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Environmental and economic implications of U.S. postconsumer plastic waste management

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ABSTRACT

With only 8.4% of end-of-life plastics collected for recycling in 2017, management of postconsumer plastic waste in the U.S. has raised significant concerns. We present an integrated approach to assess the environmental benefits and economic implications of plastic waste management by interfacing collection and sorting with process-based treatment models for energy recovery, fuel recovery and mechanical recycling. Three composite treatment scenarios are analyzed under various techno-economic contexts to understand the key drivers of greenhouse gas (GHG) emissions/savings and economic viability. At current collection volumes, both net GHG savings and net annual revenue are unfavorable. Scaling collection rates from the current level to a hypothetical 100% exposes inefficiencies beyond the collection bottleneck and results in net GHG emissions. We find that a business-as-usual increase in collection volume alone would be insufficient to offset GHG emissions and is uneconomical. We quantify the potential found in improving waste-to-energy efficiencies, developing high-yield plastics-to-fuel pathways and incorporating design for recycling considerations. From an economic standpoint, external market factors such as sale price of electricity, fuels and virgin resins are critical to financial viability of treatment processes. This analysis also allows us to appropriately assess the magnitude of investment and type of policy efforts needed to address the problem. To effectively collect and treat 100% of all U.S. postconsumer plastic waste, an upfront investment of 17–21 Billion USD is estimated. This approach underscores the importance of a systems perspective that acknowledges the complexity of the postconsumer plastic waste stream to sustainably achieve plastic waste management objectives.

1. Introduction

Today, plastics are indispensable to both consumers and industry, proving to be some of the most useful materials in modern society. However, their end-of-life fate has been under environmental scrutiny for some time: terrestrial and aquatic negative externalities of mismanaged plastic waste draw significant public concern (Rochman et al., 2013; Rochman, 2018) and resources lost to landfilling often after a short, single use (Geyer et al., 2017) underscore the inefficiency endemic to the plastic value chain. The recovery of post-consumer plastic waste has been particularly challenging in the U.S.. Even with 73% of the population equipped with curbside recycling programs (SPC, 2016), only 8.4% of the post-consumer plastic waste generated in 2017 (United States Environmental Protection Agency, 2019) was collected for recycling. This disparity is due to a number of connected reasons, including cheap availability of virgin plastic, lack of plastic products designed for recycling (Eriksen and Astrup, 2019; SPC, 2016), limited consumer knowledge of regional recycling programs (Saphores and Nixon, 2014) and the absence of a robust domestic recycling industry and market that can incentivize collection of waste plastics. This last reason has recently come under critical attention, closely following the consequences of China's waste import restrictions in 2018. With the loss of a steady recycling market overseas and social and environmental concerns about dumping of waste in technologically underequipped nations, the U.S.'s domestic collection and sorting efforts are in peril. Reports suggest recycling programs are being affected in all 50 states (Rosengren et al., 2020) due to sharp drops in demand for waste plastics.

While China's import restrictions on plastic waste have shed light on the global challenge of end-of-life plastics treatment, it also encourages innovation for recycling and offers opportunity for building domestic recycling infrastructure and creating a more transparent plastic waste market. This opportunity raises three crucial questions for the U.S.: (a) how does the plastic waste management system perform at current collection rates, (b) what are the implications of improving collection rates to a 100% and what would be needed, in terms of investment and infrastructure, if 100% of post-consumer plastic waste in the U.S. could

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be collected, and (c) how to best manage all post-consumer plastic waste domestically.

End-of-life plastic waste management can be divided into three steps - collection, sorting and treatment. Four recovery options have been generally discussed for plastic waste – mechanical recycling (Hopewell et al., 2009), chemical recycling (Rahimi and García, 2017), feedstock recycling (Al-Salem et al., 2017) and direct energy recovery (Astrup et al., 2015). Environmental assessments in the form of GHG emissions accounting or life cycle assessments commonly analyze waste management scenarios in a comparative manner, contextualized with geographic scope and goal-appropriate functional unit that cannot be easily generalized or repurposed for an integrated assessment. For instance, most studies on energy recovery deal with MSW incineration (Papageorgiou et al., 2009; Riber et al., 2008). MSW comprises organic fractions as well as recyclables such as paper and cardboard which embody biogenic carbon, while plastic waste is almost entirely composed of fossil-based carbon and leads to much higher GHG emissions. This is confirmed by many LCA studies (Lazarevic et al., 2010; Perugini et al., 2005; Chen et al., 2019) that compare treatment options, consistently ranking energy recovery as the least favorable option with the largest GHG impact. In fact, scenario analysis performed across recovery options to identify the "best" option for select end-of-life plastics usually reaffirm the order advocated by the waste management hierarchy (Lazarevic et al., 2010; Perugini et al., 2005; Shonfield, 2008; Eriksson and Finnveden, 2009; Rigamonti et al., 2014), subject to sensitivities. When waste disposal in the form of landfilling is compared with recovery options (Shonfield, 2008; Al-Salem et al., 2014), energy recovery is less favorable along the GHG impact metric. Several studies have also delved deeper into individual treatment options, such as pyrolysis (Benavides et al., 2017; Sharuddin et al., 2017; Iribarren et al., 2012) or incineration (Riber et al., 2008; amgaard et al., 2010; Burnley et al., 2015) and explored sensitivities as possible design changes to estimate environmental and cost savings.

However, such comparisons, across or within treatment options, often made on a per unit basis, sidestep two common challenges associated with postconsumer plastic waste: mixing and heterogeneity. Not only are postconsumer plastic wastes contaminated with many other types of waste materials (paper, cardboard, metals, etc.), they are themselves a diverse group of polymeric materials that have to be matched with an appropriate treatment scenario based on their resin type and contamination level. Therefore, such studies cannot be directly applied to understand the broader post-consumer waste streams in the U.S. or to plan for their efficient management. In addition to the scope and functional unit not being transferable, treatment-focused studies lack quantification of the upstream collection (Genc et al., 2019; Van Eygen et al., 2018), contamination (Vilaplana and Karlsson, 2008) and sorting (WRAP, 2009) as well as downstream secondary market challenges (Milios et al., 2018) that are viewed as the most critical bottlenecks to better plastic waste management practices. We postulate that this gap in our understanding will only widen as collection volumes increase and addressing this gap is critical to identifying pragmatic solutions that consider all stages of end-of-life management of a diverse waste stream. To do so, we develop process-based models for three treatment options – energy recovery, fuel recovery and mechanical recycling including models of upstream sorting as well as collection. By integrating the individual models into a parameterized plastic waste management framework for the U.S., we study the economic and environmental implications of improving collection volumes under different technological and legislative contexts through several scenarios. Such an analysis can illuminate hotspots that require large-scale investment and policy actions needed to develop a robust domestic recycling industry in the U.S., moving towards one that is economically and environmentally viable. The contributions of this study are three-fold: (1) integrating process-level understanding to pursue a systems level inquiry of post-consumer plastic waste treatment options; (2) using this integrated approach to study how upstream collection affects environmental and economic viability of treatment options; and (3) providing an estimate of the economic investment needed to domestically manage U.S.'s plastic waste to direct industrial efforts and shape policy.

2. Materials and methods

This study concerns postconsumer plastic waste generated in the U.S. in 2017, totaling 32.1 Tg as estimated by the U.S. EPA (United States Environmental Protection Agency, 2018). Many plastic products that consumers interact with on a daily basis are made of common thermoplastics, identified by the ASTM (American Society for Testing and Materials) International Resin Identification Code (RIC) System. In this system, a number 1–6 is assigned to the major thermoplastics (1-PETE, 2-HDPE, 3-PVC, 4-L(L)DPE, 5-PP, 6-PS) while 7 is used to denote all other unclassified polymers, comprising both thermoplastics and thermosets. The breakdown of the 32.1Tg into 1–7 plastics is shown in Fig. 1a. The RIC is also referenced throughout this article and is used in illustrating composite scenarios.

