

# The hidden economic and environmental costs of eliminating kerb-side recycling

Received: 20 October 2022

Accepted: 18 April 2023

Published online: 22 May 2023

 Check for updates


Malak Anshassi <sup>1</sup> & Timothy G. Townsend <sup>2</sup> 

Local governments provide household collection of garbage and recyclables on a routine schedule, and these recycling programmes represent the most visible opportunity for everyday citizens to engage in sustainable practices. In the face of unprecedented challenges, and citing costs as the major driver, many US communities are shrinking or eliminating kerb-side recycling. Here we show that when recycling commodity markets were most lucrative in 2011, net US recycling costs were as little as US\$3 per household annually, and when markets reached a minimum (in 2018–2020), the annual recycling-programme costs ranged from US\$34 to US\$42 per household. This investment offsets the greenhouse gas emissions from non-recycled household waste buried in landfills. If local governments restructure recycling programmes to target higher value and embodied carbon-intensive materials, recycling can pay for itself and reduce greenhouse gas emissions. Our analysis highlights that kerb-side recycling provides communities a return on investment similar to or better than climate change mitigation strategies such as voluntary green power purchases and transitioning to electric vehicles. Eliminating recycling squanders one of the easiest opportunities for communities and citizens to mitigate climate change and reduce natural resources demands.

Perhaps the most visible opportunity local governments provide citizens to engage in environmentally sustainable behaviour is participation in a recycling programme<sup>1–4</sup>. Residents separate recyclable products and packaging from other discards and place them in different containers for collection and transport to a materials recovery facility (MRF). At the MRF, mechanical equipment and hand sorting extract target materials for marketing as secondary feedstocks in new products. In recent years, local governments providing residential recycling in the United States have faced challenges unseen since their inception in the 1980s and 1990s<sup>5,6</sup> and many are reassessing the benefit of providing recycling services. Materials historically placed in the kerb-side recycling bin or cart, such as glass and mixed plastics, are increasingly no longer accepted, and some communities have eliminated residential recycling collection altogether<sup>7</sup>. These actions respond to multiple drivers, including restrictions on international recycling markets<sup>8–10</sup>, contamination in the recycling stream<sup>11,12</sup> and challenges from the COVID-19

pandemic<sup>13,14</sup>, but the overriding factor has been cost. A common refrain by government officials is that recycling is too costly compared with collecting only garbage for disposal. It is critical to assess how much more expensive recycling truly is and whether at any time, historically, the resale value of recyclables was sufficiently lucrative for a recycling programme to pay for itself.

The troubled times experienced by the recycling industry coincide with greater mainstream recognition of the relationship between our waste and the global environment. The effects of climate change become more visible every year, and targeting improvements in waste management has become a favoured mitigation strategy<sup>15–18</sup>, as the waste sector is more easily influenced through government action and because recycling reduces upstream carbon emissions from avoided materials extraction and consumption<sup>19–21</sup>. Images of plastic-strewn shorelines, islands of ocean plastics and plastic-entangled wildlife<sup>22,23</sup>, however, have done more to raise public awareness of waste-stream

<sup>1</sup>Department of Environmental Engineering, Florida Polytechnic University, Lakeland, FL, USA. <sup>2</sup>Department of Environmental Engineering Sciences, University of Florida, Gainesville, FL, USA.  e-mail: [ttown@ufl.edu](mailto:ttown@ufl.edu)

practices than anything else. The challenged recycling enterprise and the important relationship of waste and our environment require thoughtful examination of the true costs and benefits of current waste-collection and disposition practices, especially the potential environmental benefits from recycling. The role that residential recycling plays in mitigating climate change is often overlooked and not included in numerous climate policy and consumption studies (refs. 24–27), yet evidence from waste-management life-cycle assessments (LCA) demonstrates recycling can avoid large magnitudes of greenhouse gas (GHG) emissions and reduce the need for virgin natural resources<sup>19,28,29</sup>. As global policies continue to target sustainable consumption and production (for example, Sustainable Development Goal 12, extended producer responsibility, circular economy), prioritizing recycling at the household level becomes a crucial strategy.

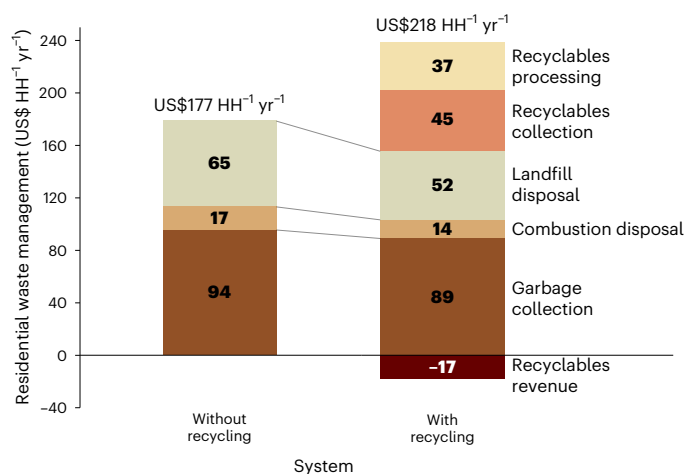
Here we develop a model to simulate the mass flows, costs and GHG emissions associated with a typical US household's waste management. We focus on the household level as this is the segment of the waste stream that local governments hold responsibility for. The model was developed in lieu of available environmentally extended input–output/multi-regional input–output models because there lacks a single existing model that focuses on the economic flows of a US household's waste-management system while combining their associated LCA-based GHG emissions. The robust data necessary to map the interconnected flows of products, environmental footprints and economic indicators for the waste-management sector are limited in many parts of the United States and across the world. Therefore, we employ a modelling framework to estimate the following: (1) the mass flows for the separate collection of garbage and recyclables, processing of recyclables at a sorting facility, sale of recovered recyclables for remanufacture, disposal of residuals from sorting facility and disposal of garbage; (2) the monthly household cost of waste management for seven US regions for a time series from April 2005 to June 2021; (3) the corresponding annual GHG emissions footprint associated with household waste management and (4) the mass, costs and GHG emissions impacts when the model was imposed with hypothetical changes to the recycling programme. For context, we contrast our results with the GHG emissions-reduction return on investment from other sustainable practices, namely purchasing green utility power at a higher rate and transitioning from gasoline to hybrid or electric vehicles. Finally, we explore how a community's recycling system can be adapted to become more cost effective while maintaining the desired conservation outcomes.

## Results

### Estimation of kerb-side recycling costs

The degree to which a recycling programme provides an economic boost or drain depends upon the prevailing market values for recovered commodities. Like any traded commodity, market values for recovered recyclables fluctuate in response to economic health, product demand and price and availability of competing resources<sup>30,31</sup>. The United States and much of the developed world historically relied on other nations as consumers of recycled commodities, and recent policy changes in receiving countries have dramatically disrupted markets and reduced commodity values<sup>32,33</sup>. Revenues from the sale of recovered commodities represent only one cost element of a recycling programme, as investments in separate collection containers, specialized collection vehicles and processing at the MRF are also necessary. We quantify the cost and GHG emissions (for a US household; HH) of a generalized residential garbage and recycling system using a modelling framework that assesses changing waste commodity prices over 15 years (from April 2005 to June 2021) (Methods).

We modelled net system costs of two waste-collection scenarios, one where everything placed on the kerb (of a household) is managed as garbage and one where 20% of this waste is collected separately for recycling; the average recycling rate for single-family households

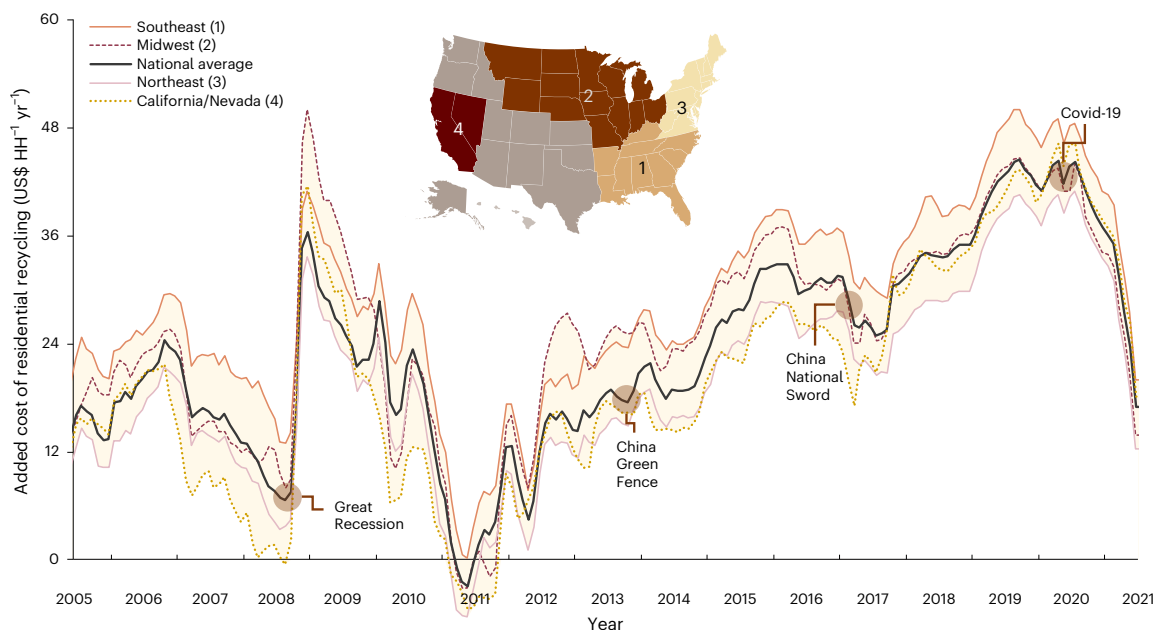


**Fig. 1 | Comparison of the annual net household costs associated with the waste management of an average US residential household in 2020 with and without recycling.** In the without recycling system, all recyclables were collected as garbage and disposed of through landfill or combustion facilities; they were not processed or sold. In the system with recycling, recyclables were separately collected, processed at a MRF and marketed using monthly 2020 US national average commodity prices. Summation of bolded values will not equal the total values on top of the bars due to rounding.

