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Abstract

As demand for plastic increases, there is an urgent need to ramp up its collection and recycling rate. This study reports results of a pilot study in the United States to recycle flexible plastic packaging (FPP) from single stream curbside collection focusing on both the economic feasibility and carbon footprint. To explore the marketability of recycled FPP, four down-stream market pathways were studied, including roof coverboard, plastic pellets, pallets, and film. Results indicated that (1) the cleaning and pelletizing process at reclaimers contributed the most to the total greenhouse gas (GHG) emissions of recycled FPP products, and (2) the GHG emissions of recycled FPP products in all four pathways are lower than the comparator products. Therefore, using a higher percentage of recycled FPP substituted in a product can result in greater GHG emission reductions. All the recycled FPP products also showed favorable economics compared to their direct competitors.

Key words: flexible plastic packaging, material recovery facility, plastic recycling, life cycle assessment, techno-economic analysis, carbon footprints

1.0 Introduction

Flexible packaging is one of the most prevalent formats of packaging both globally and in the U.S. and is expected to grow at a CAGR of 4.3% through 2028 (Vantage Market Research, 2022). Flexible plastic packaging (FPP) consists of single-resin plastic films and multilayer packaging such as bags, wraps, lay-flat pouches, standup pouches, and shrink bundling (Resource Recycling Systems (RRS), 2020). In 2019, on a unit basis flexible packaging was the largest format globally, with about a 28% percent share followed by PET bottles at 17% (Matt Leonard, 2020). For more than twenty years in the U.S., there has been a shift towards the replacement of heavier formats of packaging like glass and steel with plastic and in the past decade a steady shift to replace any rigid packaging formats like glass, steel or plastic with flexible packaging (RRS, 2019). Food packaging is the largest market for flexible packaging accounting for about 52% of all shipments (Bobby Elliott, 2016). Smaller but growing markets include medical and pharmaceutical applications 9%, consumer products (i.e., home and laundry products) at 8%, personal care application 6%, and pet food at 4% (Flexible Packaging Association, 2021). The shift to flexible packaging has been driven largely by cost savings, and improvements in both technical and environmental performance. As a lighter packaging format, flexible packaging results in significant benefits across the supply chain from the reduction in overall material use, being able to transport more units with lower weight, and performance benefits like prolonged shelf life and less product loss. These benefits result in significant cost and greenhouse gas (GHG) emissions savings. However, the trade-off in using flexible packaging compared to other mono-material packaging has been its lack of recyclability due to the complexity of flexible material structures that prevent them from being mechanically recycled (Eriksen, Christiansen, Daugaard, & Astrup, 2019). Therefore, recyclable-by-design mono-material flexible packing has been developed using a design-for-recycling approach (Guerritore et al., 2022; Pettersen, Grøvlen, Evje, & Radusin, 2020). Advances have been made to shift to mono-material flexibles where applicable, compatibilizing additives have been developed to enhance mechanical recycling, and new barrier coatings can be used to address functional performance and improve recyclability (Clare Goldsberry, 2020). However, with the growing focus on emissions, flexible packaging is likely to continue to grow and remain materially complex.

The main economic challenges to plastic recycling are high operating/collection costs, low consumer participation, low capacity and efficiency of plastic sorting operations, and insufficient volume of low cost, high-quality recyclable materials (Hennlock et al., 2015; Milios et al., 2018). Many efforts have been made to predict the quantity of recyclable waste and its quality and potential applications. For example, M. T. Brouwer et al. (2018) developed a model to predict waste generation and polymetric compositions of post-consumer plastic packaging recycling networks in the Netherlands. The model was used to study the trade-offs between quality and quantity of recycled plastic materials and reveals that more research effort is required to meet the quality requirements for a circular economy (M. Brouwer et al., 2019). In addition, many cost analyses have been performed in European countries. For instance, Groot, Bing, Bos-Brouwers, and Bloemhof-Ruwaard (2014) has conducted a comprehensive cost analysis that evaluate waste collection cost of different collection scheme in Netherlands. There was a U.K. study that analyzed the cash flow of a plastic recovery facility with an annual throughput of 80,000 tonnes/year (70% mixed rigid plastics and 30% plastic films) (AxionConsulting, 2009). The results showed that with a £29.5 million (~ $41 million using 2009 exchange rate) capital investment, the net present value (NPV) for the recovery facility was about £14.5 million (or $20.2 million) for a 27% internal rate of return for 10 years. However, currently there is a lack of techno-economic analyses (TEA) published in

the U.S. to evaluate mechanical recycling of plastic, largely due to confidentiality reasons (Bora, Wang, & You, 2020). This study aims to evaluate cost and economic feasibility of recycling FPP in a commercial single stream MRF in the U.S. and will provide a valuable data base for both industry and academia.

The GHG emissions and energy consumption associated with production of recycled and virgin plastic is another important area of study. For example, 1 kg of recycled polyethylene terephthalate (PET), polypropylene (PP), high density polyethylene (HDPE) resins can reduce total global warming potential by 1.87 kg, 1.33 kg, and 1.31 kg CO2 eq, respectively, compared to virgin plastic pellets (Association of Plastic Recyclers, 2018). Results from multiple life cycle assessments (LCAs) indicate that the composition of the incoming recyclable stream and sorting technologies are the largest factors impacting total GHG emissions within materials recovery facility (MRF) processes (Astrup, Fruergaard, & Christensen, 2009; Pressley, Levis, Damgaard, Barlaz, & DeCarolis, 2015). Single stream and dual stream from three studied MRFs consumed 13.8 kWh and 11.5 kWh of electricity per 0.907 tonne of material processed, respectively Fitzgerald, Krones, and Themelis (2012). Products made from recycled PET (Westerhout, Van Koningsbruggen, Van Der Ham, Kuipers, & Van Swaaij), HDPE, and PP consume less expended energy (e.g., energies consumed during transportation or operation processes) than virgin plastic (APR, 2020; Benavides, Dunn, Han, Biddy, & Markham, 2018). However, before claiming that recycled plastics are more energy efficient, its quality compared to virgin plastic is a key uncertainty that needs to be considered. A study found that out of 84 plastic recycling programs, over 58% of recycled plastic do not meet current market quality requirements Eriksen, Damgaard, Boldrin, and Astrup (2019). The low quality of recycled plastic, often caused by material aging, contamination levels, and melting and reforming steps in the recycling process can limit its application and replace virgin materials (Patoski, 2019; Ragaert, Delva, & Van Geem, 2017). Horodytska, Valdés, and Fullana (2018) has reviewed many LCA studies and concluded that LCA results are very sensitive to the efficiency of the sorting and recycling technologies, more importantly, the quality of the recycled plastic and rate of substitution of virgin material. To correctly assess the environmental and economic performance of a plastic recovery system, it is critical to identify end products with comparable market quality and the substitution ratio of recycled materials in them.