The analysis boundary for this assessment constitutes all lifecycle stages beyond the point of waste generation – from collection and sorting at materials recovery facilities (MRF) to three possible treatment options that have seen some degree of commercial utility in the U.S. – direct energy recovery at waste-to-energy (WTE) facilities, fuel recovery at pyrolysis plants and mechanical recycling at reprocessing converters and recyclers. Energy recovery refers to the combustion of plastic waste as a fuel source in a power plant to generate and distribute electricity and heat. In fuel recovery, plastic waste is pyrolyzed or treated at high temperatures in the absence of oxygen, causing them to break down to short chain carbon molecules, C_2—C_{40} alkanes, olefins, aromatics, etc. forming a product that is close to synthetic crude oil and can be catalytically upgraded to other fuels. Byproduct fuel gas is combusted in a highly efficient combined heat and power (CHP) unit to generate heat and electricity similar to energy recovery. In mechanical recycling, plastic waste is washed, decontaminated, melted and re-extruded to produce recycled plastic products that can, in theory, displace virgin plastic pellets.

The functional unit used in this study is the treatment (or disposal) of 32.1 Tg of plastic waste. Diversity within polymer resins (based on chemistry, molecular weight and architecture) is captured by considering resin and product composition. Since the primary function of the waste management model shown in Fig. 1b is the treatment of plastic waste, system expansion is employed, and all displaced products and services are credited in a substitutional manner. Generated electricity is assumed to substitute the average U.S. electric grid, which is 30% coal, 34% natural gas, 20% nuclear and 16% other renewables (Environmental Protection Agency, 2016), while generated heat is assumed to substitute industrial heat requirements of nearby operations (and be used for the heat requirements of the process itself). For fuel recovery purposes, pyrolysis oil is assumed to substitute fossil derived crude oil and other refined fuels. In the recycling option, recycled plastic pellets are assumed to displace virgin pellets of same resin with a substitution ratio that is dependent on quality considerations and market forces determined as per the methodology described in the supporting information (SI 1.5.2). A literature survey (details provided in SI 1.1) informs the practical feasibility of the end-of-life treatment options for these resins and the results are summarized in the table in Fig. 1b.

In this study, plastic waste mass is primarily routed to one of three end-of-life recovery options. These three options constitute the scenarios used in the present study as follows:

1 In Majority Direct Energy Recovery (abbreviated as Major-DER), most of the plastic waste, L(L)DPE, PP, PS and Other waste fractions (RIC 4–7) is sent to waste to energy plants while PET (RIC 1) and HDPE (RIC 2) are recycled due to their relatively mature recycling infrastructure and PVC is landfilled. This scenario is close to the

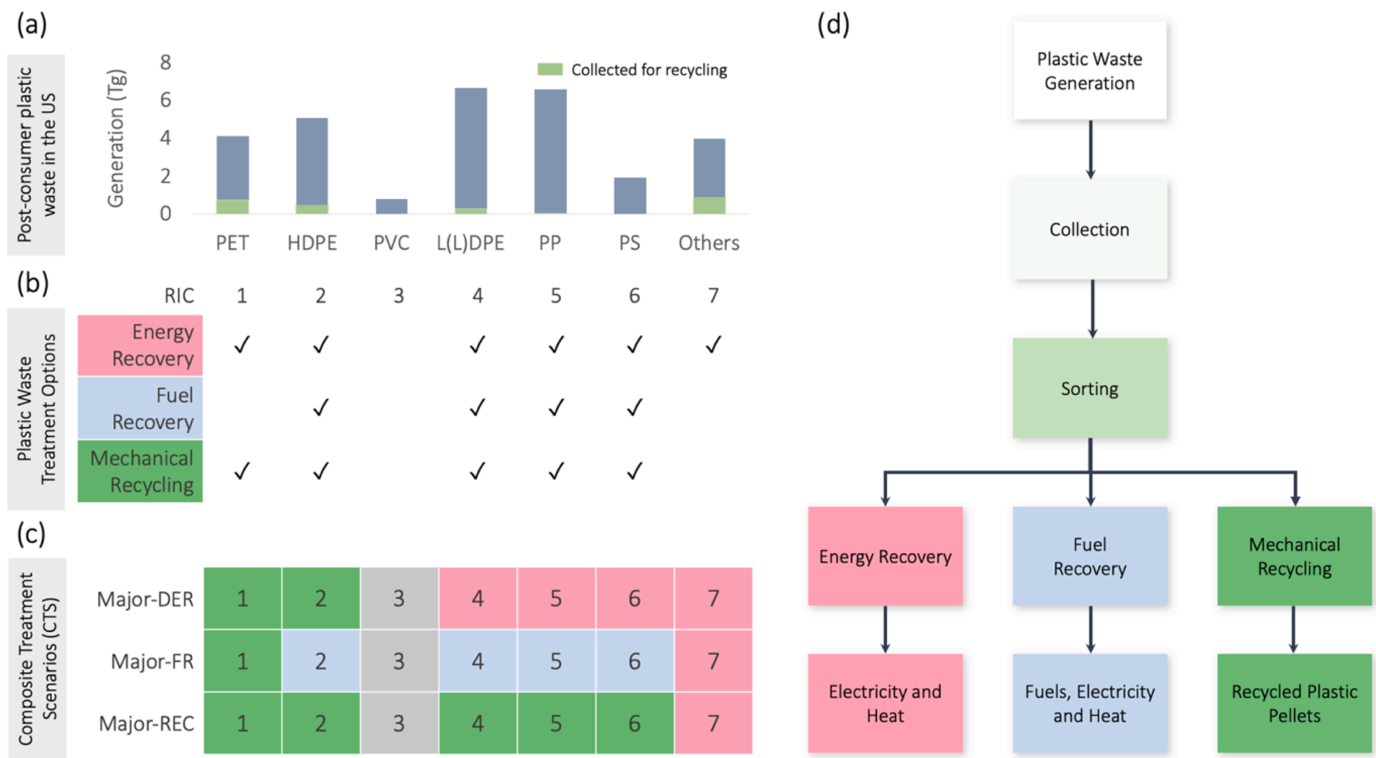

Fig. 1. (a) Post-consumer plastic waste generation in the U.S. in 2017, (b) summary of plastic waste treatment options available to each 1–7 resin, (c) composite treatment scenarios constructed for analysis in this study (bottom) and (d) analysis boundary of the study.

current state of the recovery system for several states in the U.S. (Greenpeace, 2020).

2 In Majority Fuel Recovery (abbreviated as Major-FR), all plastic types suitable to pyrolysis, namely HDPE, L(L)DPE, PP and PS (RIC 2, 4, 5 and 6 respectively) in the table are sent to fuel recovery, PET is recycled, PVC is landfilled and the Other fraction is sent to energy recovery.
3 In Majority Recycling (abbreviated as Major-REC), PET, HDPE, L(L)DPE, PP and PS are sent to recycling. PVC is landfilled and the Other fraction is once again sent to Energy Recovery.

Variation of the three composite treatment scenarios where energy recovery is replaced by landfilling is also considered to compare the two major forms of municipal solid waste disposal practiced in the U.S..