typically ranges from 15–30% (refs. 34,35). The net US residential waste-management costs when all kerb-side materials are managed as garbage comes to US\$178 HH<sup>-1</sup> yr<sup>-1</sup> (2020 recycling commodity-market values). When 20% of the materials (by mass) are collected separately for recycling, net costs increase to US\$218 HH<sup>-1</sup> yr<sup>-1</sup> (Fig. 1). As illustrated in Fig. 1, the greatest system costs result from materials collection (US\$89 HH<sup>-1</sup> yr<sup>-1</sup> for garbage collection and US\$45 HH<sup>-1</sup> yr<sup>-1</sup> for recyclables collection) and landfill disposal (US\$52 HH<sup>-1</sup> yr<sup>-1</sup>). The 2020 costs reflect a period of relatively poor commodity value, with the cost to process recyclables at the MRF (US\$115 ton<sup>-1</sup>) roughly twice the market value of the recyclables. The 2020 estimate suggests a US local government's burden to provide a recycling programme under challenging market conditions to be on the order of US\$40 HH<sup>-1</sup> yr<sup>-1</sup>, a value corroborated by existing recycling-programme financial data (Methods).

### Minimum historic cost of recycling across the United States

Annual household costs for kerb-side recycling services peaked when commodity prices were at their lowest after the 2008 Great Recession and in response to international market restrictions of the 2010s (Fig. 2). Our modelling utilizes not only peak recycling markets (for example, 2011) but also includes the worst commodity-market conditions (for example, 2019 and 2020) encountered over the past 15 years. In 2008, recycling costs averaged less than US\$14 HH<sup>-1</sup> yr<sup>-1</sup> but surged to over US\$26 HH<sup>-1</sup> yr<sup>-1</sup> when the full force of the recession hit; reports from this period cite container ships loaded with recycled US commodities turned back at Asian ports after raw material demand crashed<sup>30,36</sup>. Post-recession commodity values recovered by 2011 and were sufficiently high that recycling programmes in much of the United States paid for themselves. Policies enacted by the Chinese government to promote receipt of only the cleanest recycled-commodity imports, the 2013 Green Fence and the 2017 National Sword policies, contributed to reduced commodity prices over the past decade<sup>37</sup>. Recycling-programme costs peaked during 2019–2020 at about US\$42 HH<sup>-1</sup> yr<sup>-1</sup> at the highest, but increased market demand for mixed paper, cardboard and plastics occurring in the raw material-starved COVID-19 pandemic period dropped recycling-programme costs in 2021. More recent reports (September 2022) indicate recycling markets



**Fig. 2 | The costs associated with separately collecting, processing and marketing recyclables compared to the total costs of collecting and managing household waste for the United States and four regions.** The costs fluctuate from April 2005–June 2021 because of differences in regional data (for example, disposal fees, percentage of waste disposed of through landfill or combustion treatment, recyclables commodity prices). Regions are shaded for Southeast (region 1), Midwest (region 2), Northeast (region 3) and California/Nevada (region 4). Total costs included six waste-management economic parameters: recyclables and garbage collection, landfill and combustion disposal

and recyclables processing and revenue. Regional data did not impact collection costs. Recyclables-processing costs were assumed to be a constant US\$115 per ton of recyclables sorted at a recovery facility for all years. While recyclables commodity prices were month specific for each region and for the national average, each region's landfill and combustion disposal costs were assumed to be constant for all years. Results for two regions not included here are shown in the Supplementary Information. Highlighted here are four critical historic events, two of which relate to Chinese recycling policies.

have again fluctuated to a lower-value market, further illustrating the volatility of the market.

Our analysis finds that in nearly all cases, it has always been more expensive for local governments to provide kerb-side recycling than to collect and manage all household discards as garbage. The expense of collecting and separating bottles, cans and paper products exceeds the resale value of the recovered materials. Commodity prices, subject to change in response to the same stimuli as other traded goods and services, heavily influence net system cost. It is false to suggest that only in recent years has recycling been a cost to local government. The cost is arguably not large in comparison to overall waste-management system costs. Under the worst market conditions of the past 15 years, a waste-management system with recycling was 24% more expensive than collection for disposal only and under the best conditions, the system roughly paid for itself.

If recycling has always cost more than disposal alone, why have many local governments only recently begun to consider eliminating these services? This stems from recycling programmes having suffered through the longest duration of depressed commodity markets since their inception, a timespan extending beyond the typical contract cycle. MRF operators, needing to balance processing charges with their share of commodity sale prices, have renegotiated contracts to provide more stable pricing, placing the cost of recycling programmes in the eyes of local government decisionmakers more than ever. MRF operators report a growing degree of contamination in the recycling stream—non-marketable materials in the recycling bin requiring separation and disposal<sup>22</sup>—a result of several factors. In the United States, most kerb-side recycling programmes have migrated to single-stream collection, providing large carts for all recyclables, encouraging well-meaning residents to place anything they deem recyclable into the bin and at times to use the spacious containers as second garbage cans. The non-uniformity of materials accepted for recycling among

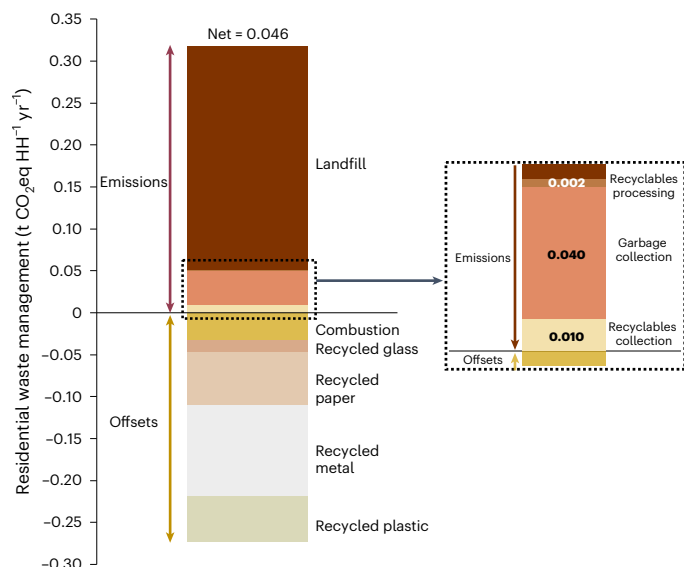
local programmes and the ever-increasing diversity of composite and light-weight packaging materials add to resident confusion as to what belongs in the recycling bin. Thus, when the international end markets tightened purity requirements, MRF operators found it necessary to remove even more contaminants, reducing the mass of materials destined for reuse and increasing the amount for disposal.

### GHG emissions reduction from recycling

Kerb-side recycling must be understood for what it is: a government service provided to residents, akin to providing energy and water, managing garbage and wastewater and maintaining roads and public spaces. Recycling comes at a cost, but more challenging for recycling compared with other community services is defining the benefit. Recognized benefits of recycling—reduced landfill reliance and resource conservation—can be difficult to meaningfully quantify, but methods for tracking recycling advantages such as GHG emissions reduction are more widely accepted in today's carbon-counting environment<sup>38</sup>. Other environmental benefits are realized when materials recovered through recycling programmes substitute virgin resources, including conserving abiotic and biotic resources, reducing water usage and nutrient loading and offsetting toxic emissions, but we examined only GHG emissions given their more robust life-cycle inventory data and more common use in materials sustainability assessments.

