Environmental Reviews

Review

Impacts of plastic waste management strategies

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Abstract

The ecological and societal impacts of plastics production, use, and waste are a complex global challenge. Management strategies to mitigate the impacts of plastics, such as recycling, waste-to-energy, and replacement with alternative materials have impacts of their own. Achieving long-term sustainability of plastics use therefore requires considering the externalized impacts of such management strategies. Here, we assessed the literature on the most common plastic waste management strategies to identify their impacts in relation to the sustainable development goals. We reviewed impacts of bans, levies, and taxes; alternative products; recycling; waste-to-energy; plastic recovery; and extended producer responsibility. Our analysis identified a total of 259 measured impacts of plastic waste mitigation strategies, from 113 papers. Ninety-three impacts were negative, 104 were positive, 11 were neutral, and 51 depended on the context of implementation. Consideration of the impacts of both plastic materials and management strategies is necessary to avoid perverse outcomes of plastic pollution mitigation efforts.

Key words: extended producer responsibility, externality, plastic alternatives, plastic legislation, plastic pollution, recovery, recycling, sustainability, trade-off, waste-to-energy

Résumé

Les impacts écologiques et sociétaux de la production, de l'utilisation et des déchets de plastiques constituent un défi mondial complexe. Les stratégies de gestion visant à atténuer les impacts des plastiques, comme le recyclage, la valorisation énergétique des déchets et le remplacement par des matériaux alternatifs, ont leurs propres impacts. Pour assurer la durabilité à long terme de l'utilisation des plastiques, il faut donc tenir compte des impacts externalisés de ces stratégies de gestion. Les auteurs ont évalué la littérature sur les stratégies de gestion des déchets plastiques les plus courantes afin d'identifier leurs impacts par rapport aux objectifs de développement durable. Ils ont examiné les impacts des interdictions, des impôts et des taxes, des produits alternatifs, du recyclage, de la valorisation énergétique des déchets, de la récupération des plastiques et de la responsabilité élargie des producteurs. Leur analyse a identifié un total de 259 impacts mesurés des stratégies de réduction des déchets plastiques, provenant de 113 articles. Quatre-vingt-treize impacts étaient négatifs, 104 étaient positifs, 11 étaient neutres et 51 dépendaient du contexte de mise en œuvre. La prise en compte des impacts des matériaux plastiques et des stratégies de gestion est nécessaire pour éviter les effets pervers des efforts de réduction de la pollution plastique. [Traduit par la Rédaction]

Mots-clés : responsabilité élargie des producteurs, externalité, solutions de rechange aux plastiques, législation sur les plastiques, pollution plastique, récupération, recyclage, durabilité, compromis, valorisation énergétique des déchets

Introduction

Plastic production has outpaced the global community's capacity to manage the resulting waste (Borrelle et al. 2020). Designed to be durable, plastics do not degrade naturally; thus, waste can present severe long-term environmental and social impacts (Villarrubia-Gómez et al. 2018). Disposal methods in the current linear plastics economy include recycling, landfilling or incineration, often producing significant levels of plastic "leakage" into the environment, even when plastic waste is managed (Borrelle et al. 2020).

The impacts of plastics are not limited to the end-of-life phase (i.e., pollution). From the extraction of the raw materials (oil and gas) to the management and mismanagement of waste, there are risks to human health and wellbeing, social equity issues, detrimental ecological impacts, greenhouse gas (GHG) emissions and increased economic costs (Table S1) (Newman et al. 2015; Royer et al. 2018; Beaumont et al. 2019; Stoett and Vince 2019; Bucci et al. 2020; Murphy et al. 2021). The costs of these impacts disproportionately affect marginalised communities, making plastics and plastic pollution a critical environmental justice issue (CIEL 2019; Liboiron 2021; UNEP 2021; Fuller et al. 2022). Addressing this global challenge requires significant, system-wide action (Borrelle et al. 2017, 2020). Plastic waste management strategies (hereafter, "PWMS") are employed throughout the plastic life cycle to address the plastics challenge. The waste hierarchy is a heuristic that can be used to evaluate and prioritise PWMS (Fig. 1). Actions at the top of the waste hierarchy that prioritize the prevention, reduction and reuse of materials are preferred for reducing the negative impacts of plastics (Hultman and Corvellec 2012), as they limit the downstream impacts of plastics from accruing (Fig. 1).

Complex ecological and societal issues, such as plastics, require a holistic understanding of how interventions affect all dimensions of sustainability, including interventions aimed at mitigating adverse impacts, like PWMS. However, there have been no systematic reviews of the sustainability aspects of PWMS. In this review, we evaluate the current body of literature to identify the impacts of PWMS beyond their stated objectives of reducing plastics and plastic pollution. In other words, we are evaluating the impacts that are unrelated, or external, to the primary goal of PWMS: reducing the impacts of plastics pollution.