The composition of the treatment scenarios determines the plastic sorting configuration as well as purity levels needed. All residues, including sorting losses from the MRF in the sorting stage are sent to energy recovery, and therefore plastics, which are intended to be combusted after collection, are simply not set to be recovered, directing them to the residual MRF stream. For instance, Major-DER only needs sorting units for PET and HDPE in the MRF but Major-FR and Major-REC employ sorting units for L(L)DPE, PP and PS as well. Major-REC has higher sorting requirements than Major-FR due to the higher purity of bales needed for mechanical recycling into pellets than pyrolysis to produce fuel oil. Details pertaining to the working of the MRF model can be found in SI 1.5.1. Incorporating such differences in sorting requirements to accommodate the various composite treatment scenarios allows the model to approximate the heterogeneity and mixing of real-life postconsumer plastic waste.

The metrics assessed include GHG savings/emissions and the economic benefits/impacts based on the revenue/cost borne by the plastic waste management system. Process-based GHG emissions and cost models were developed for sorting and treatment stages to estimate these metrics. Plants/facilities are modelled as a sequence of unit steps, each with its own equipment and yield or efficiency parameters. Additionally, fixed capital needs as well as variable needs such as raw materials, fuel and energy usage is estimated as a function of feedstock input, processed volume, plant capacity and operational hours. The collection stage was modelled differently since techno-economic studies break down beyond regional levels and an average for the whole country was sought. Collection cost was estimated from empirical data while collection GHG emissions are derived directly from estimates in literature (complete variable definitions and details are found in SI 1.3).

GHG emissions from processing ($GHG_s^{processing}$), raw-material usage ($GHG_s^{raw\_materials}$), fuel combustion ($GHG_s^{fuel\_combustion}$), and electricity for various equipment ($GHG_s^{equipment\_electricity}$) used is accounted for (Eq. (1)). Here s refers to the end-of-life stages shown in Fig. 1d – sorting, transportation, energy recovery, fuel recovery, mechanical recycling. Sorting outputs are either sent to or bought by treatment facilities while treatment outputs are sold and substitute other products and services. Therefore, GHG emissions ($GHG_s^{emissions}$) for all the stages are added up and subtracted from GHG savings accrued from avoided production ($GHG_s^{avoided\ products}$) to calculate the system-wide net savings ($GHG_{scenario}^{net\ savings}$) for a certain scenario (Eq. (2)). A positive value of the GHG metric indicates net savings while a negative value indicates net emissions. Detailed breakdown of the GHG variables is given in SI 1.5.

$$GHG_s^{emissions} = GHG_s^{equipment\_electricity} + GHG_s^{raw\_materials} + GHG_s^{fuel\_combustion} + GHG_s^{processing} \quad \text{(Eq. 1)}$$

$$GHG_{scenario}^{net\ savings} = -GHG_{collection} - GHG_{transportation} + \sum_s \left(GHG_s^{avoided\ products} - GHG_s^{emissions}\right) \quad \text{(Eq. 2)}$$

Similarly, process-based cost models have been used often in literature to evaluate cost and revenue using operational and process characteristics (Johnson and Kirchain, 2010; Fuchs et al., 2006; Nadeau et al., 2010). Fixed costs such as building cost ($C_s^{building}$) and equipment investment cost ($C_s^{equipment}$) are annualized while variable costs pertaining to labor(C_s^{labor}), energy (C_s^{energy}), feedstock ($C_s^{feedstock}$) and raw materials ($C_s^{raw\_materials}$) as well as maintenance ($C_s^{maintenance}$) and overhead ($C_s^{overhead}$) are evaluated. The system-wide net revenue metric ($R_{scenario}^{net}$) for the scenario is then calculated by adding revenues (R_s) and subtracting costs (C_s) of each stage. An in-depth conceptual diagram of the method can be found in SI 1.5 alongside the detailed mathematical architecture. A simple view is afforded by Eqs. (3) and 4.

$$C_s = C_s^{equipment} + C_s^{building} + C_s^{energy} + C_s^{raw\_materials} + C_s^{labor} + C_s^{feedstock} + C_s^{maintenance} + C_s^{overhead} \quad \text{(Eq. 3)}$$

$$R_{scenario}^{net} = - C_{collection} - C_{transportation} + \sum_s R_s - C_s \quad \text{(Eq. 4)}$$

Information about process efficiencies, transfer coefficients, equipment specifications, material usage, as well as cost and emissions factors were gathered from a variety of sources such as techno-economic assessments, industrial and governmental reports, life cycle inventories as well as some personal correspondences. More than 600 parameters were identified and estimated, and these are grouped and listed in the SI (Tables S8-S38) along with flow diagrams (Figures S2-S9) depicting unit steps considered in the modeling of the various facilities. With such a large parametric space, associated uncertainty can be great. Two types of sensitivity analyses are performed: first, sensitivity of key techno-economic variables such as WTE efficiencies, pyrolysis temperatures, recycling yields and sale prices are explored for both the system-wide metrics in the results to discuss technological improvements and policy interventions needed to improve plastic waste management infrastructure. Second, a Monte Carlo analysis was performed on all other parameters for the three composite treatment scenarios and the top ten sensitive parameters are listed in SI 2.7 where their variability is framed in the context of present-day U.S. plastic waste treatment infrastructure.

3. Results

3.1. Current collection volumes

We assess the net annual GHG savings and revenue for the three composite treatment scenarios given the amount and composition of plastic waste at current collection volumes, at resin and product level detail, mixed with other single stream recyclables. The status quo for several U.S. states is similar to Major-DER (Greenpeace, 2020). The baseline assumptions, listed in detail in the SI, reflect today's technological feasibility. Key assumptions include, for instance: waste-to-energy plants have an electricity conversion efficiency of 22% without any waste heat recovery and displace U.S. average grid electricity; pyrolysis produces synthetic fuel oil that can displace conventional crude oil; recycling produces high grade recycled pellets from sorted post-consumer plastic waste. The result, presented as a mass flow and contribution analysis, is shown for Major-DER in Fig. 2 and contrasted with those of Major-FR and Major-REC in Figure S10.

In Fig. 2a, a Sankey diagram of the Major-DER scenario is shown. It depicts mass flows from resins collected (2.7 Tg or 8.4% of 32.1 Tg plastic waste generated) reaching intended end-of-life fates (e.g., recycled material or electricity production) as well as the losses and misrouted flows arising from mixing, heterogeneity and incompatibility of end-of-life plastics with each other and the recycling infrastructure. Sorting losses (6% of collected plastic waste or 0.16 Tg as calculated by our model) are incurred due to mis-sorting of plastics into other non-plastic waste stream such as paper or metals and are assumed to be removed and landfilled. Since the MRF model assumes all residues from near infra-red (NIR) sorting is sent to energy recovery, a portion of PET and HDPE (32% of collected 1 and 2 plastics or 0.44 Tg) destined for recycling that were discarded in the pursuit of high-grade purity, end up joining the energy recovery flow. Incompatibility losses (5.3% of collected plastic waste or 0.14 Tg) refer to products that are unsuitable for recycling despite containing the desired resin due to factors such as resin property mismatch, multi-polymer or multi-material design, additive and filler contamination, or use-phase degradation. Incompatibility loss increases as collection rate is increased as products that are not recycling-friendly enter the single stream in higher percentages (Figure S13). This is a critical limitation of scaling mechanical recycling. In the MRF model, manual sorting is used to remove incompatible objects after automatic sorting. In comparing scenarios Major-DER and Major-REC (Figure S10), we note that the recycling flow is similar in size despite a greater number of resins routed to mechanical recycling. This is because the additional PP, PS and LDPE flows are a negligible, as they are collected at <5%, compared to >15% for PET and HDPE.