Using our modelled waste-management scenarios along with waste-management-focused LCA tools, we estimate that over the past 15 years, the average annual GHG emissions footprint associated with waste management for a US household was 0.046 metric tons of CO<sub>2</sub> equivalents (t CO<sub>2</sub>eq HH<sup>-1</sup> yr<sup>-1</sup>) (Fig. 3). Methane emissions from decomposing garbage in landfills contribute the most (0.27 t CO<sub>2</sub>eq HH<sup>-1</sup> yr<sup>-1</sup> based on a 100-year time horizon for global warming potential, and this value will increase if using a 20-year time horizon), but these are largely offset by recycling (approximately 0.24 t CO<sub>2</sub>eq HH<sup>-1</sup> yr<sup>-1</sup>,





**Fig. 3 | Annual GHG emissions associated with the waste management of an average US residential household in 2020.** The footprint is divided into nine waste-management processes and corresponds to a household with a 20% recycling rate. The area highlighted by a dotted line in the main bar chart is shown in detail in the inset.

counteracting nearly all emissions sources)<sup>39,40</sup>. On the basis of our results, recycling drives the household waste GHG emissions footprint to nearly neutral (0.046 t CO<sub>2</sub>eq HH<sup>-1</sup> yr<sup>-1</sup>), suggesting that investments in better capture technology for sorting and processing recyclables may provide even larger benefits.

Parameters incorporated in these life-cycle results include population density, energy grid mixture, landfill gas management, remanufacturing mechanisms (for example, amount of recycled material used in place of virgin sources) and material transport distances. Each of these parameters has an influence on the net footprint, and we evaluated the potential impact to each material type and its management through a Monte Carlo sensitivity analysis. The parameter assumptions used as part of our waste-management LCA originate from robust life-cycle inventories pertaining to the US recycling industry, landfills and municipal solid waste incinerators (Methods). For some parameters, such as remanufacturing mechanisms, the datasets have not been recently updated<sup>41</sup> and assume high substitution ratios (many nearing a one-to-one ratio, ref. 42), which may overestimate the climate change mitigation potential. In contrast, our results may also be an underestimate for certain materials given recent manufacturers' and national policy interest in increasing the recycled content of products (for example, plastics)<sup>43</sup>. Additional research is needed to update the remanufacturing life-cycle inventories to reflect current substitution ratios and to understand the potential impact on GHG emissions offsets when the recycled-content usage increases during new product manufacture. We attempted to consider these impacts as part of the recycling sensitivity analysis (Supplementary Information Section 2.7 provides the methodology and results).

A local government's investment in recycling provides GHG emissions reduction, but as most government officials do not routinely budget resources or make decisions based on metric tons of carbon dioxide equivalents, we provide additional context. Using the results above, we calculated an environmental return on investment (ROI) by normalizing reduction in GHG emissions associated with household recycling to corresponding cost. Under poor market conditions, such as those encountered in 2020, 0.0058 t CO<sub>2</sub>eq is offset for every US dollar spent on recycling, and under the best market conditions (for example, 2011), the ROI increases to 0.081 t CO<sub>2</sub>eq offset per US dollar.

To provide a comparison to the ROI achieved through recycling, we also estimated ROI for a household that: (1) replaces their internal combustion engine vehicles with either a hybrid electric vehicle or an electric vehicle and (2) switches from a traditional electricity grid utility to a green utility service.

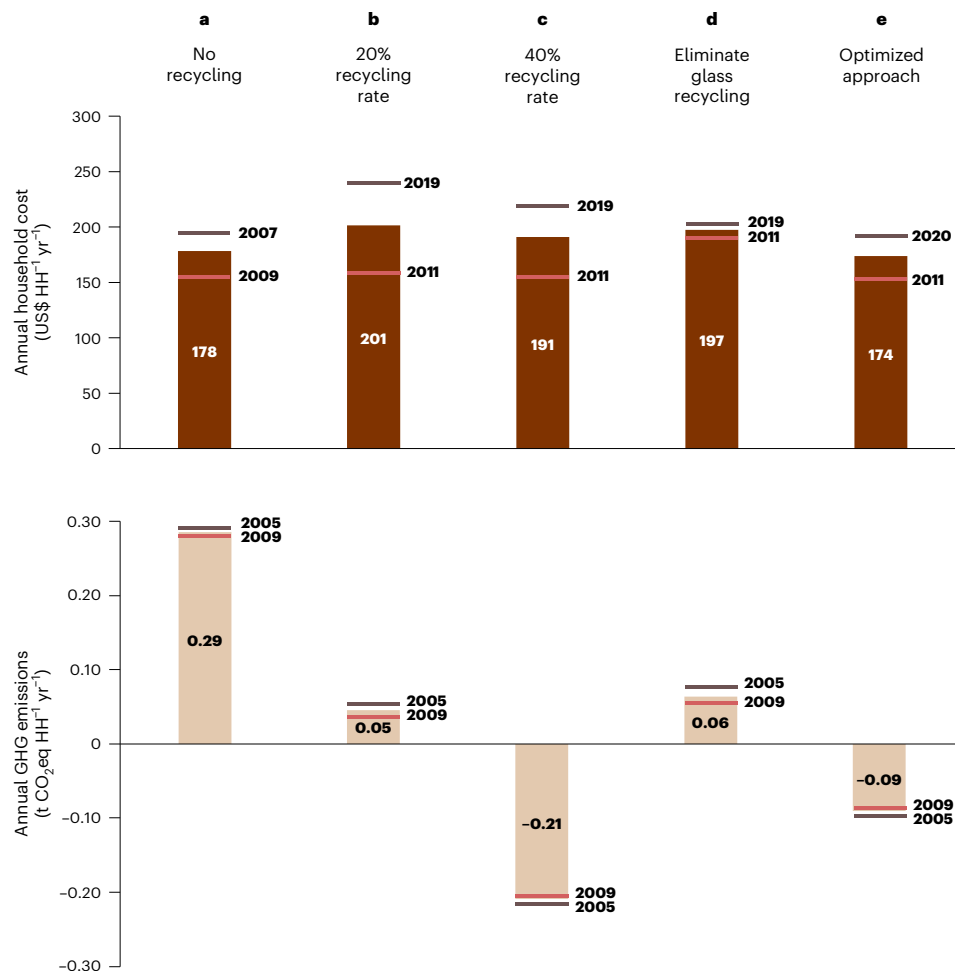
For the first ROI activity, we calculated for a household the average costs of purchasing and operating a vehicle and the average GHG emissions associated with the manufacture and operation of a vehicle. Data on the economic costs (in units of US\$ km<sup>-1</sup>) and the GHG emissions impacts (in units of g CO<sub>2</sub>eq km<sup>-1</sup>) were compiled from literature<sup>44–51</sup> to estimate an average economic cost and GHG emissions factor (Supplementary Table 20 and Methods). Then we estimated the average annual km travelled by car per household using data from the US Department of Transportation, which reports 1.88 vehicles per household<sup>52</sup> and data from the Annual Vehicle Functional System Travel report, which estimates 16,741 km per person (ref. 53) (Methods). Using those two sources, the total average annual km travelled by a household in the United States was estimated as 31,000 km HH<sup>-1</sup> yr<sup>-1</sup>. The economic cost and GHG emissions factors were applied to this value to estimate the ROI. The average ROI was calculated as ranging from an offset of 0.007 to 0.016 t CO<sub>2</sub>eq US\$<sup>-1</sup> spent switching from internal combustion engine vehicles to electric vehicles and for an offset of 0.0038 to 0.023 t CO<sub>2</sub>eq US\$<sup>-1</sup> spent switching from internal combustion engine vehicles to hybrid electric vehicles (Methods).

In the second ROI, we evaluated the benefits associated with a practice where some US utilities offer customers an option to pay an additional fee to receive electricity from predominantly renewable sources instead of the traditional fossil and non-fossil mixed sources. Using US Energy Information Administration data on the average monthly consumption (in kWh) and average price (cents kWh<sup>-1</sup>) for all 50 US states in 2020<sup>54</sup>, we calculated the average household annual costs (US\$963–1,952 HH<sup>-1</sup> yr<sup>-1</sup>), and using regional GHG emissions intensity factors from the US Environmental Protection Agency eGRID database (Supplementary Table 23 and Methods) for electricity generation, we estimated the average annual household GHG emissions (2,885–14,095 kg CO<sub>2</sub>eq HH<sup>-1</sup> yr<sup>-1</sup>). Assuming renewable energy will have no associated emissions (that is, zero GHG emissions intensity) and using the Energy Information Administration data, the environmental ROI ranged from an offset of 0.011 to 0.045 t CO<sub>2</sub>eq US\$<sup>-1</sup> spent, depending on location and provider costs (additional cost of 2.4–3.8 cents kWh<sup>-1</sup> for green energy).

The recycling environmental ROI is similar to or greater than both the ROI of switching to electric and hybrid vehicles and the ROI for voluntary green power purchase programmes. Even in the worst markets, the climate change mitigation benefits of recycling are on par with these other common climate change mitigation initiatives, and during the best markets, the investment in recycling pays out more. These results highlight that an investment in kerb-side recycling, an added cost in most cases compared with not offering such services, is indeed a financially sound strategy for achieving desired sustainability outcomes.

