use	GHG emissions	Comments
1	Li, H. et al	2023	Research	USA	2022	Monoalcohols	0.5	8,400	36,500			1.60	energy only reported in kJ/mol as Gibbs free energy barriers
2-A	Yadav, G. et al	2023	Research	USA	2016	Naphtha	0.4	7,884	78,840	11200	2836	0.74	supply chain energy requirements; supply chain + pyrolysis emissions; water use as impact category
2-B	Yadav, G. et al	2023	Research	USA	2016	BTX	0.2	7,884	78,840	9768	8470	1.03	supply chain energy requirements; supply chain + pyrolysis emissions; water use as impact category
2-C	Yadav, G. et al	2023	Research	USA	2016	BTX	0.3	7,884	78,840	8430	4260	0.91	supply chain energy requirements; supply chain + pyrolysis emissions; water use as impact category
2-D	Yadav, G. et al	2023	Research	USA	2016	Ethylene	0.1	7,884	78,840	5556	4452	0.52	supply chain energy requirements; supply chain + pyrolysis emissions; water use as impact category
3	Fivga, A. and Dimitriou, I.	2018	Research	UK	2013	Heavy fuel oil		7,012	701				
4	Kulas, D.G. et al	2023	Research	USA	2019	Pyrolysis oil (naphtha)	0.2	8,400	84,000	315		0.15	landfil avoidance scenario; GWP drops to 0.044 with heat integration
5	Tan, K. Q. et al	2023	Review										
6	Sahu, J. N. et al	2013	Research	Malaysia	2011	Heavy fuel oil (diesel)	0.9	8,400	10,000	994			energy from Aspen energy bal power (kW)
7	Bora, R.R. et al	2020	Research	USA	2012	Light HC fuels	0.9	8,400	100,000			0.62	
8	Cappello, V. et al	2022	Research	USA	2016	HQL ("PAO lubricant")	0.9	8,760	91,250			1.33	
9	Selvam, E. et al	2023	Research			API Grade I lubricants	0.9	8,000	32,000				
10	Almohamadi, H. et al	2021	Research		2020	Hydrocarbon fuel	0.4	7,884	328,500	26784			energy from MW of pretreatment, drying, grading, pyrolysis
11	Riedewald, F. et al	2021	Research	Belgium	2020	P-oil as Fuel Oil No.6 or Heavy Bunker Oil	0.7	7,500	40,000				
12	Gracida-Alvarez, U.R. et al	2019	Research	USA	2017	Gasoline (low molecular weight hydrocarbon mixture)	0.5	8,400	175,000	10705			energy as process energy inputs per section in base case (no heat exchanger network)
13	Gracida-Alvarez, U.R. et al	2019	Research	USA	2017	Gasoline (low molecular weight hydrocarbon mixture)	0.5	8,400	175,000				
14-O	Larrain, M. et al	2020	Research	Belgium	2019	Naphtha	0.4	8,000	120,000				
14-C	Larrain, M. et al	2020	Research	Belgium	2019	Naphtha	0.7	8,000	120,000				
15	Jiang, G. et al	2020	Research	UK	2018	Oil/wax	0.8	8,000	8,000	2383			energy use as electricity to heat up molten salt (pyrolysis section alone)
16	Solis, M. and Silveira, S.	2020	Review										
17-A1	Ocean Recovery Alliance	2015	Report	USA	2014	Synthetic crude oil	0.7	8,059	15,584				
17-A2	Ocean Recovery Alliance	2015	Report	USA	2014	Synthetic crude oil	0.7	8,059	18,207				
17-MKA	Ocean Recovery Alliance	2015	Report	India	2014	Synthetic crude oil	0.6	7,183	3,308				
17-PS1	Ocean Recovery Alliance	2015	Report	India	2014	Synthetic crude oil	0.5	7,183	992				
17-PS2	Ocean Recovery Alliance	2015	Report	India	2014	Synthetic crude oil	0.5	7,183	1,796				
17-PS3	Ocean Recovery Alliance	2015	Report	India	2014	Synthetic crude oil	0.5	7,183	3,977				
17-GR1	Ocean Recovery Alliance	2015	Report	USA	2014	Distillate fuel oiil	0.6	7,884	4,386				
17-GR2	Ocean Recovery Alliance	2015	Report	USA	2014	Distillate fuel oiil	0.6	7,884	6,316				
17-JBI1	Ocean Recovery Alliance	2015	Report	USA	2014	Distillate fuel oiil	0.9	6,570	3,425				
17-JBI2	Ocean Recovery Alliance	2015	Report	USA	2014	Distillate fuel oiil	0.9	6,570	8,772				
17-NF1	Ocean Recovery Alliance	2015	Report	USA	2014	Distillate fuel oiil	0.7	8,410	14,308				
17-NF2	Ocean Recovery Alliance	2015	Report	USA	2014	Distillate fuel oiil	0.9	8,410	23,346				
17-V1	Ocean Recovery Alliance	2015	Report	USA	2014	Distillate fuel oiil	0.7	7,884	18,702				
17-V2	Ocean Recovery Alliance	2015	Report	USA	2014	Distillate fuel oiil	0.7	7,884	21,152				
18-PE	Xu, Z. et al	2023	Research	USA	2022	Fatty acids for soap	0.9	8,000	10,000				
18-PP	Xu, Z. et al	2023	Research	USA	2022	Fatty acids for soap	0.9	8,000	10,000				
19-PY	Hernández, B. et al	2023	Research	USA	2021	Lubricants	0.5	8,760	25,000			0.77	
19-HC	Hernández, B. et al	2023	Research	USA	2021	Gasoline	0.7	8,760	25,000			0.66	
19-HTL	Hernández, B. et al	2023	Research	USA	2021	Gasoline	0.3	8,760	25,000			1.01	
19-HO	Hernández, B. et al	2023	Research	USA	2021	Lubricants	0.2	8,760	25,000			0.35	