In Fig. 2b, the net GHG savings of the different life cycle stages is categorized by key emitting processes. Thus, displacement of products and services is assigned a positive value, leading to GHG savings while all other factors are emitters. The largest contributor is energy recovery, not only because it has the largest mass flow, but also because almost all its emissions arise from direct combustion of plastics as fuel. The next largest contribution is from recycling and is positive due to savings from displaced virgin resin. Emissions from electricity use, sourcing of raw material, landfilling of residues and other processes are negligible. In total, the emissions outweigh the savings and 0.88 $TgCO_2e$ (or 0.31 $kgCO_2e/kg$) is emitted annually. In Major-FR (Figure S10), GHG savings are also accrued from displaced crude oil. Due to similar mass flows at low current collection rates, Major-DER and Major-REC (0.83 $TgCO_2e$) have similar net GHG emissions, and Major-FR (1.04 $TgCO_2e$) is only slightly higher (Table S39). At negative net revenue, operating plastic waste management is a net cost burden of 660 million USD per year or

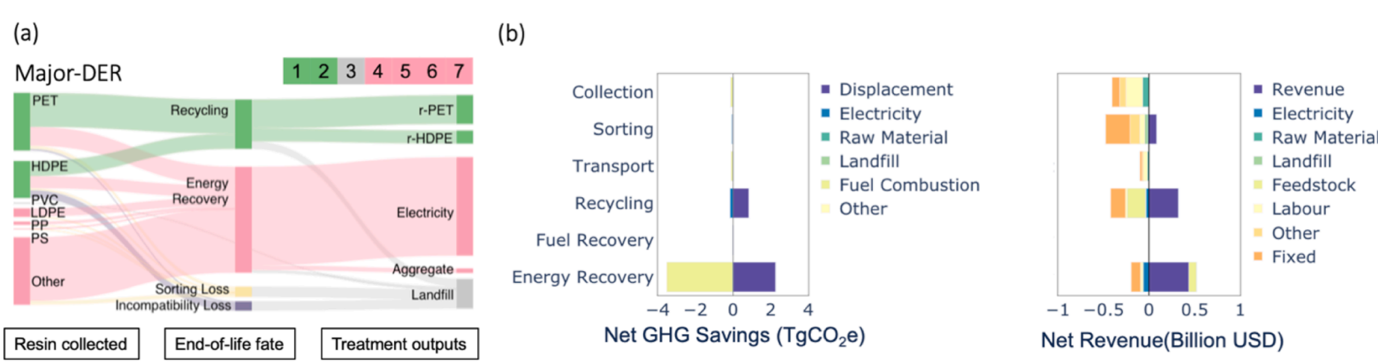

Fig. 2. (a) Mass flow Sankey diagram for scenario Major-DER, and (b) contributions to system-wide Net GHG Savings of −0.88 $TgCO_2e$ (left) and system-wide Net Revenue of −0.66 billion USD (right) for composite treatment scenario Major-DER at current (2017) collection volumes (8.4%). Positive indicates savings/revenue while negative means emissons/cost.

247 USD/Mg of collected postconsumer plastic waste. This burden increases to 960 million USD for Major-FR and 1 billion USD for Major-REC (Table S39). Most of this cost stems from collection and sorting activities. Collection is dominated by labor costs, while sorting has high fixed costs due to investment in sorting equipment. Recycling and pyrolysis costs are evenly divided between fixed costs and feedstock costs for purchased bales. The feedstock cost is positive for energy recovery because a tipping fee is paid by MRFs to incinerate the waste fraction. Revenue from sale of electricity, fuel oil or recycled resins for all three scenarios is less than total fixed and operational costs. Therefore, plastic waste management is not financially self-sustainable in the current model.

3.2. Implications of expanded collection

Improving current collection rates of plastics has been central to the discussion on sustainable end-of-life treatment. We present a hypothetical scaling of plastic waste collection volumes, with everything else being constant to our current analysis. Fig. 3 illustrates the business as usual (BAU) evolution of net annual GHG savings and net annual revenue for the three composite treatment scenarios as a function of plastic waste collection rate. This is overlaid on the contributions of various lifecycle stages and outcomes to breakdown the major costs/benefits. We find that all three scenarios are net GHG emitters, and the emissions increase with higher collection. Of the three treatment options, it is evident that the energy recovery process is a major net emitter, even in Major-FR and Major-REC where a majority of plastics are sent to fuel recovery and recycling, respectively. This is due to the direct combustion of fossil-derived, carbonaceous polymeric feedstock, the emissions of which are not quite offset by the displacement of the U.S. average grid having around 30% renewable input. Moreover, the lack of quality restrictions for energy recovery and routing of unsorted 1–6 plastics to the residual stream leads to *a* >1 yield from the MRF to the WTE plant, unlike other treatment options which face substantial yield losses along the way. *A* >1 recovery yield means more plastics than were intended ended up in the residual stream destined for energy recovery because rejects from recycling and pyrolysis destined streams also added to the residue. This also contributes to the disproportionately large emissions from energy recovery.

In contrast, fuel recovery offers net savings despite on-site fuel gas combustion in a CHP unit, negating some of the emissions associated with avoided extraction processes. Recycling also has a net positive contribution to GHG savings. Emissions due to sorting are negligible, arising mainly from equipment electricity needs, and collection and transport emissions are also minor compared to the treatment processes.

For net annual revenue, we estimate that all three scenarios also do not generate positive net revenue. Notably different from the GHG emissions breakdown, single stream collection and sorting have a high cost burden due to the high fixed costs associated with material collection and separation. Sorting cost is higher for Major-DER and Major-REC due to slightly differing causes: in Major-DER, the higher proportion of plastic waste being sent to energy recovery incurs a tipping fee while in