### Opportunities to optimize recycling systems

Our results support that providing households with kerb-side recycling service (when assuming average US conditions over the past 15 years and 20% recycling rate) costs roughly 13% more than a system with no recycling (Fig. 4a,b) but decreases GHG emissions sixfold. If a community were to hypothetically double its recycling rate to 40% (Fig. 4c), overall system costs decrease and the net emissions footprint flips from a GHG emissions source when no recycling is practised (0.29 t CO<sub>2</sub>eq HH<sup>-1</sup> yr<sup>-1</sup>) to GHG emissions offset (–0.21 t CO<sub>2</sub>eq HH<sup>-1</sup> yr<sup>-1</sup>). In this hypothetical approach, our modelling does not anticipate the impacts on market or commodity prices when additional sources of secondary materials are introduced from existing recycling infrastructure (for example, current kerb-side programmes and recycling processing facilities).



**Fig. 4 | Annual net household costs and net GHG footprints associated with the waste management of an average US residential household under six scenarios.** **a**, All recyclables were collected as garbage for disposal; no recyclables were processed or marketed for sale. **b**, Typical waste-management process, including separate recyclables collection, processing and sale, were modelled with a 20% recycling rate. **c**, Same system in scenario **a** but using a 40% recycling rate. **d**, All were modelled under the same system as scenario **a**, but

glass was collected as garbage for disposal and not processed for sale.

**e**, Optimized approach refers to a typical waste-management process, including separate recyclables collection, processing and sale, were modelled with an 18% recycling rate based on recycling 100% of only high-commodity values (newspaper, cardboard, aluminium and steel cans, HDPE and PET bottles). Bars represent average annual household cost or GHG emissions for 2005–2020; red and brown lines correspond to minimum and maximum annual value.

Increasing kerb-side recycling rates in the United States, especially to levels as high as 40%, however, has proven impractical because of the necessary level of participation and the less-than-lucrative commodities that must be added to the bin to reach this level of recovery. Local governments have instead focused on removing materials with poor commodity value, with the most common being glass<sup>55</sup>. When glass is removed from the original recycling scenario, modest cost savings are achieved (US\$4 HH<sup>-1</sup> yr<sup>-1</sup>), but GHG emissions increase (from 0.05 to 0.06 t CO<sub>2</sub>eq HH<sup>-1</sup> yr<sup>-1</sup>; Fig. 4b,d).

The approach providing the best ROI does not emphasize placing the greatest amount of material in the recycling bin but instead on capturing those commodities with the greatest market value and highest GHG emissions-offset potential when they are remanufactured. Consider materials such as newspaper, cardboard, aluminium and steel cans and high density polyethylene (HDPE) and polyethylene terephthalate (PET) plastic bottles. If we eliminate all materials except these from our baseline of 20% recycling and increase the recycling rate for these higher value commodities to 75%, the overall recycling rate drops to 14%; however, the net system cost decreases (from US\$201 yr<sup>-1</sup> to US\$184 yr<sup>-1</sup>) and GHG emissions decreases (from 0.05 t CO<sub>2</sub>eq HH<sup>-1</sup> yr<sup>-1</sup> to 0.003 t CO<sub>2</sub>eq HH<sup>-1</sup> yr<sup>-1</sup>; Supplementary Fig. 5). At 100% recovery of the high-value materials (Fig. 4e), costs drop to a point where

having a recycling programme saves money in nearly all market conditions and still provides a sizable GHG emissions-reduction benefit (Supplementary Fig. 5).

## Discussion

We believe our findings provide needed insight to decisionmakers faced with restructuring or eliminating kerb-side recycling programmes as a community service. Yes, kerb-side recycling programmes cost money compared with collection of garbage for disposal alone. But this has always been the case—it is not new. In times of poor market conditions for recycled commodities, annual costs in the range of US\$37–42 per household should be expected. On the other hand, strong recycling markets should result in costs as low as US\$3 per household per year, though the challenges of maintaining robust markets should not be dismissed. Our analysis highlights that the GHG emissions-reduction benefit derived from recycling compares well or better than other citizen sustainability-based actions, even under times of challenging markets.

Among the actions needed to buttress local recycling programmes from the inevitable vagaries of recovered material markets, two emerge from our analysis. First, policies to support more resilient markets for the resources extracted from our waste are needed. Such policies

include promoting extended producer responsibility, which require manufacturers to be responsible—financially and/or operationally—for the recycling and disposal of their own products. Since 2021, Maine, Oregon, Colorado and California promulgated packaging producer responsibility laws, which will require manufacturers to provide financial support for each state's recycling infrastructure. Other policies, such as recycled-content mandates require manufacturers to use a certain percentage of post-consumer materials in new product manufacture. US states such as California and Washington have enacted recycled-content mandates in 2020–2021 for plastic packaging; the goals begin at 15% post-consumer content and increase to 50% by 2030–2031. Some US product manufacturers also established targets for integration of secondary materials into their packing or products (for example, the Coca-Cola Company's World Without Waste target of manufacturing bottles using 50% recycled content by 2030). In some cases, insufficient recovery rates impede widespread integration of secondary materials even when manufacturing practices exist to produce products with substantial quantities of recycled content. For 2021, the California Department of Resources Recycling and Recovery reported that 8 of 81 PET bottled beverage manufacturers used more than 15% post-consumer PET resin in their products<sup>36</sup>.

Second, kerb-side recycling programmes can benefit from targeting a smaller number of higher value commodities. We understand a reduction in the types of material allowed in the kerb-side recycling bin may not be popular with avid recyclers, but too many of the wrong materials in the recycling bin hinders optimization of the recycling system. Prioritizing higher value materials (both from economic and sustainability perspectives), especially if embraced in a more uniform manner across political boundaries, would probably result in less material collected because they represent approximately 18% of the total waste stream. Such prioritization would result in greater amounts of the remaining materials reaching landfills or combustion facilities, and this might possibly suppress market innovations for less-coveted secondary materials, but this must be balanced with the risk of dropping recycling programmes altogether.

## Methods

### Overall modelling approach

We formulated a model to calculate the costs and greenhouse gas (GHG) emissions associated with municipal solid waste (MSW) management for a typical US single-family residential home. Using this model, we estimated the collected, recycled, landfilled and combusted mass flows of 19 products (four paper products, three plastic products, two metal products, glass, food waste, yard trash and seven other products; Supplementary Table 1). In addition, we estimated the cost and GHG emissions of each waste-management process on a per household basis for seven US regions and the US national average for a monthly time series from April 2005 to June 2021. We designed the model to provide results for six waste-management processes: the separate collection of garbage and recyclables, processing of recyclables at a sorting facility, sale of recovered recyclables for remanufacture, disposal of residuals from sorting facility and disposal of garbage. The model's estimated costs were verified by comparing to actual reported household costs for several US communities (Supplementary Information Section 5). An illustration of the general data used and outputs of the model is provided in Supplementary Fig. 11.

Then we evaluated the mass, cost and GHG emissions impacts when the model was imposed with hypothetical changes to the recycling programme. These changes included eliminating the recycling programme, doubling the collected recycled mass, eliminating glass from the recycling programme and optimizing the recycling programme. For these evaluations, our model was not designed to be inherently dependent on adding new recycling infrastructure (for example, new kerb-side recycling programmes or recycling process facilities). Instead, the objective was to understand the impact to existing

communities. Furthermore, we evaluated the recycling-programme alternatives for the same monthly time series, whereby 15 years of varying commodity price data were used. Because the variable commodity price data were used, the impact to commodity prices and recovered market capacity (for example, lower prices due to increased supply) were not evaluated. The Methods section here provides a brief overview of key steps used to develop the model, beginning with a review of the waste mass generation, composition and disposition data used, followed by the GHG emissions modelling approach, the cost-modelling approach and how the alternative waste-management scenarios were implemented in the model.

### Data

The annual total waste (garbage and recyclables) generated for a typical US single-family residential home was estimated using daily household waste-generation rates, population statistics and household density. The total waste generated per household was modelled for 2005–2021 based on an assumed daily household waste-generation rate of 1.13 kg per person per day in 2005 that increased annually based on a growth rate estimated for each year using US Environmental Protection Agency MSW generation statistics for 2005–2017<sup>37</sup>. The initial waste-generation rate was determined by assuming that approximately 60% (ref. 58) of the average US total waste generated per person (2 kg per person per day (ref. 57)) was associated with single-family residents. For each year relative to 2005, the population was assumed to increase based on a growth rate estimated for each year using US population statistics for 2005–2021<sup>39</sup>. The household density (in persons per house) was provided for each year from 2005–2019 by the US Census Bureau<sup>60</sup>. In cases where data sources were not available for recent years, we used the most recent data available (for example, the household density for 2020 was used for 2021 as well). For each year, the annual total waste generated was divided by 52 to estimate the weekly total waste generated per household.

The weekly total waste generated per household was divided into garbage and recyclables generated per household for each year assuming a 23% recycling diversion rate or 20% recycling rate. The recycling rate is less than the recycling diversion rate because materials that cannot be marketed for sale (called residuals) are not included in this metric; the same value was used for 2005–2021. We selected a conservative recycling diversion rate, for example, the residential recycling rate in Ontario, Canada, was 62.8% (ref. 61), the average rate for the five New York City boroughs ranged from 14.6–20.7% (ref. 62), the rate for Seattle was 62.7% (ref. 63) and the Washington, D.C., rate was 20.8% (ref. 64). Data on the garbage and recyclables composition, and the dispositions of the disposed of garbage that were used are found in Supplementary Tables 1 and 2.