This study examines a pilot that was undertaken to equip a MRF to sort flexible plastic packaging (along with mono-resin films) collected in a curbside program and evaluates the economic and environmental feasibility of sorting and producing bales of FPP (rFlex) in a commercial single stream MRF. By conducting the TEA and LCA for the plastic sorting, recycling, and downstream manufacturing processes, we aim to answer four specific research questions:

1) What are the costs and GHG footprints of recycling FPP to produce an rFlex bale?
2) Which market pathways would be most receptive to rFlex in terms of technical and quality requirements, costs, and environmental footprint?
3) How sensitive is the material substitution ratio of recycled material to virgin material on the cost and GHG footprints for each potential market pathways?
4) If FPP recycling and certain market pathways are not cost-competitive, how can improvements be incorporated to increase the market economic incentive (e.g., high selling price, additional plastic processing fee, plastic tax, carbon credit)?

2.0 Methodology

2.1 System boundary/scope

The system boundary (or research scope) considered in the FPP Recycling scenario as shown in Figure 1 has three stages. In stage 1, the MRF sorts all collected FPP to produce rFlex. Stage 2 is to ship rFlex to the reclaimer for cleaning and pelletizing, and Stage 3 is to send FPP pellets to converters/end markets. Cost and operational data were collected from a MRF that was upgraded to sort FPP from single stream recyclables (Resource Recycling Systems (RRS), 2020). Although the collected data were from a pilot scale (i.e., FPP was not being collected from 100% of households), the calculations were done for a commercial scale recycling operation. Therefore, the results of this study reflected a scaled-up operation. The FPP recycling scenario could be found in Figure 2 (stage 1) where six yellow blocks represent upgraded equipment to an existing MRF to sort FPP into rFlex.

Section 2.3 and 2.4 provide more details on LCA methodology and TEA methodology, respectively.

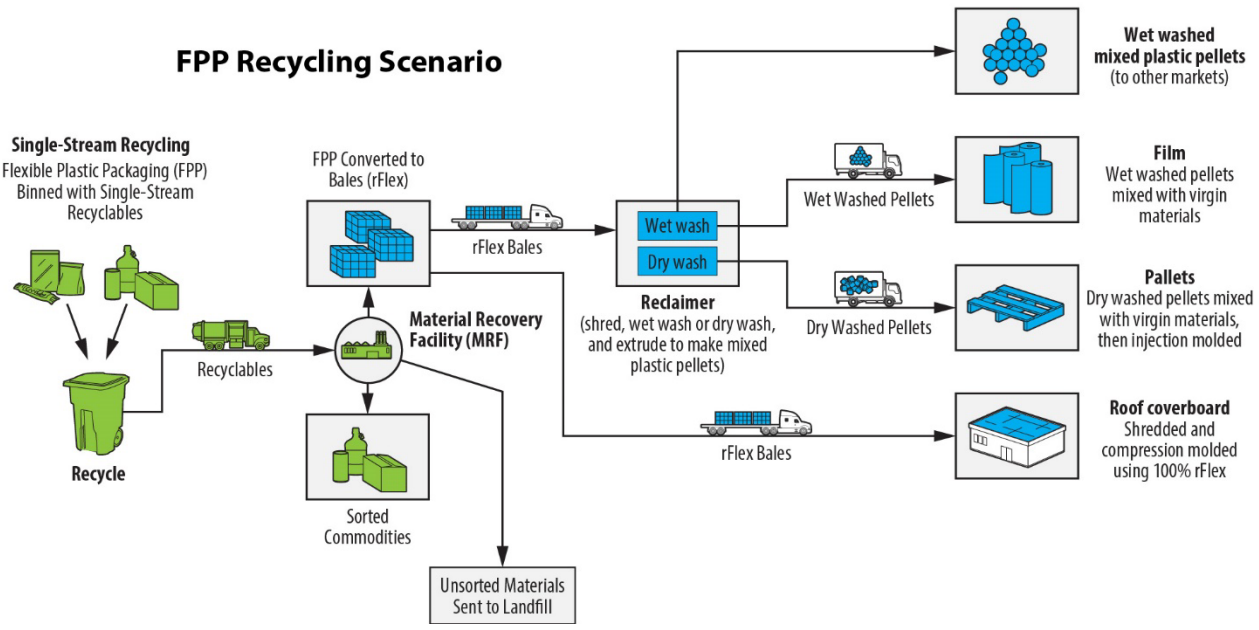

Figure 1: System boundary of this study. There are three stages included in the evaluated recycling scenario, including stage 1: waste transportation to MRF and sorting in MRF, stage 2: rFlex delivery to reclaimer for cleaning and pelletizing, and stage 3: FPP pellets delivery to converters and end markets.

2.2 Market pathways for rFlex

The choice of the four market pathways (roof coverboard, plastic pallet, wet-washed plastic pellets and plastic film) was the result of a previous effort conducted by the RRS Materials Recovery for the Future (MRFF) research team. Due to the Covid-19 pandemic, three of the four end markets were tested, ranging from those that could utilize the entire rFlex bale to those that could only use the plastic fraction of the bale known as the material substitution ratio (Anshassi & Townsend, 2021).