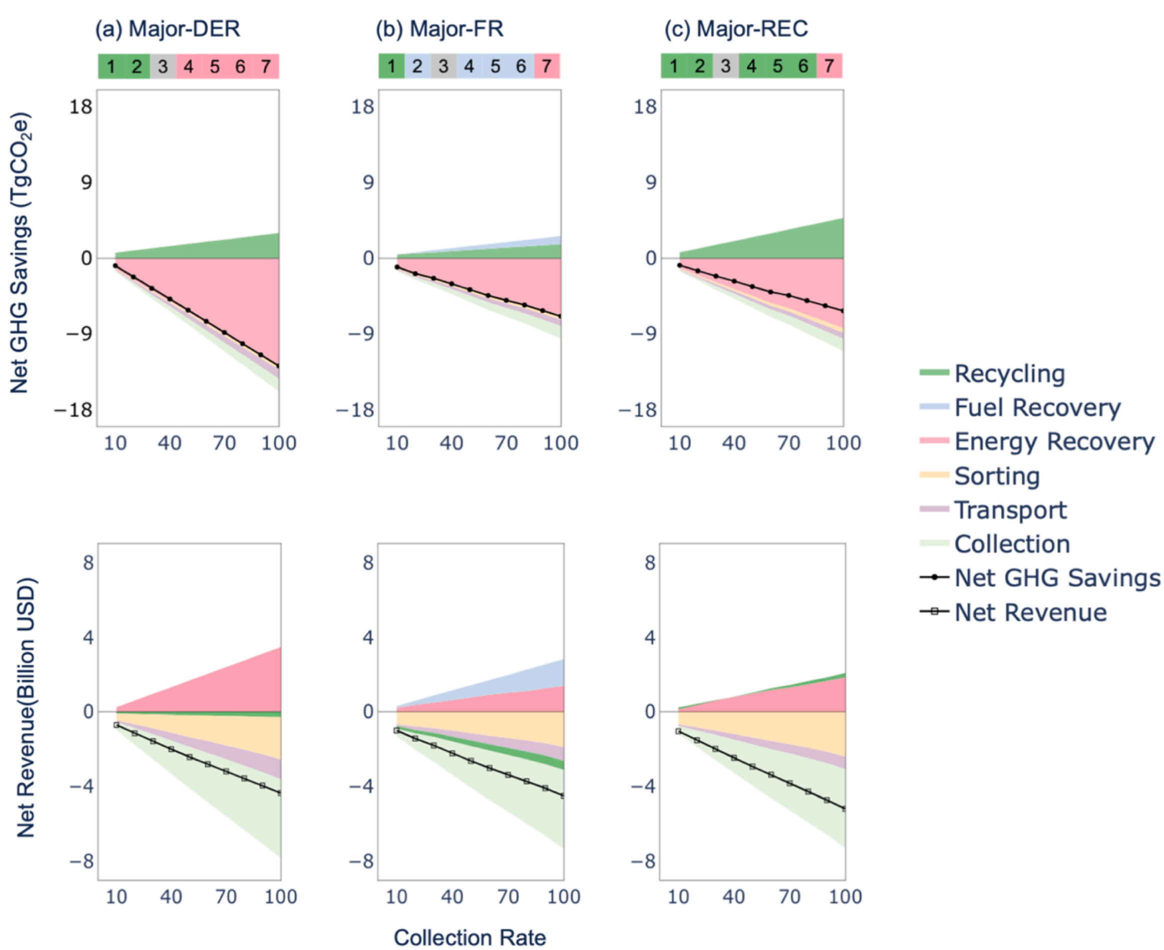

Fig. 3. BAU evolution of (top) Net GHG Savings and (bottom) Net Revenue for Major-DER, Major-FR and Major-REC as a function of collection rate. Positive indicates savings/revenue while negative means emissons/cost. Assumptions include energy recovery with an electricity conversion efficiency of 22% without any waste heat recovery displacing U.S. average grid electricity, pyrolysis producing synthetic fuel oil that can displace conventional crude oil and recycling of relevant resins into high grade recycled pellets from sorted post-consumer plastic waste.

Major-REC, the pursuit of higher purity increases sorting effort required. Energy recovery shows a large positive contribution indicating net revenue from tipping fees and the sale of electricity. Recycling is net revenue positive in Major-DER and Major-REC, where at least 2 resins are recovered, but net revenue negative in Major-FR when only PET is recovered. In all cases, the financial returns from recycling is dependent on whether it can successfully compete with virgin plastic both in terms of quality assurance and robust supply. Fuel recovery (blue area) generates net revenue, but is dependent on fuel prices (crude oil, in this case) which are subject to frequent fluctuations (will also depend on pyrolysis feed price). We explore sensitivity to economic factors such as sale price of electricity, heat, crude oil or refined fuels and virgin pellets in SI 2.7.

The unfavorable environmental and economic view provided by this assessment, particularly at increased collection rates, points to inefficiencies beyond the collection bottleneck. To demonstrate the importance of a systems perspective to identify these inefficiencies, we note similarities and differences of this study with the general LCA or GHG emissions literature related to waste plastics treatment. Table 1 compares per unit process specific GHG emissions from our analysis with those found in treatment literature for end-of-life processes. Due to differences in mass flows and composition of treatment inputs as a function of composite treatment scenarios and collection rate, a range of values are observed within our analysis. Figure S11 and S12 show that emissions and cost factors improve with increasing collection volumes due to economies of scale achieved by increased processing volumes. Our individual values generally agree with those found in literature, falling within the variability range arising from variations in scope, boundary and assumptions across studies. For sorting, our values are higher because we consider extensive plastic sorting and allocate burden from non-plastic sorting to the plastics sorting per unit GHG factor. The difference in energy recovery can be explained by the WRAP study's (Shonfield, 2008) displacement of gas fired electricity, indicating sensitivity to grid electricity. Many studies on energy recovery deal with MSW incineration (Astrup et al., 2015), which combust biogenic carbon that does not contribute to GHG emissions and reap benefits of avoided methane production in landfills. This makes MSW based WTE plants more environmentally friendly than plastic based WTE which not only has a higher carbon content but is also fossil-based. The higher carbon content also raises the calorific value such that more energy is generated per unit of plastic mass than per unit of MSW, thereby increasing the electricity or heat displaced in a plastic-only WTE facility compared to an MSW combustion facility.

A simple scaled sum of emission factors does not capture the impact of losses resulting from the diversity in postconsumer plastic waste. Many plastic products, even if collected and composed of thermoplastics that are theoretically recyclable, face practical issues in recyclability (multi-material products, irregular form, color, etc.) and are eliminated as incompatibility loss in our model. This type of loss becomes particularly important at higher collection rates (Figure S13) when all possible post-consumer plastic products are assumed to enter the single stream bin. We compare "expected" diversion rate as the sum of all collected plastic waste that is not destined for landfilling to the model-evaluated "actual" diversion rate, which accounts for sorting loss, incompatibility loss and treatment loss (details in Figure S13). To understand this, consider evaluating a recycling system with 100% collection by adding up collection and sorting (0.01kgCO2e/kg) with recycling (0.7 kgCO2e/kg), netting a benefit of 0.7 kgCO2e/kg recycled. In our Major-REC scenario, we model 100% collection rates (and 98% "expected" diversion rates) but estimate that "actual" diversion rate is only 52% due to losses throughout the process. Emissions associated with the disposition of the other 48% (incineration emissions and potential savings lost to landfilling) lead to net negative consequences (Figure S14). This is the reason why despite mechanical recycling having a much better unit GHG emissions factor compared to Pyrolysis (Table 1), scenarios Major-FR and Major-REC have similar system-wide Net GHG Savings (Fig. 3). In our model, fuel recovery is more accommodative of contamination than recycling which demands high purity. The GHG cost of this purity is not only reduced yield for recycling but also the residues being sent to energy recovery/landfilling. To avoid losing a large percentage of plastics due to incompatibility, better design for recycling practices must be instituted at the product level that encourage both consumer action as well as ensure recyclability after entering the single stream. To explore the effect of design for recycling, a compatibility yield parameter is varied in Figure S17 which illustrates that improving recyclability much before the plastic product becomes waste is a strong GHG impact improvement lever.

We find that cost per tonne will improve with increased collection volumes as a result of economies of scale (Figure S12), but a significant investment in additional infrastructure will be necessary to collect and treat the increased amount of plastic waste (Figure S15). Energy recovery has the highest upfront investment costs but the lowest annual variable costs while collection has a significant contribution to both. Sorting upfront and annual variable costs are both higher for Major-FR and Major-REC due to increased sorting equipment and labor needs. Therefore, significant investments in the plastic economy will be required to treat all plastic waste generated in the U.S. in 2017, ranging from 17 to 21 billion USD in upfront costs and 9–13 billion USD in variable annual costs. As a comparison, the current cost of curbside collection, largely paid for by local taxes (The Recycling Partnership, 2020), ranges from 4.2–5.9 billion USD for all recyclables with plastics only comprising 5% by mass of this commingled stream.

Table 1
Comparison of per unit GHG emissions factors from this study with emissions factors in literature. Negative = savings; Positive = burden.