### GHG emissions modelling

For each of the 19 products, we measured the GHG emissions associated with collecting them as waste, processing of the recyclables at a sorting facility, remanufacturing and disposal via landfilling and combustion. The approach used to estimate the GHG emissions for waste management followed previously established methods in which GHG emissions impact factors (in units of metric tons of carbon dioxide emissions per Mg of material managed) were developed for each material and its management (for example, collection, processing at a sorting facility, remanufacturing, combustion and landfilling)<sup>42</sup>.

The process of estimating a representative impact factor was iterative and began with first developing impact factors using three waste-specific life-cycle assessment (LCA) models (or tools) (that is, Waste Reduction Model (WARM) v15 (accessed June 2020)<sup>65</sup>, MSW-Decision Support Tool (MSW-DST) v1 (accessed June 2020)<sup>66</sup> and Solid Waste Optimization Life-cycle Framework (SWOLF) v0.9.5 (accessed June 2020)<sup>40</sup>). The input assumptions used in these models are provided in detail in Supplementary Information Section 2. Changes



that departed from the default assumptions were based on expert discussions with US recyclers, sorting operators, landfill operators, landfill gas data reported by the US Environmental Protection Agency, MSW incinerators (MSWI) operators and previous research findings. The main goal of the first iteration was to determine which LCA tool to utilize as part of the model used to estimate GHG emissions for a typical US single-family residential household. Ultimately, the SWOLF model was selected because it has the greatest user flexibility (for example, users can change every input assumption); WARM and MSW-DST v1 limit the user's input changes. Note, the updated MSW-DST v2 (2021) has adopted most of the same input assumptions as SWOLF but still limits the user-input changes.

As part of the second iteration, a second set of impact factors were developed using default input assumptions for SWOLF. Two goals were associated with this iteration: (1) to compare the results of the net GHG emissions impacts for the baseline 20% recycling-rate scenario and the four hypothetical alternative scenarios when using the default impact factors (from the second iteration) to when using the first iteration SWOLF impact factors and (2) to use the default impact factors as a baseline and compare it to results of a Monte Carlo sensitivity analysis of the key input assumptions to the recycling, landfilling and combustion systems. The final impact factors that were used to generate the results in the manuscript were those created during the first iteration for SWOLF (those created when changing defaults). Supplementary Information Section 2.5 provides the baseline SWOLF impact factors, the input parameters that were altered for the sensitivity analysis, the results of how the net GHG emissions compare when using baseline SWOLF impact factors to the first iteration factors and the results of the 1,000-iteration Monte Carlo sensitivity analysis (per material, management and input assumption).

Supplementary Information Section 2 also describes, in detail, the specific LCA parameters included in the recycling, landfilling and combustion waste-management system boundaries, the material components and their characteristics, the life-cycle inventories and the life-cycle impact-assessment method. In general, the functional unit was 1 Mg of product (for example, 1 Mg of newspaper) disposed of by a typical US single-family residential home. All the models except for WARM rely on a reasonable mass of waste to model the emissions associated with individually constructing and operating a waste-treatment facility. Therefore, the functional unit will remain one short ton. However, to estimate the emissions associated with one ton, the modelled mass (or reference flow) was 100,000 Mg. This value was chosen to represent a hypothetical community of 50,000 people generating waste at 2.04 kg per person per day (equivalent to the reported mass of waste generated by a US resident<sup>57</sup>). In brief, the three waste-management systems included (1) a single-stream materials recovery facility (MRF) in which paper, plastic, metal and glass are sent to their respective remanufacturers, (2) a traditional non-bioreactor mixed-waste landfill and (3) a mass-burn mixed-waste municipal solid waste incinerator. Under the zero-burden assumption, the waste entering any of these processes is considered to carry none of the emissions associated with the extraction, processing, manufacture and use (with some exception); these life stages are referred to as upstream<sup>67–69</sup>. This assumption is commonly adopted because the emissions associated with the upstream stages are not typically considered with respect to solid waste decisionmaking; however, certain processes, such as recycling, do account for the upstream emissions by assuming that the recycled material offsets the emissions associated with using a virgin material<sup>42</sup>. Similarly, when electricity is generated from landfill gas or combustion, that electricity offsets the use of fossil fuels used to produce electricity<sup>70</sup>. Supplementary Information Section 2 provides more on the possible GHG emissions-offset sources.

### Cost modelling

Several waste LCA models account for the costs of collecting garbage and recyclables from single-family homes<sup>40,66</sup>. The SWOLF and

MSW-DST models estimate collection costs based on a mechanistic modelling approach in which numerous interdependent user-defined parameters are used to estimate the number of collection vehicles, the vehicle fuel consumption and the vehicle distance travelled. For this study, we created a simplified mechanistic model to estimate collection costs based on reviewing the SWOLF and MSW-DST collection cost-modelling technical assumptions and underlying equations and directly communicating with private and public waste haulers to survey current collection cost practices.

The collections costs per household were calculated as the sum of the annual vehicle fuel costs, the annual operating and maintenance costs of a collection vehicle and the annualized capital costs of a collection vehicle and storage bin. The collection costs were estimated based on determining the collection schedule, collection operation times, labour requirements, vehicle operational parameters, travel speeds, travel distances, fuel-usage rates and household waste storage systems. Specific parameter data used in the model are presented in Supplementary Table 9, and more details on the equations used to calculate the collection costs are detailed in Supplementary Information Section 3.

We determined for each of the seven regions an average landfill and MSWI disposal fee by aggregating state-specific disposal fee data into their respective regions (Supplementary Tables 10 and 11). For example, Region 1 landfill and MSWI disposal fees were estimated as the average of the fees associated with the states included in that region. The disposal fees were applied to the mass of garbage collected for disposal and to the residual mass from a recycling sorting facility. We identified the mass of garbage/residual landfilled and combusted (sent to a MSWI) based on aggregating state-specific disposal disposition data (percent of total waste disposed of landfilled or combusted)<sup>71</sup> into their respective regions.

We estimated the annual revenue associated with recycling by a single-family residential home by using the composition of recyclables sorted and each material's commodity price. The commodity prices were either reported directly or forecasted, and historical prices were inflated to 2020 US dollars prices based on gross domestic product and a 2012 deflator (commodity price data are provided in Supplementary Data). The annual revenue was estimated for the seven regions based on region-specific data and Supplementary Table 12, Supplementary Information presents the annual household revenue. Data for the forecasted commodity prices are available in the Supplementary Information Section 3 methods as well.

We also estimated the added cost for residential single-family recycling for all seven regions in US\$ per household per month. The costs were estimated using region-specific disposal fees, disposal disposition and recycling commodity prices for April 2005–June 2021. Independent of time or region, we applied the same assumptions previously describing the garbage and recyclables-generation rate, material stream compositions, 20% recycling rate, collection costs for garbage and recyclables and recyclables-processing costs. The model developed for the study relies on robust US national datasets for each state; therefore, we validated the model results by compiling solid waste-management cost data from across a region (Florida) and comparing the actual costs incurred by residents to those estimated by the model.

### Alternative waste-management scenarios

We evaluated the annual net residential costs and GHG emissions impacts (in units US\$ per household per year and t CO<sub>2</sub>e per household per year) of increasing the recycling rate from 20% to 40%, eliminating recycling, eliminating mixed-plastic recycling, eliminating glass recycling, eliminating mixed-paper recycling and eliminating certain materials while increasing the recycling rate of others (referred to as a hybrid approach). The evaluation was conducted using US national average data on costs and using the GHG emissions impact factors developed for 2005–2021.

The recycling rate was increased from 20% to 40% by identifying the total additional recycled mass needed to meet a 40% recycling rate and distributing that mass among the ten recyclables products for each year. The additional recycled mass was assumed to be transferred from the original collected garbage mass to the collected recyclables mass. The additional recycled mass was distributed according to the material composition of the collected garbage stream to ensure that the additional mass did not exceed the available transferable mass. The additional recycled mass ranged from 4.5–4.9 kg per household per week for 2005–2019, and the composition used (independent of time) to distribute that mass among PET, HDPE, mixed plastics, aluminium, steel, glass, mixed paper, newspaper, corrugated containers and aseptic cartons was 4%, 2%, 26%, 2%, 3%, 9%, 40%, 5%, 10% and 1%, respectively. The net costs accounted for the increase in recyclables collection vehicles, decrease in garbage collection vehicles, increase in recyclables revenue and processing costs and decrease in disposal costs. The GHG emissions accounted for a greater avoidance due to increased recycling and a decrease in emissions from reduced landfilling/combustion.