The first pathway (roof coverboard) was to shred and mold rFlex using compression molding into sheets of roof coverboard (pathway 2-A in Figure 2). In this process, the yield of recyclable material from the bale was over 95%. Roof coverboard from rFlex has been used in commercial roof installations. The successful testing, commercial application of rFlex, and low level of post-processing established roof coverboard as the first pathway.

The second pathway (plastic pallet) was to shred, dry wash, extrude to pellets, and use in molded pallets (include pathways 2-B-1 and 3-B-1 in Figure 2). The Covid-19 pandemic prevented the successful completion of this test. However, the intent was to use a small percentage of dry-washed rFlex pellets in compression or injection mold for durable applications like plastic pallets or crates. Plastic reclamation experts were convened in March 2020 to help determine the most promising applications and end markets. They had high confidence in the technical and economic feasibility of this pathway if a 70% yield of usable plastic could be achieved from the rFlex bale. Given the scale and the attractiveness of the plastic pallet market, an evaluation of this pathway was desired.

The third pathway (wet-washed plastic pellets) was to shred, wet wash, and extrude using filtration extrusion to make wet washed rFlex pellets (pathways 2-B-2 in Figure 2). Because recycled pellets could be a commodity and used in multiple applications, it was chosen with a potential to be used in a wide variety of markets beyond film.

The fourth pathway (plastic film) was to utilize the wet washed rFlex pellets in blown film (pathways 3-B-2 in Figure 2). Using a relatively low percentage of rFlex pellets, a film manufacturer successfully blew a low-quality film using the wet-washed rFlex pellets that could be used as a slip sheet or in heavier gauge applications. This test represented the highest degree of post-processing and is the most theoretical. However, as it represented an extreme case, it was selected as one of the pathways to assess.

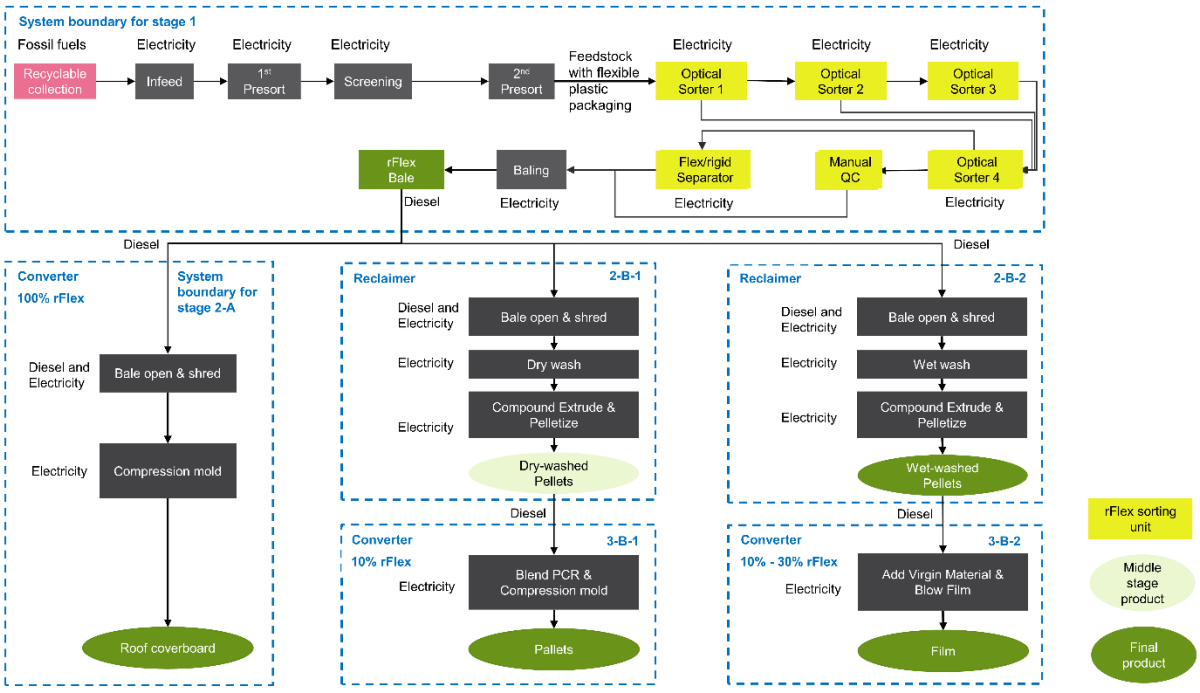

Figure 2: Process flow diagram describing rFlex production (stage 1) to product manufacturing processes (stage 2: 2-A, 2-B-1, 2-B-2, stage 3: 3-B-1 and 3-B-2). TEA and LCA were conducted to evaluate cost and footprints of all unit operations presented in the process flow diagram.

2.3 LCA methodology

In MRF sortation, the volume of processed materials impacted efficiency and throughput within each process. Because FPP was light weight but voluminous, processing this material might increase energy consumption and decrease sorting productivity. Ideally, the functional unit of this LCA should be based on volume to reflect the impact of low weight, high volume FPP on the rFlex footprint. However, due to difficulty in measuring input and output material on a volume basis for each piece of equipment, we chose to calculate rFlex footprint on a processed weight basis. Further discussion of the difference in methodology is provided in the supporting document.

The functional unit for Stage 1 was one tonne of rFlex produced at the MRF. For Stage 2 (reclaimer) and 3 (converter/end market), the functional unit was one unit of product produced. For example, in reference to Figure 2, the functional unit was one piece of board for the roof coverboard pathway (2-A), or one tonne of pellets produced for the dry- (2-B-1) and wet-washed plastic pellets (2-B-2) pathways.