End-of-life Process	Emission Factors in this study (kgCO2e/kg)	Emission Factors in literature (kgCO2e/kg)	Variability due to:
Sorting	0.026–0.042	0.004–0.015 (Anthony Rush Combs, 2012) 0.011 (Fitzgerald et al., 2012)	Type of recycling scheme (single stream, dual stream, mixed waste, pre-sorted, etc.)
Energy Recovery	0.826–0.852	1.83 (Shonfield, 2008)	Plastic waste composition and source of electricity displacement.
Pyrolysis	-(0.01–0.05)	-(0.15–0.25) (American Chemistry Council and RTI International, 2012) -(0.03) (Iribarren et al., 2012) 0.37 (Faraca et al., 2019)	Estimates based on pilot processes using proprietary technologies with possible differences in reactor types, process flows, catalysts, temperatures, etc.
Mechanical Recycling	-(0.928–0.965)	-(0.1–1.1) (Brogaard et al., 2014) -(0.45–0.57) (Shonfield, 2008) −0.45[17] -(0.25–2.32) (Turner et al., 2015)	Differences in resin type, contamination levels, recycling scheme, quality and other yield assumptions.
Chemical Recycling	Not enough data for process-based modeling.	−1.47 (Shen et al., 2010) −0.98 (Patagonia, 2005) (0.13 - 0.86) (Nakatani et al., 2010)	Due to lack of transparency about estimation of process emissions, it is difficult to ascertain exact cause of the range of variation but difference in end product (bottle grade, fiber-grade resin) as well as intermediaries (BHET, DMT) are noted.

3.3. Identifying key assumptions

Fig. 4 explores the sensitivity of net GHG emissions (first row) and net revenue (second row) to variation in key technological parameters including electric efficiency of a WTE plant (column 1), heat to power ratio of a WTE plant equipped with heat recovery (column 2), operation temperature of a pyrolysis plant (column 3), and grade of recycled resin produced at a recycling facility (column 4). For the Major-DER scenario, improving electric efficiencies beyond the current WTE state-of-the-art in the U.S. improves environmental performance, as shown in Fig. 4a. As a benchmark, coal power plants average 34% efficiency (EIA, 2020) and best practices natural gas combined cycle plants are 53% (EIA, 2019). Another way of improving the Major-DER scenario could be utilizing waste heat for industrial heating, with better performance seen at higher power conversion efficiencies such as those seen in gas turbines and fuel cells. However, in this case, the environmental benefit is incumbent on the geographical proximity of other energy intensive industrial centers with medium temperature steam requirements. Net annual revenue shows similar trends. Higher efficiencies improve revenue as more electricity and/or heat is sold. The cost associated with improving efficiencies is not included, and the cost of building and operating more efficient plants is assumed to be the same as current WTE plants.

For fuel recovery at higher temperatures, gas yield increases (Figure S8), which improves environmental performance as fuel gas is assumed to be combusted on site in efficient CHP units. Using higher pyrolysis temperatures can offer GHG savings for Major-FR (Fig. 4b), but this is a weak lever compared to the ones described for Major-DER. From an economic standpoint, the Major-FR scenario is net revenue negative and shows the opposite trend with temperature, as the extra cost of building and operating highly efficient CHP units is not entirely offset by sale of electricity. Two other fuel options, a mixture of naphtha, diesel and heavy fuel oil obtained by refining crude oil and catalytically produced gasoline, were also considered (Figure S16). They also result in negative impact as the energy and cost of further refining offsets benefits of a more refined fuel.

For Major-REC, sensitivity is performed over the 4 grades (1 - food-grade, 2-high, 3-medium and 4-low, see SI 1.5.2 for explanation). The grades are indirectly tied to yields. Improving quality allows for better substitution rates, leading to higher displacement credit, but reduces recycling yields. The economics of the Major-REC scenario is net revenue negative for all grades with the assumed yield loss parameters. Among the grades studied, medium grade is least net revenue negative. Improving quality to high and food-grade increases costs disproportionately due to the extra equipment and energy required. In fact, in the high and food-grade scenario, neither GHG savings nor positive net revenue can be realized. This points to an optimum quality or grade of recycled plastic beyond which high costs associated with intensive sorting and more complicated machinery as well as increased selectivity or more loss make the scenario less preferred.

From the range of possibilities presented in Fig. 4, at first glance, building more efficient WTE plants seems most promising. However, we assume a displacement of U.S. average grid electricity, which is fossil intensive. Displacing a greener grid would reduce environmental benefits and may even plunge it into net emissions, meaning it may not be a good option in the long term (Figure S18). Nevertheless, there will

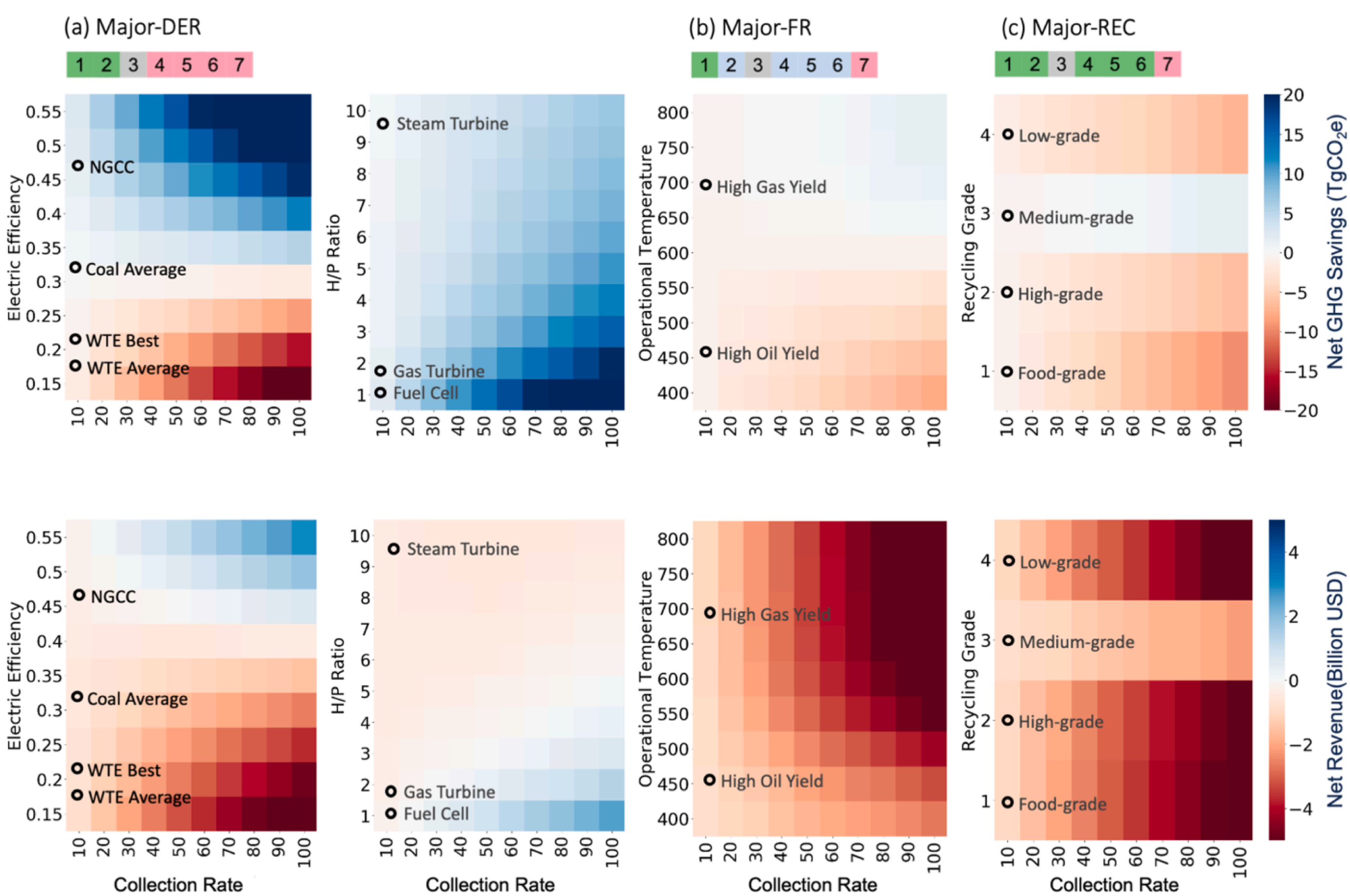

Fig. 4. Sensitivity of key yield/efficiency parameters – (a) electric efficiency and heat to power ratio of a WTE plant in Major-DER, (b) operation temperature of a pyrolysis plant in Major-FR and (c) grade of recycled resin in Major-REC. For axes Net GHG Savings and Net Revenue, positive indicates savings/revenue while negative means emissons/cost.