For the eliminate recycling, eliminate mixed plastic, eliminate glass and eliminate mixed-paper scenarios, the net costs and GHG emissions were modelled using the same assumptions for a system with a 20% recycling rate, including the previously used garbage and recyclables-generation rate, material stream compositions, collection costs for garbage and recyclables and recyclables-processing costs. However, for each scenario, the garbage/recyclable generation rate and material stream composition assumptions were modified. For the eliminate recycling scenario, we assumed that all the originally recycled mass would be collected as garbage and no recyclables revenue, processing costs or GHG emissions avoidances were included. In the scenarios where a material was eliminated from recycling, we assumed that the originally recycled mass for that material would be collected as garbage and it would not contribute to recyclables revenue, processing costs or GHG emissions avoidances. Note that in the eliminate mixed-paper recycling scenario, both mixed paper and aseptic cartons were assumed to be eliminated as well.

In the optimized approach scenario, we used the same waste-generation rate as the system with a 20% recycling rate; but here we modelled the net costs and GHG emissions assuming only newspaper, glass, steel, aluminium, corrugated containers, PET and HDPE were recycled, and when recycled, each individual material reached a 100% recycling rate for a total recycling rate of 18%. The same collection, disposal and recyclables-processing costs as the other scenarios were applied. The recyclables revenue and GHG emissions recycling avoidance were estimated for only the seven products.

### Reporting summary

Further information on research design is available in the Nature Portfolio Reporting Summary linked to this article.

### Data availability

All data used to produce the results of our analysis are available in the Supplementary Information.

### Code availability

The custom computer code used to generate the missing recycling market value prices in this study can be made available to researchers upon request.

### References

- Anshassi, M., Preuss, B. & Townsend, T. G. Moving beyond recycling: examining steps for local government to integrate sustainable materials management. *J. Air Waste Manage. Assoc.* **71**, 1039–1052 (2021).
- Fitzgerald, G. C., Krones, J. S. & Themelis, N. J. Greenhouse gas impact of dual stream and single stream collection and separation of recyclables. *Resour. Conserv. Recycl.* **69**, 50–56 (2012).
- Larsen, A. W., Merrild, H., Møller, J. & Christensen, T. H. Waste collection systems for recyclables: an environmental and economic assessment for the municipality of Aarhus (Denmark). *Waste Manage.* **30**, 744–754 (2010).
- Wagner, T. P. & Broaddus, N. The generation and cost of litter resulting from the curbside collection of recycling. *Waste Manage.* **50**, 3–9 (2016).
- Brooks, A. L., Wang, S. & Jambeck, J. R. The Chinese import ban and its impact on global plastic waste trade. *Sci. Adv.* **4**, eaat0131 (2018).
- Qu, S. et al. Implications of China's foreign waste ban on the global circular economy. *Resour. Conserv. Recycl.* **144**, 252–255 (2019).
- Tucker, Brian. How many curbside recycling programs have been cut? *Waste Dive* (30 September 2020).
- Endo, Jun. Philippines slams the door on world's plastic waste. *Nikkei Asia* (14 September 2019).
- Szczepanski, Mallory. India announces plans to ban scrap plastic imports. *Waste360* (3 July 2019).
- Implementation of Import of Non-hazardous and Toxic Material Waste as Raw Material Industry* (Minister of Trade, Minister of Environment and Forestry, Minister of Trade Industry & Head of State Policy of the Republic of Indonesia, 2020).
- Tanimoto, A. 2019 The West Coast Contamination Initiative: Results from California, Oregon, and Washington (2020); <https://recyclingpartnership.org/blog-the-west-coast-contamination-initiative-results-from-california-oregon-and-washington/>. The Recycling Partnership
- Townsend, T. G. & Anshassi, M. *Examining Contamination Rates at Florida Materials Recovery Facilities* (2020); <https://flrecycling.org/wp-content/uploads/2020/06/UF-MRF-Contamination-Report-Final.pdf>. Florida Recycling Partnership Foundation
- Ikiz, E., Maclaren, V. W., Alfred, E. & Sivanesan, S. Impact of COVID-19 on household waste flows, diversion and reuse: the case of multi-residential buildings in Toronto, Canada. *Resour. Conserv. Recycl.* **164**, 105111 (2021).
- Kulkarni, B. N. & Anantharama, V. Repercussions of COVID-19 pandemic on municipal solid waste management: challenges and opportunities. *Sci. Total Environ.* **743**, 140693 (2020).
- Liao, N. et al. Can waste management system be a greenhouse gas sink? Perspective from Shanghai, China. *Resour. Conserv. Recycl.* **180**, 106170 (2022).
- Fei, X., Fang, M. & Wang, Y. Climate change affects land-disposed waste. *Nat. Clim. Change* **11**, 1004–1005 (2021).
- Gómez-Sanabria, A., Kiesewetter, G., Klimont, Z., Schoepp, W. & Haberl, H. Potential for future reductions of global GHG and air pollutants from circular waste management systems. *Nat. Commun.* **13**, 106 (2022).
- Duren, R. M. et al. California's methane super-emitters. *Nature* **575**, 180–184 (2019).
- Morris, J. Comparative LCAs for curbside recycling versus either landfilling or incineration with energy recovery (12 pp). *Int. J. Life Cycle Assess.* **10**, 273–284 (2005).
- Kaplan, P. O., Ranjithan, S. R. & Barlaz, M. A. Use of life-cycle analysis to support solid waste management planning for Delaware. *Environ. Sci. Technol.* **43**, 1264–1270 (2009).
- van Ewijk, S., Stegemann, J. A. & Ekins, P. Limited climate benefits of global recycling of pulp and paper. *Nat. Sustain.* **4**, 180–187 (2021).
- Borrelle, S. B. et al. Predicted growth in plastic waste exceeds efforts to mitigate plastic pollution. *Science* **369**, 1515–1518 (2020).