2.3.1 Life Cycle Inventory (LCI) Analysis

In this section, an LCI analysis was conducted to document and quantify the total energy usage and emissions generated in each stage presented in Fig. 2. Data sources for the LCI included (1) actual data collected from a MRF (Resource Recycling Systems (RRS), 2020), (2) the Greenhouse gases, Regulated Emissions and Energy use in Transportation (GREET®) model developed by the Argonne National Laboratory (Wang et al., 2021) and (3) data from similar operations in plastic manufacturing industries. The transportation LCI for waste collection, MRF to landfill, and MRF to end market or manufacturing facility were summarized in Table S1.

For Stage 1 (rFlex production), processing units and their energy consumption were summarized and presented in Table S2 in the supporting socument. The energy consumption per tonne of rFlex produced, e_{rFlex} , was estimated as shown in Equation 1. To provide a conservative estimate of energy consumption at the MRF and downstream producers, a utilization factor of 100% was applied to the energy rating of all equipment. The energy to produce rFlex for each equipment was calculated by multiplying total energy use per equipment by the flowrate share of FPP. Energy of all equipment involved in FPP handling were totaled. Finally, this total energy was divided by total rFlex mass output to obtain energy consumption per tonne of rFlex produced.

$$e_{rFlex} = \frac{\sum_{i=1}^{n} E_{rFlex\_i}}{F_{rFlex\_total}} = \frac{\sum_{i=1}^{n} (E_i * \frac{F_{rFlex\_i}}{F_{total\_i}})}{F_{rFlex\_total}} \quad \text{Equation 1}$$

Where:

E_{rFlex_i} is the energy to produce rFlex of equipment *i* (kWh)
n is the number of pieces of equipment involved in rFlex production
F_{rFlex_total} is the total rFlex output of the entire process per hour (tonnes/hr)
E_i is the energy consumption of equipment *i* per hour (kWh)

F_{rFlex_i} is the flowrate of rFlex or FPP being processed by *i*

F_{total_i} is the total flowrate (tonnes/hr) of all material being processed by *i*.

For example, if the energy consumption E_i of a piece of equipment *i* is 10 kWh per productive hour, the flowrate of rFlex produced by this process F_{rFlex_i} is 4 tonne per productive hour, and the total flowrate of all materials F_{total_i} going through this machine is 20 tonnes per productive hour, then the energy to produce rFlex of that equipment is 2 kWh. Once the energy of all equipment involved in rFlex sorting and production is totaled, the unit energy of 1 tonne of rFlex can be derived by dividing that by total rFlex output.

2.3.2 Manufacturing process and market comparator for the four selected market pathways

The process of 2-A pathway (roof coverboard) was shown in Figure 2 and the LCI was presented in Table S3, including material input and energy consumption during transportation and the manufacturing process. The selected market comparator was drywall (gypsum board), which is made by calcium sulfate ($CaSO_4 \cdot 2H_2O$) and is the most common cover boards used in the market.

Table S4 and S6 show the LCI for the 2-B-1 and 3-B-1 (plastic pallet) pathway using 10% rFlex and 90% HDPE. The hourly productivity for this pathway was assumed to be about 30 pieces at 85% online time (PalletMach, 2022). In this LCA, we selected plastic pallets made of 100% virgin HDPE resins as the market comparator to compare the footprints with the plastic pallet made of 90% virgin HDPE and 10% rFlex.

The LCI data collected for the 2-B-2 (rFlex wet-washed pellets) pathway was presented in Table S5. The market comparator was chosen to be LDPE plastic pellets made from recycled LDPE pellets. As we do not have LDPE recycling in our database, we used HDPE recycling as an approximate (Cappucci et al., 2020). GHG emissions of producing HDPE and LDPE resins using virgin plastic pellets were referenced from the GREET database (Wang et al., 2021). It was noted that in both the 2-B-1 and 2-B-2 pathways, the fiber and contaminants in rFlex (about 40%) were removed during the washing processes and accounted as yield loss. Therefore, the landfill of residues collected from the washing step was also included in the total footprint of wet-washed pellet.

In the plastic film pathway (3-B-2), all the processing steps of pathway 2-B-2 were included, with an additional step of blown film extrusion (LCI presented in Table S7). The current design was sized to produce up to 1 tonne of film with 30% rFlex and 70% LDPE material (CharterNexGen, 2021). In this pathway, the market comparator was heavy gauge film such as agricultural films or slip sheets used to separate goods on a pallet.

2.4 TEA methodology

The TEA was divided into two separate sections: (1) the MRF rFlex production and (2) the downstream pathways utilizing rFlex. The described methodologies required analysis of three main expenditure categories: (1) capital expenditures (CAPEX) of the production equipment, (2) CAPEX of the facility, and (3) operating expenditures (OPEX). Additional details can be found in the supporting document.

2.4.1 MRF rFlex Production TEA

A cost allocation system was developed to accurately attribute the costs of production to each category of materials. A variety of input data were required including equipment price, recycled commodity

selling prices, and composition of products through each piece of equipment. Each machine's operating and capital expenditures were allocated to a particular product by the mass composition of the material moving through the machine. For example, if the composition of rFlex moving through a machine was 35% by mass, 35% of the total CAPEX and OPEX would be allocated to rFlex. This process was completed for every piece of equipment and every product that was input into the system. Following the calculation of allocated total CAPEX and OPEX for rFlex, a summation for all annual costs related to rFlex production was derived. This total cost was divided by rFlex annual output quantity to achieve a unit output cost in USD/tonne of rFlex.