always be a fraction of plastic waste that cannot be recycled or pyrolyzed, and for that reason, improved WTE efficiencies must be pursued. Additionally, industrial heat is more difficult to renewably electrify (IEA Paris, 2018). Using heat by cogeneration requires colocation of industries with use for medium temperature steam and should be actively considered for new installations. For existing plants, providing residential heat to neighboring communities can be expanded, albeit with lower environmental benefits (Figure S19). Since pyrolysis fuel gas is used to power CHP operation that also generates electricity and heat – the same caveats apply to the Major-FR scenario as well.

3.4. Towards good practices

Our analysis identifies the various inefficiencies and limitations inherent in the current end-of-life plastics value chain and sheds light on several different levers for improvement. We leverage this understanding to identify good practice contexts to better explore the nuanced answer to our third question, how to best manage postconsumer plastic waste in the U.S..

Fig. 5 visualizes the economic and environmental metrics together for business-as-usual (BAU), 'good practices', and 'replacing DER by Landfill' scenarios as collection volumes are increased. BAU scenarios in the third quadrant are neither environmentally nor economically advantageous. To craft good practices scenarios that improve environmental savings, we vary intrinsic technological parameters relevant to the largest treatment option in the three composite scenarios. Extrinsic contextual parameters are also varied to evaluate the impact or benefit of possible evolution in energy (Electric Grid), material (Product Design) and market factors (Oil Price). For Major-DER, the BAU scenario is the least environmentally preferable option. Informed by previous sensitivity analysis, two improvement options are presented for Major-DER, (a) increasing electric efficiency from 0.22 in BAU to 0.34 and (b) operation of the incineration plant as a combined heat and power plant (total efficiency = 0.7, H/P ratio = 5) with medium steam export to nearby industries. The latter is not only better in the current context but is also more robust to the expected transition of the electric grid towards

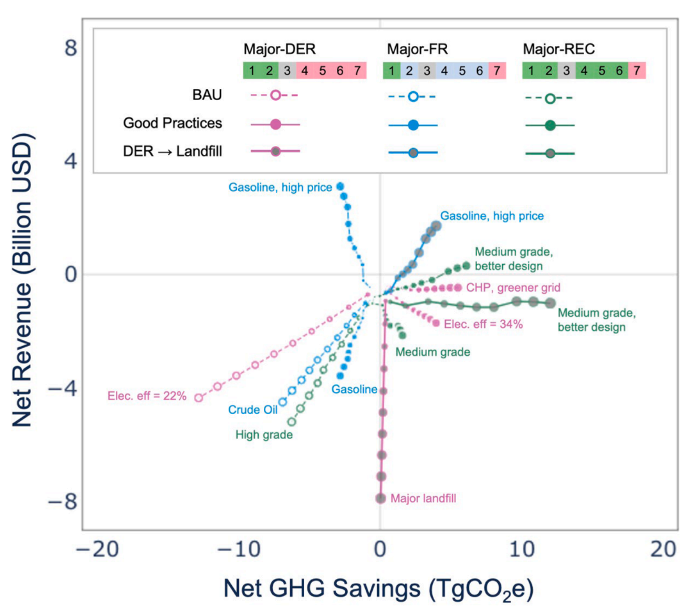

Fig. 5. Economic and environmental trade-off as a function of collection volumes (size of circles vary with collection volume). Business as usual (BAU) scenarios follow assumptions from current (2017) context as shown in Fig. 3. Good practices scenarios are crafted to improve environmental savings from the BAU. DER → Landfill considers landfilling instead of combustion for energy recovery in all three composite scenarios. Positive indicates savings/revenue while negative means emissons/cost.

renewable sources in the future. A key barrier to the adoption of this good practice is the co-location of appropriate industries with steam requirements. For Major-FR, producing gasoline directly through catalytic pyrolysis can improve scenario performance compared to crude-oil through thermal pyrolysis, particularly on the economic side if future oil prices are high. However, catalytic pyrolysis of plastic waste for gasoline production is still a lab-scale concept whose technology readiness is limited by low and/or inconsistent yield. For Major-REC, reducing quality requirements to make medium grade recyclates can reduce material loss in pursuit of purity and improve both metrics for the scenario compared to BAU. Furthermore, if better design for recycling principles are instituted in the packaging industry, it is possible to reach the first quadrant where both GHG savings and profits can realized when 70% or more of plastic waste can be successfully collected. We note that the scope of this analysis only considers single stream recycling and within this constraint, higher quality grades perform worse than the medium grade due to mixing and heterogeneity discussed before. This points to a need for source separation and other collection methods to successfully recycle at high and food-grade levels. There is also significant opportunity in the development and adoption of better collection, sorting and reprocessing unit operations that would improve the constituent recoveries. Since energy recovery contributes to GHG emissions from combustion greatly, we also considered the option of landfilling instead (DER → Landfill). If in Major-DER, RIC 4–7 plastics are landfilled along with RIC 3 (called Major landfill) we see that there is carbon neutrality but at a great economic cost. In Major-FR with catalytic pyrolysis producing gasoline sold at high prices, if RIC 7 plastics are landfilled instead of combusted for energy, the scenario nets positive GHG savings. However, net revenue is reduced, and the scenario is financially viable beyond 50% collection rate threshold, compared to a 30% threshold with energy recovery. A similar trend of improved GHG savings potential but reduced net revenue is seen in Major-REC as well. When the scenario that produces medium-grade recycled pellets given better design for recycling practices is modified to send RIC 7 plastics to landfill, net revenue is never positive but higher GHG savings can be realized. From a purely GHG emissions perspective, landfilling plastic waste is better than incinerating it until better electric efficiencies can be achieved or broader adoption of CHP practices is realized. However, landfilling is a short-term solution to an exponentially growing problem and its environmental impact on land use and contamination as well as non-renewable energy use offers a different perspective. One study ranked landfilling as the worst option along all end-environmental metrics evaluated except global warming potential (Shonfield, 2008).

It is interesting to see that the better "good practice" depends not only on technological advancement of treatment facilities and external context but also on collection volume. This indicates that a phased response to the plastics treatment problem, with continuous evaluation of its performance, is needed.

4. Discussion

Previous work comparing treatment options for plastic waste reveals a pattern mirroring the waste management hierarchy. This is observed in our emissions factors as well: recycling leads to a higher environmental benefit than fuel recovery, which is in turn more beneficial than energy recovery. However, such comparisons become less useful as we try to expand the scope to address the diverse and heterogenous postconsumer plastic waste in a vast geographical region such as the U.S.. In the U.S., the low collection rates have been criticized (United States Environmental Protection Agency, 2020) and characterized (United States Environmental Protection Agency, 2018) to various extents. Seldom has scrutiny extended beyond the collection bottleneck to identify and address barriers (US Environmental Protection Agency, 2019) in the broader end-of-life plastics treatment infrastructure. We find that any improvement in collection activity alone will be insufficient to minimize GHG emissions impact, and the economic burden of doing so will deter direct action. While improving collection rates is important to avoid landfilling plastic waste and their leakage into the environment, changes to plastic waste facilities and flows must be pursued alongside to do so sustainably.