23. Lau, W. W. Y. et al. Evaluating scenarios toward zero plastic pollution. *Science* **369**, 1455–1461 (2020).
24. Dubois, G. et al. It starts at home? Climate policies targeting household consumption and behavioral decisions are key to low-carbon futures. *Energy Res. Soc. Sci.* **52**, 144–158 (2019).
25. Schmidt, S. et al. Understanding GHG emissions from Swedish consumption—current challenges in reaching the generational goal. *J. Clean. Prod.* **212**, 428–437 (2019).
26. Castellani, V., Beylot, A. & Sala, S. Environmental impacts of household consumption in Europe: comparing process-based LCA and environmentally extended input–output analysis. *J. Clean. Prod.* **240**, 117966 (2019).
27. Ivanova, D. et al. Quantifying the potential for climate change mitigation of consumption options. *Environ. Res. Lett.* <https://doi.org/10.1088/1748-9326/ab8589> (2020).
28. Anshassi, M., Laux, S. J. & Townsend, T. G. Approaches to integrate sustainable materials management into waste management planning and policy. *Resour. Conserv. Recycl.* **148**, 55–66 (2019).
29. Andreasi Bassi, S., Christensen, T. H. & Damgaard, A. Environmental performance of household waste management in Europe—an example of 7 countries. *Waste Manage.* **69**, 545–557 (2017).
30. Gu, F., Wang, J., Guo, J. & Fan, Y. Dynamic linkages between international oil price, plastic stock index and recycle plastic markets in China. *Int. Rev. Econ. Financ.* **68**, 167–179 (2020).
31. Shamsuyeva, M. & Endres, H.-J. Plastics in the context of the circular economy and sustainable plastics recycling: comprehensive review on research development, standardization and market. *Compos. Part C* **6**, 100168 (2021).
32. ISRI staff. *The Plastic Scrap Market Since China’s Import Ban* (ISRI, 2019). <https://www.isri.org/docs/default-source/commodities/the-plastic-scrap-market-since-china’s-import-ban.pdf?sfvrsn=2>
33. *Secondary Materials Pricing & Secondary Fiber Pricing* (Recycling Markets, 2020); <https://www.recyclingmarkets.net/secondarymaterials/>
34. *2020 State of Curbside Recycling Report* (Recycling Partnership, 2020); [https://recyclingpartnership.org/wp-content/uploads/dlm\\_uploads/2020/02/2020-State-of-Curbside-Recycling.pdf](https://recyclingpartnership.org/wp-content/uploads/dlm_uploads/2020/02/2020-State-of-Curbside-Recycling.pdf)
35. *2018 Solid Waste Annual Reports* (Florida Department of Environmental Protection, 2019); <http://southernwasteinformationexchange.com/fdep-solid-waste-annual-reports/>
36. Morris, J. & Pasterz, P. Rhythms and reasons in pricing. *Resource Recycling* (1 May 2017).
37. Huang, Q. et al. Modelling the global impact of China’s ban on plastic waste imports. *Resour. Conserv. Recycl.* **154**, 104607 (2020).
38. Gephart, J. A. et al. Environmental performance of blue foods. *Nature* **597**, 360–365 (2021).
39. Christensen, T. H. et al. Application of LCA modelling in integrated waste management. *Waste Manage.* **118**, 313–322 (2020).
40. Levis, J. W., Barlaz, M. A., DeCarolis, J. F. & Ranjithan, S. R. Systematic exploration of efficient strategies to manage solid waste in U.S. municipalities: perspectives from the Solid Waste Optimization Life-Cycle Framework (SWOLF). *Environ. Sci. Technol.* **48**, 3625–3631 (2014).
41. Rigamonti, L., Niero, M., Haupt, M., Grosso, M. & Judl, J. Recycling processes and quality of secondary materials: food for thought for waste-management-oriented life cycle assessment studies. *Waste Manage.* **76**, 261–265 (2018).
42. Anshassi, M. & Townsend, T. G. Reviewing the underlying assumptions in waste LCA models to identify impacts on waste management decision making. *J. Clean. Prod.* **313**, 127913 (2021).
43. Circular Matters *US Company Recycled Plastic Content Goal Analysis—Supply & Demand* (2021); <https://cdn.ymaws.com/www.ameripen.org/resource/resmgr/docs/AMERIPEN-recycled-content-pa.pdf>. AMERIPEN
44. Muneer, T. et al. Energetic, environmental and economic performance of electric vehicles: experimental evaluation. *Transp. Res. Part D* **35**, 40–61 (2015).
45. Orsi, F., Muratori, M., Rocco, M., Colombo, E. & Rizzoni, G. A multi-dimensional well-to-wheels analysis of passenger vehicles in different regions: primary energy consumption, CO<sub>2</sub> emissions, and economic cost. *Appl. Energy* **169**, 197–209 (2016).
46. Samaras, C. & Meisterling, K. Life cycle assessment of greenhouse gas emissions from plug-in hybrid vehicles: implications for policy. *Environ. Sci. Technol.* **42**, 3170–3176 (2008).
47. van Vliet, O. P. R., Kruihof, T., Turkenburg, W. C. & Faaij, A. P. C. Techno-economic comparison of series hybrid, plug-in hybrid, fuel cell and regular cars. *J. Power Sources* **195**, 6570–6585 (2010).
48. Pero, F. D., Delogu, M. & Pierini, M. Life cycle assessment in the automotive sector: a comparative case study of internal combustion engine (ICE) and electric car. *Procedia Struct. Integrity* **12**, 521–537 (2018).
49. Elgowainy, A. et al. Cost of ownership and well-to-wheels carbon emissions/oil use of alternative fuels and advanced light-duty vehicle technologies. *Energy Sustain. Dev.* **17**, 626–641 (2013).
50. Doucette, R. T. & McCulloch, M. D. Modeling the prospects of plug-in hybrid electric vehicles to reduce CO<sub>2</sub> emissions. *Appl. Energy* **88**, 2315–2323 (2011).
51. Burnham, A. et al. *Comprehensive Total Cost of Ownership Quantification for Vehicles with Different Size Classes and Powertrains*; (OSTI, 2021) <https://doi.org/10.2172/1780970>
52. *Summary of Travel Trends 2017 National Household Travel Survey* (US Department of Transportation, 2018); <https://doi.org/10.2172/885762>
53. US Department of Transportation. Table VM-2M—highway statistics 2019. *Highway Statistics Series* <https://www.fhwa.dot.gov/policyinformation/statistics/2019/vm2m.cfm> (2020).
54. *2020 Average Monthly Bill—Residential Table 5a* (US Energy Information Administration, 2021).
55. Tucker, Brian. How recycling has changed in all 50 states. *Waste Dive* (15 November 2019).
56. *2021 Plastic Beverage Container Virgin and Postconsumer Resin Report* (CalRecycle, 2022).
57. *Advancing sustainable materials management: 2017 Fact Sheet* (US EPA, 2019); [https://www.epa.gov/sites/production/files/2019-11/documents/2017\\_facts\\_and\\_figures\\_fact\\_sheet\\_final.pdf](https://www.epa.gov/sites/production/files/2019-11/documents/2017_facts_and_figures_fact_sheet_final.pdf)
58. *Municipal Solid Waste Generation, Recycling, and Disposal in the United States: Facts and Figures for 2010* (US EPA, 2011); <https://www.nrc.gov/docs/ML1409/ML14094A389.pdf>
59. Population, Total—United States (World Bank, 2020); <https://data.worldbank.org/indicator/SP.POP.TOTL?locations=US>
60. *Historical Households Tables* (US Census Bureau, 2020); <https://www.census.gov/data/tables/time-series/demo/families/households.html>
61. Lakhani, C. Diversion, but at what cost? The economic challenges of recycling in Ontario. *Resour. Conserv. Recycl.* **95**, 133–142 (2015).
62. *Annual Report: New York City Curbside and Containerized Municipal Refuse and Recycling Statistics 2020* (New York City Department of Sanitation, 2020); [https://dsny.cityofnewyork.us/wp-content/uploads/2020/08/about\\_dsny-collections-annual-2020.pdf](https://dsny.cityofnewyork.us/wp-content/uploads/2020/08/about_dsny-collections-annual-2020.pdf)

63. Seattle Public Utilities. *2018 Waste Prevention & Recycling Report* (Seattle Public Utilities, 2019); [https://www.seattle.gov/Documents/Departments/SPU/Documents/Recycling\\_Rate\\_Report\\_2018.pdf](https://www.seattle.gov/Documents/Departments/SPU/Documents/Recycling_Rate_Report_2018.pdf)
64. *FY 2011 Public Report on Recycling* (Washington DC District Department of the Environment, 2012); <https://doee.dc.gov/sites/default/files/dc/sites/ddoe/publication/attachments/FY%202011%20Recycling%20Report%20with%20table%20of%20contents.pdf>
65. ICF International *Documentation for Greenhouse Gas Emission and Energy Factors Used in the Waste Reduction Model (WARM): Background Chapters* (US EPA, 2016); [https://www.epa.gov/sites/production/files/2016-03/documents/warm\\_v14\\_background.pdf](https://www.epa.gov/sites/production/files/2016-03/documents/warm_v14_background.pdf)
66. Thorneloe, S. A., Weitz, K. & Jambeck, J. Application of the US decision support tool for materials and waste management. *Waste Manage.* **27**, 1006–1020 (2007).
67. Ekvall, T., Assefa, G., Björklund, A., Eriksson, O. & Finnveden, G. What life-cycle assessment does and does not do in assessments of waste management. *Waste Manage.* **27**, 989–996 (2007).
68. Gentil, E. C. et al. Models for waste life cycle assessment: review of technical assumptions. *Waste Manage.* **30**, 2636–2648 (2010).
69. Martin, E. W., Chester, M. V. & Vergara, S. E. Attributional and consequential life-cycle assessment in biofuels: a review of recent literature in the context of system boundaries. *Curr. Sustain. Renew. Energy Rep.* **2**, 82–89 (2015).
70. Anshassi, M., Smallwood, T. & Townsend, T. G. Life cycle GHG emissions of MSW landfilling versus Incineration: expected outcomes based on US landfill gas collection regulations. *Waste Manage.* **142**, 44–54 (2022).
71. Michaels, T. & Krishnan, K. *Energy Recovery Council 2018 Directory of Waste-to-Energy Facilities* (2018); <http://energyrecoverycouncil.org/wp-content/uploads/2019/10/ERC-2018-directory.pdf>. Energy Recovery Council

## Acknowledgements

This work was financially supported by the Hinkley Center for Solid and Hazardous Waste Management in Gainesville, Florida. We thank N. Robey for her review of the Methods and manuscript. We acknowledge the Florida Recycling Partnership Foundation for comments and discussion on the planning and analysis of the paper. We appreciate the data for the recycling commodity markets provided by Recycling Markets Limited and the access to the life-cycle assessment models, Solid Waste Optimization Framework (SWOLF)

and Municipal Solid Waste Decision Support Tool (MSW-DST), provided by North Carolina State University and RTI International, respectively. We thank our colleagues from multiple Florida counties (for example, Alachua County, Indian River County, Palm Beach County, Hillsborough County, Sarasota County, Orange County, Lee County) who supported our analysis through their data-collection efforts on solid waste-management costs used in the analysis. The authors are responsible for the content of the paper, and the findings do not represent the views of the funding agencies.

## Author contributions

M.A. performed the research and analysed the data. T.G.T. conceived the idea and designed the study. Both authors wrote the manuscript.

## Competing interests

The authors declare no competing interests.

## Additional information

**Supplementary information** The online version contains supplementary material available at <https://doi.org/10.1038/s41893-023-01122-8>.

**Correspondence and requests for materials** should be addressed to Timothy G. Townsend.

**Peer review information** *Nature Sustainability* thanks Costas Velis, Eleni Iacovidou, Matthew Franchetti, and George F. Banias for their contribution to the peer review of this work.

**Reprints and permissions information** is available at [www.nature.com/reprints](http://www.nature.com/reprints).

**Publisher's note** Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.

Springer Nature or its licensor (e.g. a society or other partner) holds exclusive rights to this article under a publishing agreement with the author(s) or other rightsholder(s); author self-archiving of the accepted manuscript version of this article is solely governed by the terms of such publishing agreement and applicable law.

© The Author(s), under exclusive licence to Springer Nature Limited 2023

## Reporting Summary

Nature Portfolio wishes to improve the reproducibility of the work that we publish. This form provides structure for consistency and transparency in reporting. For further information on Nature Portfolio policies, see our [Editorial Policies](#) and the [Editorial Policy Checklist](#).