2.4.2 Downstream Pathways TEA

Although the 2-A pathway can consume rFlex directly instead of washed or pelletized rFlex, the methodology for all four pathways was identical with the only difference being the price of the input feedstock. To calculate the CAPEX, all the process units were assumed to need a 30,000 square-foot manufacturing facility. The cost breakdown of the facility can be seen in Table S8 (Wisconsin Public Service, 2017). Regarding OPEX, each process unit required a series of yearly, operating costs to be calculated. Feedstock costs for roof coverboard process and plastic reclaiming were assumed to be $20/ton and $10/ton, respectively. These values were independent from the rFlex production cost from the MRF TEA to reflect current market willingness to pay. Tables S8 – S13 in the supporting document present detailed information and assumptions for roof coverboard, dry-washed pellets, pallet, wet-washed pellets, and film pathways, respectively. An assumed profit margin of 25% was utilized to convert from rFlex product cost to price. Direct competitor prices were reported as a calculated and a literature reported value. The calculated value was determined by incorporating 100% of the competitor feedstock into the TEA calculations for the rFlex product cost.

2.5 Sensitivity analysis

To reveal the impact of important factors on the GHG emissions and economics related to rFlex production, a sensitivity analysis was conducted. The sensitivity analysis for TEA evaluated impact of rFlex bales production cost on the price of downstream rFlex products. For LCA, the analysis considered parameters include the energy consumptions of the three main contributors/pieces of equipment to the total GHG emissions. The sensitivity is calculated as:

$$Sensitivity = \frac{Change\ in\ output/output}{Change\ in\ input/input} \qquad \text{Equation 2}$$

3.0 Results and Discussions

3.1 GHG footprints and Economics of recycling FPP to produce rFlex

Figure 3 shows the GHG emissions produced in each stage in rFlex MRF production, starting from waste collection, ending at MRF to landfill stage. In the process, for every tonne of recyclables processed, 3.24 kg of CO_2-eq was generated. For every tonne of rFlex produced, 55.06 kg of CO_2-eq was generated, with the *Flex/rigid separator* being the largest footprint contributor (30%), followed by the *Optical sorters (optical sorter 1, 2, 3 and 4)* (28%) and the *Air conveyance* (24%). The sensitivity of the total GHG emissions to the energy consumptions of these three processes were calculated as 0.30, 0.28 and 0.23. Sensitivity analysis results suggested that these three processes had linear impacts on the total GHG

emissions (as shown in Figure 3). More specifically, when energy consumption of the *Flex/rigid separator* increased or decreased by 50%, the total GHG emissions decreased or increased by 8.23 kg of CO_2 (15%). Compared to *Flex/rigid separator*, the impacts of *Air conveyance* and *Optical sorter unit* were slightly smaller. When energy consumption increased or decreased by 50%, the total GHG emissions varied in the range of 46.83 to 63.29 kg of CO_2 (±14%) and 48.54 to 61.58 kg of CO_2 (±12%), respectively.

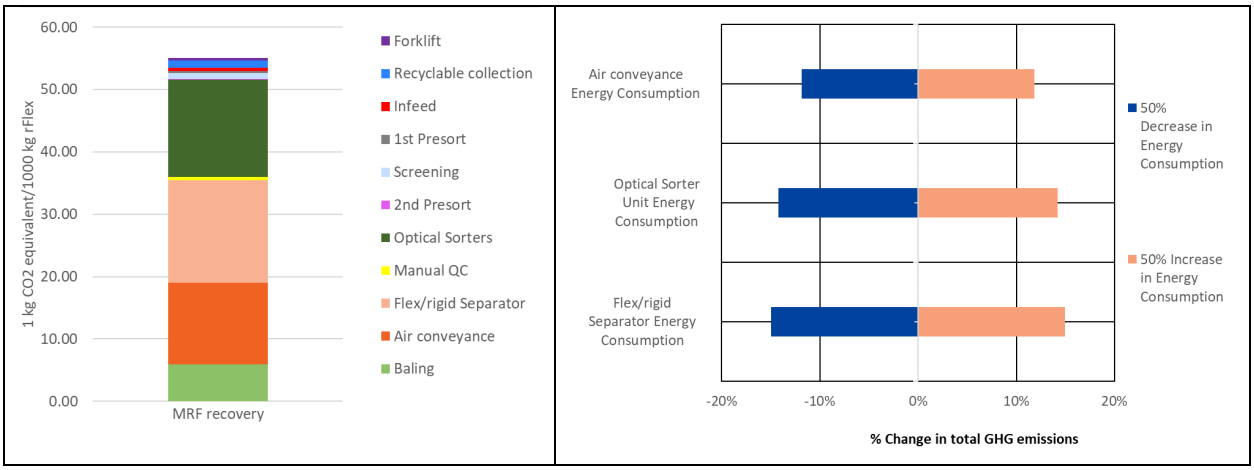

Figure 3: GHG emissions of rFlex recovery: from waste collection to rFlex baling. The figure presented the contribution of each modeled operation to the total footprints of rFlex recovery and the change in total environmental impact in response to altering the energy consumption of the three largest footprint contributors by ±50% (flex/rigid separator, optical sorter, and air conveyance).

Recycling plastic has been largely reported to have lower environmental impacts than landfilling or incineration, mainly due to the GHG emissions reductions from avoided production of new virgin plastic (Civancik-Uslu et al., 2021; Hou et al., 2018). For each kg of recyclables processed, the emissions were generally reported to be in a range from 0.011 to 0.015 kg CO_2-eq (Combs, 2011; Fitzgerald et al., 2012). Higher emission range of 0.026 to 0.042 kg CO_2 eq./kg recyclables processed have also been reported in other post-consumer plastic recycling systems (Basuhi et al., 2021). Our calculations of ~0.009 kg CO_2-eq per kg of recyclables processed was aligned with the first range. Regarding the footprints of rFlex production, because this is the first study to investigate FPP recycling, there is no literature value to compare with. In the future, to minimize the footprints of rFlex production, the R&D efforts should focus on either reducing the energy consumption or increasing the hourly throughput of rFlex in the flex/rigid separator, air conveyance, and the optical sorter.