For energy recovery, improvement of process efficiencies and utilization of waste heat have both been widely recommended (Papageorgiou et al., 2009; Eriksson and Finnveden, 2009). We find that improving beyond 30% efficiency is enough to offset carbon emissions of the fossil intensive grid (Fig. 4). If the average grid becomes less fossil-intensive, this efficiency will have to be higher to meet carbon-neutrality. A better option in that case could be the effective use of waste heat: if 40% of incinerators operated in CHP mode, and non-CHP plants operated at 30% efficiency, we find it is possible to have net positive GHG savings even when the grid is 50% renewable (Figure S20). To have net positive revenue in fuel recovery, advanced catalytic pyrolysis to refined fuels such as gasoline that fetch a high price, over 5 USD/gallon, is desired (Figure S23). In the four recycling grades investigated, higher quality does not necessarily mean more environmental savings. This is a consequence of upstream resin and product design practices that can limit technical yields due to many plastic waste products being incompatible with each other or the recycling process itself. Examples include multi-material (Ellen MacArthur Foundation, 2017), multi-polymer (Eriksen et al., 2019) products as well as single polymer products containing contaminants (Vilaplana and Karlsson, 2008; Hahladakis et al., 2018; M.K. Eriksen et al., 2018) objectionable (FDA, 2006) to the desired secondary end-use. The incompatibility problem is impossible to solve at the point of end-of-life alone, as it requires a degree of sorting beyond practicality. The answer lies further upstream than the scope of this study – "design for recycling" practices must be implemented at production and manufacturing stages. We show that improving compatibility yield through better design can considerably improve both system wide net GHG savings and net revenue (Figure S17). Other important upstream interventions include plastic waste reduction mechanisms such as new delivery or reuse/refill models as well as elimination of overpackaging. Plastic waste reduction is not explored in this study in order to conserve the functional unit of 32.1 Tg across scenarios for simple comparison.

Such a change can be encouraged through policy tools such as extended producer responsibility (OECD, 2006) (EPR), but considering the sheer extent of the plastics industry such a change is bound to be gradual. An end market study for recycled products (More Recycling and Plastic Forming Enterprises, 2017) suggests that demand for domestic post-consumer resin does not meet the supply. Commonly cited barriers for adoption of recycled resins were the lack of a price advantage of virgin resin and unavailability of recyclates that match the product manufacturer's specifications. In the meantime, using a tiered approach to recycling where all recycling grades are optimally pursued by identifying appropriate secondary use-markets will be crucial. This would require supplementing single-stream collection for low and medium grade products with efficient use of deposit and buy-back schemes to capture the high-volume food-grade and high-grade products for recycling with minimal loss or contamination. EPR can also be invoked to support such targeted collection efforts (H.R., 2020). Material taxation at the point of manufacture based on recycling-friendliness of the product can not only allow for fiscal support for downstream processes but also help in incorporating tenets of practical recyclability early in the supply chain.

This work answers the questions central to developing robust domestic end-of-life plastic waste management system in the U.S. related to current performance, requirements for increasing collection rates, and best management of all post-consumer plastic waste. By analyzing performance at current collection volumes, we find that economic and environmental unfavourability points to critical inefficiencies in the system. Scaling collection volumes in a BAU manner exposes challenges inherent to the heterogenous waste stream, namely, sorting and incompatibility losses, and highlights the need for an integrated assessment. We estimate that an investment of 17–21 billion USD and annual cost of 9–11 billion would be needed to manage 100% of the generated plastic waste, if it can be collected. This underscores the need for dedicated policy tools and legislative efforts moving forward to engage a sustainable domestic recovery infrastructure. If current processing trends are continued, then even at high collection volumes, the system would emit 12 $TgCO_2e$ annually, primarily due to direct energy recovery. To mitigate emissions arising from treatment, sensitivity on key technical parameters and market factors were leveraged to identify good practices scenarios that provide context-dependent answers. Utilizing waste heat from direct energy recovery through highly efficient CHP processes can provide major environmental benefits. In the absence of waste-to-energy efficiency improvements, if sorting residues are landfilled instead of incinerated, catalytic pyrolysis producing gasoline at high yields can provide net positive revenue and GHG savings. In the long run, we show that with better design for recycling for PET, PE, PP and PS, both net revenue and net GHG savings can be realized at high collection volumes.

We note several limitations of this study. First, a singular environmental metric, GHG emissions/savings is used to characterize and comment on environmental impacts or benefits; this limits our ability to comment on what constitutes a better option (Shonfield, 2008). Moreover, it is possible to miss key concerns surrounding some recovery/-treatment options. For instance, in our study, landfilling appears to be better than energy recovery (Fig. 5) from a purely GHG perspective but other metrics that account for land use, solid waste or non-renewable energy use rank landfilling lower (Shonfield, 2008). Second, our scope is limited to post-consumer plastics in the MSW stream and doesn't consider other sources of plastic waste such as automotive shredder residues and post-commercial streams that do not enter the single stream scheme. We focus only on the recycling of the post-consumer plastic wastes, yet in the real world the production volumes of virgin plastics, and recycled plastic production/manufacturing scrap, both exceed current post-consumer recycling volumes. Our boundary choice allows us to analyze the fate of the plastics fraction in detail but a broader consideration of MSW would be needed to comprehensively comment on economic and environmental benefits of collection and sorting in the single stream scheme. Third, the role of markets for recycling has been understated and only incorporated through the substitution rates based on a European study (M.K. Eriksen et al., 2018). A value chain analysis (Milios et al., 2018) is needed to understand supply-demand dynamics. Fourth, the model collection costs are extrapolated from low collection rate following economies of scale and miss any factor that might be needed to incentivize human behavior to reach the hypothetical high collection rates. Therefore, the collection cost and all subsequent cost metrics are a lower bound. It is notable that while this high-level analysis indicates that the system would not become more economically viable simply by increasing volume, it is likely that there are local deviations from this. In some locations, where transportation and sorting costs are low, increased volumes might lead to positive net revenues. Spatially granular models should explore the regional variations in these results. Future work should also focus on spatial heterogeneity within the U.S.. In the absence of federal directives on post-consumer plastic waste treatment, different regions have their own waste management schemes (SPC, 2016), practices (Energy Recovery Council, 2018), policies (NCSL, 2020) and attitudes (Saphores and Nixon, 2014; EEC, 2020) that have not been captured in this high level, national study. Further if the MSW recycling system is structured correctly and right products are chosen, there may be more than enough value in the system products to pay for collection, transportation, sorting and processing of the sorted fractions. However, to achieve such economically viable system one must optimize the recovery of all recyclable/reusable fractions including, metals, plastics, paper, and compostables, and from a broad array of other collection streams.

CRediT authorship contribution statement

R. Basuhi: Conceptualization, Data curation, Formal analysis, Writing - original draft. **Elizabeth Moore:** Conceptualization, Data curation, Formal analysis, Writing - original draft. **Jeremy Gregory:** Conceptualization, Supervision, Writing - review & editing. **Randolph Kirchain:** Conceptualization, Methodology, Writing - review & editing. **Adam Gesing:** Conceptualization, Data curation, Writing - review & editing. **Elsa A. Olivetti:** Conceptualization, Writing - original draft, Supervision.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.resconrec.2020.105391.

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