### Statistics

For all statistical analyses, confirm that the following items are present in the figure legend, table legend, main text, or Methods section.

- | n/a                                 | Confirmed   |
|-------------------------------------|---|
| <input type="checkbox"/>            | <input checked="" type="checkbox"/> The exact sample size ( $n$ ) for each experimental group/condition, given as a discrete number and unit of measurement   |
| <input type="checkbox"/>            | <input checked="" type="checkbox"/> A statement on whether measurements were taken from distinct samples or whether the same sample was measured repeatedly   |
| <input checked="" type="checkbox"/> | <input type="checkbox"/> The statistical test(s) used AND whether they are one- or two-sided<br><i>Only common tests should be described solely by name; describe more complex techniques in the Methods section.</i>   |
| <input checked="" type="checkbox"/> | <input type="checkbox"/> A description of all covariates tested   |
| <input checked="" type="checkbox"/> | <input type="checkbox"/> A description of any assumptions or corrections, such as tests of normality and adjustment for multiple comparisons  |
| <input checked="" type="checkbox"/> | <input type="checkbox"/> A full description of the statistical parameters including central tendency (e.g. means) or other basic estimates (e.g. regression coefficient) AND variation (e.g. standard deviation) or associated estimates of uncertainty (e.g. confidence intervals) |
| <input checked="" type="checkbox"/> | <input type="checkbox"/> For null hypothesis testing, the test statistic (e.g. $F$ , $t$ , $r$ ) with confidence intervals, effect sizes, degrees of freedom and $P$ value noted<br><i>Give <math>P</math> values as exact values whenever suitable.</i>                            |
| <input checked="" type="checkbox"/> | <input type="checkbox"/> For Bayesian analysis, information on the choice of priors and Markov chain Monte Carlo settings   |
| <input checked="" type="checkbox"/> | <input type="checkbox"/> For hierarchical and complex designs, identification of the appropriate level for tests and full reporting of outcomes   |
| <input checked="" type="checkbox"/> | <input type="checkbox"/> Estimates of effect sizes (e.g. Cohen's $d$ , Pearson's $r$ ), indicating how they were calculated   |

*Our web collection on [statistics for biologists](#) contains articles on many of the points above.*

### Software and code

Policy information about [availability of computer code](#)

- |                 |   |
|-----------------|---|
| Data collection | No software was used to collect the data used in this study. Instead, the cost, mass flow, and greenhouse gas emissions data for waste management were collected from various sources and harmonized to provide results for a typical US single-family residential household. |
| Data analysis   | Study developed spreadsheet model was used to conduct the analysis that calculated the cost, mass flow, and greenhouse gas emissions for a typical US single-family residential household.  |

For manuscripts utilizing custom algorithms or software that are central to the research but not yet described in published literature, software must be made available to editors and reviewers. We strongly encourage code deposition in a community repository (e.g. GitHub). See the Nature Portfolio [guidelines for submitting code & software](#) for further information.

### Data

Policy information about [availability of data](#)

All manuscripts must include a [data availability statement](#). This statement should provide the following information, where applicable:

- Accession codes, unique identifiers, or web links for publicly available datasets
- A description of any restrictions on data availability
- For clinical datasets or third party data, please ensure that the statement adheres to our [policy](#)

The authors declare that nearly all data supporting the findings of this study are available within the article and its supplementary information files. Requests for the data code used to project some of the commodities' recycling market values can be made available.



## Human research participants

Policy information about [studies involving human research participants and Sex and Gender in Research](#).

Reporting on sex and gender	NA
Population characteristics	NA
Recruitment	NA
Ethics oversight	NA

Note that full information on the approval of the study protocol must also be provided in the manuscript.

## Field-specific reporting

Please select the one below that is the best fit for your research. If you are not sure, read the appropriate sections before making your selection.

Life sciences  Behavioural & social sciences  Ecological, evolutionary & environmental sciences

For a reference copy of the document with all sections, see [nature.com/documents/nr-reporting-summary-flat.pdf](https://www.nature.com/documents/nr-reporting-summary-flat.pdf)

## Ecological, evolutionary & environmental sciences study design

All studies must disclose on these points even when the disclosure is negative.

Study description	We employed a modeling framework to estimate the following: (1) the mass flows for the separate collection of garbage and recyclables, processing of recyclables at a sorting facility, sale of recovered recyclables for remanufacture, disposal of residuals from sorting facility, and disposal of garbage; (2) the monthly household cost of waste management for seven US regions for a time series from April 2005 to June 2021; (3) the corresponding annual GHG emissions footprint associated with household waste management; and (4) the mass, costs, and GHG emissions impacts when the model was imposed with hypothetical changes to the recycling program. We contrasted our results with the GHG emissions reduction return-on-invest from other sustainable practices, namely purchasing green utility power at a higher rate and transitioning from gasoline to hybrid or electric vehicles.
Research sample	Greenhouse gas emissions data originated from waste management-based lifecycle assessment models (SWOLF version 2019) that utilizes most US national average statistics concerning collection, recycling, landfilling, and combustion disposal of waste. The economic data used represents a typical US single-family household's waste management costs and it is comprised of collection costs, disposal costs, recyclables processing costs, and recyclables sale for remanufacture. Collection cost data was retrieved from the economic sub-models within the MSWDST and SWOLF models, as well as from several Florida local government contracts with private waste haulers. Disposal cost data for landfilling and combusting for each of the 50 US states were retrieved from reports published by Environmental Research and Education Foundation (EREF) and Tishman Environment and Design Center, respectively. Recycling processing costs data were derived from reports on materials recovery facilities' processing rate in the northeastern US region and through communications with operators in Florida. Recycling commodity data for each of the six US regions for 2005 to 2021 were extracted from an industry standard index price database managed by Recycling Markets Limited (RML). Four main data components were used as part of the mass flow estimates: waste generation, recycling rate, recyclables and garbage composition, and disposal disposition. The waste generation rate per capita for the US was based on US EPA estimates. The recycling rate was assumed to be 20% based on The recycling composition for 11 material components were estimated from 9 Florida-specific studies and 1 northeastern US study. The garbage composition for 19 material components were based on seven Florida-specific studies. The disposition of combusted and landfilled waste for each of the 50 states were used and retrieved from the Energy Recovery Council reports. The data sources for the environmental return-on-investment included US DOT, US EPA, and published literature.
Sampling strategy	Cost data was collected for the US and six US regions, waste and recycling composition data was collected for Florida, GHG emissions data were estimated for the US.
Data collection	All data were collected online through publicly available datasets and reports or through communications with industry experts. However, the recycling commodity market prices for some materials were forecasted using a linear regression code. In addition, the greenhouse gas emissions were modeled using lifecycle assessment models not directly publicly available (i.e., SWOLF).
Timing and spatial scale	The economic, mass flow, and greenhouse gas emissions data used represents a temporal scale from 2005 to 2021 and a spatial scale for seven regions of the US specific to a typical single-family residential household. Wherever possible data from each of the US 50 states was used, and when data was limited for a state, Florida-specific data was utilized as a proxy.
Data exclusions	No data was excluded from the study. However, due to limited time and data availability, some regions' (e.g., US northeastern and southern, Florida,US) data (e.g., garbage and recyclables composition) were extrapolated and used for other regions (e.g., midwest, northwest, central).

Reproducibility

We provide detailed calculation formulas and steps, as well as all datasets and sources associated with our study in the supplementary methods and dataset spreadsheet to ensure full transparency and reproducibility.

Randomization

No randomization was used because we targeted providing waste management costs, mass flow, and greenhouse gas emissions results for the US on a national basis and on a regional basis.

Blinding

No blinding was used as part of this study.

Did the study involve field work?  Yes  No

## Reporting for specific materials, systems and methods

We require information from authors about some types of materials, experimental systems and methods used in many studies. Here, indicate whether each material, system or method listed is relevant to your study. If you are not sure if a list item applies to your research, read the appropriate section before selecting a response.

### Materials & experimental systems

- | n/a                                 | Involvement in the study                               |
|-------------------------------------|--|
| <input checked="" type="checkbox"/> | <input type="checkbox"/> Antibodies                    |
| <input checked="" type="checkbox"/> | <input type="checkbox"/> Eukaryotic cell lines         |
| <input checked="" type="checkbox"/> | <input type="checkbox"/> Palaeontology and archaeology |
| <input checked="" type="checkbox"/> | <input type="checkbox"/> Animals and other organisms   |
| <input checked="" type="checkbox"/> | <input type="checkbox"/> Clinical data                 |
| <input checked="" type="checkbox"/> | <input type="checkbox"/> Dual use research of concern  |

### Methods

- | n/a                                 | Involvement in the study                        |
|-------------------------------------|---|
| <input checked="" type="checkbox"/> | <input type="checkbox"/> ChIP-seq               |
| <input checked="" type="checkbox"/> | <input type="checkbox"/> Flow cytometry         |
| <input checked="" type="checkbox"/> | <input type="checkbox"/> MRI-based neuroimaging |