The results of the MRF TEA showed that rFlex production cost was $82.45 USD per tonne of rFlex. This cost also represented the price that rFlex would need to be sold at to break even. A breakdown of this cost was: (1) operational cost, $40.07 USD/kg or 48.6%, (2) equipment capital cost, $40.73 USD/kg or 49.4%, and (3) facility capital cost, $1.65 USD/kg or 2.0%, with the allocated operational and equipment capital costs being the biggest contributors to production cost. It should be noted that equipment cost was annualized over an assumed period of ten years, while facility cost was annualized over a period of thirty-nine years (TaxAct, 2022). Additionally, insurance, taxes, and maintenance were included as part of the operational cost.

3.2 GHG footprints of downstream pathways and their sensitivities to rFlex compositions

Table 1 presented comparisons of GHG emissions for the four products that used virgin plastic and different ratios of rFlex. In the 2-A pathway, manufacturing one piece of roof coverboard using 100%

rFlex produces about 4.50 kg of CO_2-eq, of which ~ 64% GHG emissions came from the manufacturing process, ~ 34% of the emissions from MRF production stage, and 2% from the transportation stages. Compared to its market comparator (gypsum drywall), rFlex roof coverboard could reduce carbon footprints by 40% (2.96 kg CO_2-eq) per board.

Using 10% to 30% of dry-washed rFlex pellets substituted in plastic pallets in the 2-B-1 and 3-B-1 pathways produced about 44.28 to 38.40 kg CO_2-eq per piece of pallet, respectively. This meant a respective reduction in total GHG emissions of approximately 6% to 19% compared to pallets made from 100% virgin HDPE pellets (47.23 kg CO_2-eq per pallet). In the whole process, the largest contributor to the GHG emissions was the virgin HDPE resins blended with rFlex pellets, which ranged from 75% to 83%. The second largest contributor was the reclamation stage (2-B-1), contributing about 2% to 7%, when the rFlex substitution ratio was 10% to 30%, respectively.

Table 1. Summary of LCA results for rFlex products and market comparators. For all pathways, the rFlex product produces lower GHG emissions than the conventional/competitor product.

Pathway	Product unit	GHG emission (kg CO2-eq/product) (Weight-based allocation)	Comparator	Comparator's GHG emission (kg CO2-eq/product)	% GHG emission reduction
rFlex Roof coverboard (2-A) (100% rFlex)	1 board	4.50	**Gypsum drywall**	7.46	40%
Dry-washed rFlex pellets (2-B-1) (100% rFlex)	1 tonne pellets	525.42	**Recycled HDPE/LDPE resins**[1]	560.00	6%
Wet-washed rFlex pellets (2-B-2) (100% rFlex)	1 tonne pellets	469.97			16%
rFlex Pallet (3-B-1) (10% rFlex + 90% virgin HDPE resins)	1 pallet	44.28	**Pallet** (100% virgin HDPE resins)	47.23	6%
rFlex Film (3-B-2) (30% rFlex + 100% virgin LDPE resins)	1 tonne	1801.05	**Film** (100% virgin LDPE resins)	2337.67	23%

Note: [1] As we do not have LDPE recycling in our database, we used HDPE recycling as an approximate.

In the 2-B-2 pathway, the production of 1 tonne wet washed rFlex pellets generated about 469.97 kg of CO_2-eq, which was 75%, 79% and 16% less than the production of virgin HDPE, LDPE pellets and recycled HDPE/LDPE resins, respectively. During the rFlex pellet production process, operations at the reclamation facility such as wet washing and drying, and pelletizing accounted for 22% and 32% of the total footprints. MRF recovery contributed about 19% to the total footprint, other processes such as rFlex transportation only contributed 5% to the total emissions.

In the 3-B-2 pathway, the wet washed rFlex pellets were blended with virgin LDPE resins using different substitution ratios (10% -30% of rFlex) to produce rFlex film. The production of 1 tonne rFlex film with 10% to 30% rFlex pellets generated 2158.80 to 1801.05 kg CO_2-eq, respectively. Compared to plastic film that used 100% virgin material, rFlex film has 8% to 23% lower total GHG emissions corresponding to 10% and 30% rFlex content. When using different rFlex blend ratios, the largest contributor was virgin

LDPE resins with about 86% to 92% of the total emissions, while the reclamation stage was the second largest contributor with GHG emissions of about 2% to 6%, respectively.

In all of the investigated pathways, the 2-A pathway (roof coverboard) was the most receptive pathway in terms of GHG emissions reduction (42%) compared to its market comparator. The second receptive pathway was the 3-B-2 pathway (rFlex film) that reduced total emissions by 23%. Although the 2-B-2 pathway (wet washed rFlex pellets) reduced the GHG emissions by 75% to 79% compared to virgin HDPE and LDPE pellets, when compared to recycled plastic resins that has comparable quality, it only reduced about 16% of GHG emissions. In the 3-B-1 pathway (rFlex pallet), due to the limited substitution ratios (10%) of rFlex, the GHG emissions reductions compared to market comparator was only 6%.

As discussed above, recycled rFlex had much lower carbon footprints compared to both recycled and virgin plastic. Therefore, the investigated market pathways with the highest rFlex blend ratios (i.e. the 2-A and the 3-B-2 pathways) reduced the GHG emissions the most compared to their market comparators (Gu, Guo, Zhang, Summers, & Hall, 2017). In pathways that already uses 100% rFlex, the decontamination process played the most important role in the total the environmental performance, which had also been reported by another study (Civancik-Uslu et al., 2021).

3.3 Economics of downstream pathways and a sensitivity analysis of rFlex composition

The economic results of the downstream pathways can be seen in Table 2. Each downstream rFlex product was compared to its most direct competitor by computing the cost of the rFlex product and applying an assumed 25% profit margin to the product. Both a published literature and a calculated price were reported for the direct competitor for the rFlex product. As seen in the table, rFlex products were all cheaper than their competitors. A discussion on the price variance of film can be found in the supporting document.

Table 2: Results of downstream pathways TEA showing rFlex pathway production costs compared to its competitors. For all pathways, the rFlex product was cheaper to produce than the conventional/competitor product.

rFlex Downstream Product	Unit	rFlex Product Price (Calculated)	Competitor	Competitor Product Price (Calculated)	Competitor Product Price (Literature)	% Price Reduction
rFlex Roof coverboard (2-A) (100% rFlex)	USD /piece	$13.34	**Gypsum Drywall**	----	$24.65	46%
Dry-washed rFlex pellets (2-B-1) (100% rFlex)	USD /tonne	$456.75	**Recycled cHDPE Pellet**	----	$1,071.45	57%
Wet-washed rFlex pellets (2-B-2) (100% rFlex)	USD /tonne	$660.12	**LDPE Pellet**	----	$1,763.20	63%
rFlex Pallet (3-B-1) (10% rFlex + 90% virgin HDPE resins)	USD /pallet	$33.42	**Recycled cHDPE Pellet**	$35.23	$34.65	3.5%
rFlex Film (3-B-2) (30% rFlex + 100% virgin LDPE resins)	USD /tonne	$2,133.12	**LDPE Film**	$2,574.35	$3,303.44	35%

Table 2 shows that the rFlex products were cheaper to produce than their competitors in all pathways. This was largely attributed to the dramatically lower feedstock cost of utilizing rFlex versus competitors

such as recycled cHDPE or LDPE. In the instance of rFlex pellets (dry-washed or wet-washed) and roof coverboard, the feedstock price was the selling price of an rFlex bale. The pallets and film received their feedstock from rFlex pellets dependent on the initial rFlex selling price per bale. When a sensitivity analysis on the rFlex bale price was performed as shown in Figure 4, it was seen that roof coverboard was the most sensitive to feedstock price change. This can be explained by the fact that it was the cheapest product and relied on 100% rFlex. The least dependent products were pallets and film because the rFlex feedstock cost was inconsequential compared to the OPEX and CAPEX required to produce these two products.

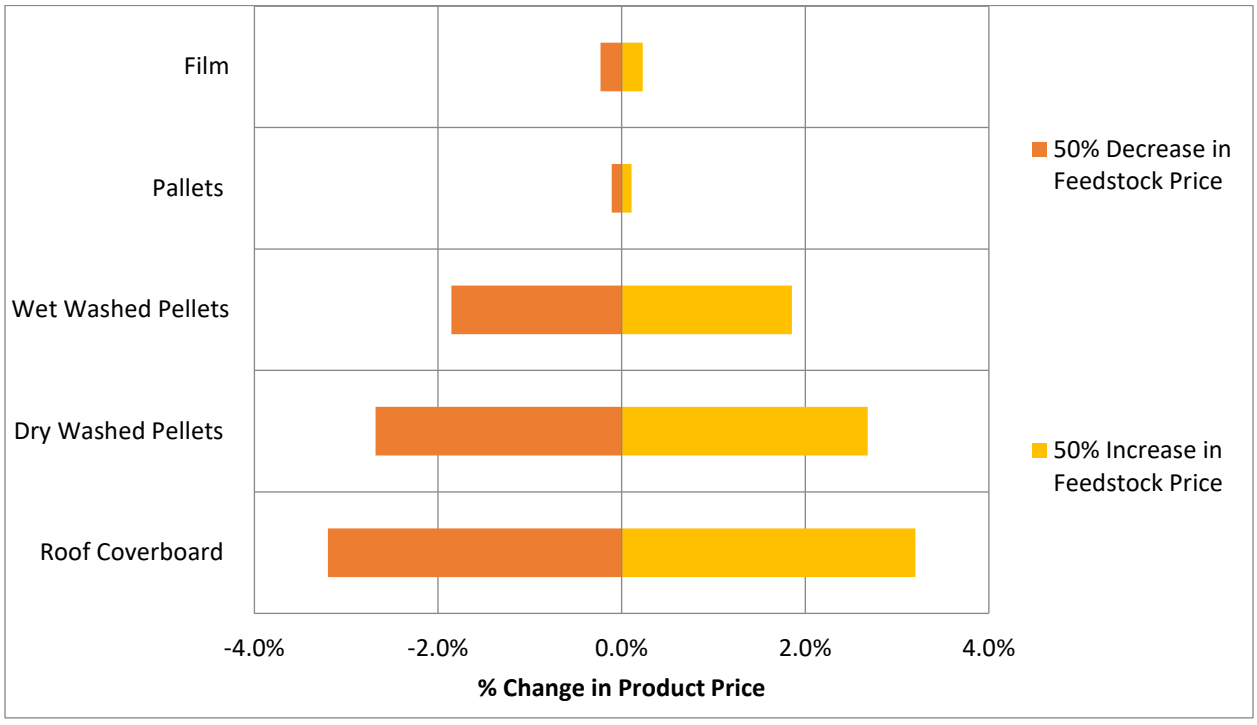

Figure 4: Sensitivity analysis of feedstock rFlex bale price (± 50%) on the price of downstream rFlex products. Roof coverboard saw the highest sensitivity to rFlex bale price due to the minimal amount of processing steps required to produce the products which placed the larger relative cost on the feedstock.

An additional sensitivity analysis was performed for the quantity of rFlex contained within two of the downstream pathways: pallets and film. The base case scenario assumed that 10% of rFlex could be substituted into the pallet feedstock, whereas 30% of rFlex could be substituted into the film feedstock. Figure S1 (supporting document) shows that with increased rFlex content in the feedstock, the overall price of the downstream product will decrease. At 50% rFlex, pallets were approximately $26 USD per pallet, and film was approximately $1,839 USD per tonne ($1,668 USD per ton).

3.4 Impacts of the pilot study on stakeholders and communities and ways to improve economic competitiveness for FPP recycling

In the case of this pilot, producers in the FPP value chain have shown interest in creating demand for rFlex; as one example, one of the companies involved in the pilot program used rFlex-based roof coverboard for a warehouse facility (Megan Smalley, 2021). The Association of Plastic Recyclers Demand Champions program is an industry-wide effort to drive demand for recycled-content products and thereby make plastic recycling more profitable for MRFs and re-processors (The Association of Plastic

Recyclers, 2021). In addition, residents expressed a high level of interest in the program and the pilot MRF received positive feedback from residents and community staff for adding FPP as an accepted item (Resource Recycling Systems (RRS), 2020).

As shown above, the cost of FPP recycling was higher than its revenue. In considering the possibility of adding FPP collection to residential recycling more broadly, there are numerous approaches to how these added costs might be allocated amongst the various stakeholders in the recycling system. Ultimately the factors influencing this allocation include policy, consumer preference, and relative market power of various stakeholders. One approach looks at increasing the MRF processing fee to account for the additional costs. At the time of this study, most MRFs in the U.S. charged a processing fee between $42 and $88/tonne for single stream recyclables, with the caveat that this has continued to increase in the past several years (APTIM, 2019; Chittenden Solid Waste District, 2022; Paben, 2019). If MRFs begin to collect FPP as part of single stream, an increase in this processing fee by $1.99/tonne would cover the full additional cost of sorting FPP. The minimum additional processing fee per tonne of single stream is calculated by subtracting annual rFlex cost by annual rFlex revenue, then divided by annual throughput quantity. The willingness of residents and communities to bear all or part of this additional cost for the ability to recycle additional material is not tested in this study and is in practice likely to vary among communities and regions.

The second hypothetical approach is to apply a carbon credit for rFlex recycled products. This carbon credit could be traded on a market. Companies who need to reduce their carbon footprint can purchase this credit and provide a financial resource for FPP recycling. While there is not currently policy driving carbon credit markets at a national scale in the US, there are voluntary carbon credit markets such as the California cap-and-trade program, Nori carbon market, and Ecosystem Services market consortium price carbon credits at $15 USD/tonne CO_2 (Mishra, Gautam, Mishra, & Scown, 2021). In addition, Verra has a Plastic Waste Reduction Program that provides standard and audits/assessment that enable recycling industry to earn Waste Recycling Credits from collecting and recycling plastic (Verra, 2022).

A third hypothetical approach is based on a “green premium” concept. At the brand or consumer level, if customers are willing to pay more for products with recycled content from FPP, the willingness to pay for rFlex bales will increase and boost revenue for the MRF. The levels of green-premium largely depends on demand and supply of specific low-CO_2 products. For example, an article published by McKinsey projects that the global supply of low-CO_2 high-quality recycled plastics could increase to 41 million tonnes by 2030, some 56 million tonnes less than the projected demand of 97 million tonnes (Marcelo Azevedo, 2022). This supply-demand balance will largely increase the green premium for high-quality recycled plastics. While none of these approaches are surefire solutions for overcoming the complex market barriers to adding FPP as a collected recyclable, they could ultimately all be used in part given the right set of policy and demand drivers.

3.5 Limitations and suggestions for future work

3.5.1 Disconnect between MRF TEA & downstream rFlex cost

The TEA conducted for the MRF output a cost to produce rFlex of $82.45 USD per tonne ($74.80 USD/ton). However, the selling price for rFlex utilized in downstream TEAs range from $11.02 to 22.04 USD per tonne ($10 to $20 USD/ton). Many product streams recycled by the MRF do not always generate a revenue higher than its processing cost. This is why MRFs incorporate a processing fee to take in recyclable materials as a way to increase revenue and create a positive profit business.

3.5.2 Future Work

Based on this study, a system dynamics model with a user interface could be developed to quantify the impact of FPP recycling across the U.S. using the rFlex bale as the example. This model would be capable of evaluating the four product pathways compared to the base case of landfilling. By incorporating recycling economics, rFlex production capacity expands depending on the target market adoption. Additionally, by increasing the demand for rFlex, a subsequent increase in price would be seen which would further improve the economics of the recycling process. The model would offer scenario analysis to improve recycling economics.

4.0 Conclusions

This study reports a pilot scale effort in the U.S. to recycle flexible plastic packaging (FPP) from curbside collection of single stream recyclables, focusing on both the economic feasibility and GHG emissions based on collected data from a commercial material recovery facility (MRF). To further explore the marketability of recycled FPP, four market pathways were studied, including roof coverboard, dry washed plastic pellets to pallets, wet washed plastic pellets, and wet wash plastic pellets to film. Results indicated that emissions of recycled FPP products in all four pathways are better than those of the comparator products, and higher blends of recycled FPP in products result in reduced GHG emission. Results also suggested that the dry washing and pelletizing process contributed the most to the total GHG emissions in reclamation. From an economic standpoint, rFlex cost approximately $82.45 USD per tonne to produce when FPP sorting equipment was added via an upgrade. Of the studied pathways, all were cost competitive with their closest proxy competitor.

Given the rapid growth of flexible plastic packaging in the marketplace, as described in the introduction, the findings of this study are relevant to a wide range of stakeholders, including policymakers, waste service providers, and members of the plastic packaging value chain (Anshassi, Laux, & Townsend, 2019). This is the first pilot project in the US that attempts to capture the full range of FPP, both multi-resin and polyethylene-based, via curbside collection mixed with all other household packaging. A sustainable materials management (SMM) approach, which focuses on net environmental outcomes through the materials management chain, takes a positive view towards the use of less material-intensive packaging, of which FPP is an example (Anshassi et al., 2019). Under a sustainable materials management approach, recycling FPP would be preferable to landfilling the material in situations where this leads to lower overall GHG emissions. As shown in this study, those lower emissions are possible in each of the four market pathways in contrast to comparator products they would displace, thus showing the favorable potential for recycling FPP under policy frameworks incorporating SMM principles. As the recycling system in the U.S. continues to evolve, with extended producer responsibility (EPR) being put into place in a number of states, the findings from this study can be useful in evaluating FPP as a packaging choice and the best end-of-life option for this material.

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:

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