Approach

We conducted a semi-systematic review of peer-reviewed literature on PWMS along the plastics life cycle, including bans, levies, and taxes; alternative products; recycling; extended producer responsibility (EPR); waste-to-energy (WTE); and waste recovery (Table S2). We characterized impacts within the context of the sustainable development goals (SDGs) (UN General Assembly 2015). We used the SDGs as they evaluate sustainability as a complex problem, including dimensions of environmental and human well-being (see Table S3). Though we recognize the limitations of the SDG framework to analyze social and environmental sustainability, they are useful to group analysis of several kinds of impacts (Wackernagel et al. 2017). We classified impacts as positive if they promote meeting the SDGs or negative if they are incongruent with the SDGs, relative to the conventional plastics system. In evaluating how PWMS can advance or impede different aspects of sustainability, we demonstrate the need for in-depth evaluations of the potential consequences of PWMS.

We searched Google Scholar and the Oxford University library databases for English-language literature from January 2000 to July 2020 using the search terms in Table S4. Peer-reviewed literature, dissertations, theses and white papers were included. Review papers and policy or commentary pieces were reviewed but excluded from the quantitative analysis. We limited our study to research on PWMS for addressing consumer or "disposable" plastics, which represent ~50% of plastic waste generated globally (Geyer et al. 2017). Several papers were added by snowball sampling from the initial search, or when papers known to the authors were missing (see Table S4).

We did not evaluate landfilling, as we assumed the historic precedent for disposal of plastics is either landfilling or unmanaged disposal. Nor did we evaluate the impact of PWMS on litter, as reducing plastic pollution is an implied objective of all interventions. In instances where a paper evaluated multiple PWMS, we recorded the impacts for each PWMS independently. Impact categories were generated based on the impacts that emerged from the literature; a total of 26 categories were identified. These were aggregated into 13 categories aligning with the SDGs and their indicators (Table S5). Compared to the conventional plastics system, impacts were categorized as positive (e.g., decreased GHG emissions), negative (e.g., increased GHG emissions), neutral (e.g., GHG emissions not affected), or depends on the context of implementation. Here, context refers to assumptions about the system being studied, such as the location of study or the waste management system.

This approach has several limitations. Direct comparisons between studies were not possible because studies differed in research design, methods, and context. For many PWMS, the sample size of studies evaluating specific impacts is quite small, especially for social impacts, such as gender issues. The literature prioritizes certain impacts and neglects other potential impacts. As such, these results are not comprehensive of the potential impacts of PWMS, but rather, a broad assessment of the current state of the literature evaluating impacts. Below, we provide a description of (1) the types of impacts that have been evaluated in the literature and (2) the direction of impacts from this body of research for each PWMS (positive, negative, neutral, and context dependent). Finally, we discuss the limitations and policy implications of this study.

Findings

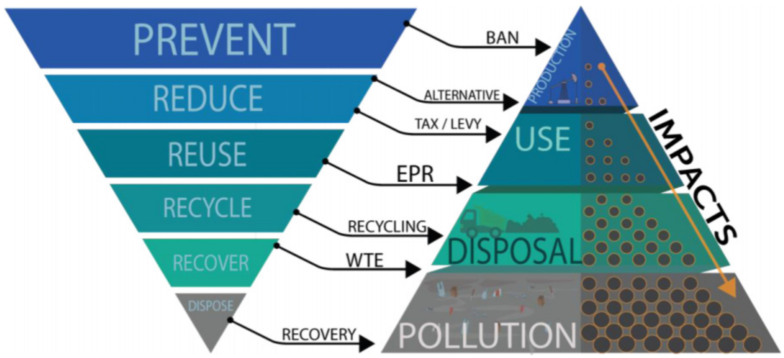
Types of PWMS and their impacts

WTE was the most frequently evaluated PWMS, with 34 studies that met our evaluation criteria and 104 impacts identified. Plastic alternatives received second most attention in the literature with 66 impacts found in 21 studies (Fig. 2). Recycling was the next most evaluated with 23 studies reporting 36 impacts. Sixteen studies evaluated impacts of levies, taxes, and (or) bans, 11 evaluated recovery, and 10 evaluated EPR, reporting 18, 18, and 17 impacts, respectively.

Combined, 40% of impacts were positive, 36% negative, 20% context dependent, and 4% neutral. The distribution of positive and negative impacts varied between PWMS (Fig. 2*d*). As shown in the heatmap, the types of impacts evaluated also varied between PWMS, as shown in Fig. 2*a*–2*c*. For example, studies on WTE tended to evaluate GHG emissions, energy, and air pollution. Studies on alternative products tend to evaluate impacts in multiple categories, due to the nature of the life-cycle analyses (LCA) methods used in the studies evaluating these products (Figs. 2*a*–2*c*).

Most research to-date has focused on a few impact categories, predominately GHG emissions, energy, financial impacts, and pollution (both air and water). GHG emissions were the most evaluated impact by type, followed by financial impacts, environmental and social justice, air pollution, and water (Fig. 3). The distribution of positive and negative impacts

Fig. 1. A conceptualization of the plastic life cycle in the right triangle, encompassing plastic from initial production to disposal and pollution. Management strategies, represented as black arrows, occur at different stages of the plastic life cycle. They can be evaluated with the waste hierarchy, a tool for prioritizing waste management shown as the left triangle. Plastic materials cause impacts that accrue throughout the life cycle, as shown by the increasing number and size of dots in the pyramid on the right. Actions that occur higher on the waste hierarchy, such as actions that reduce plastic, mitigate cumulative impacts of management strategies across the plastic lifecycle. Reduction of plastic avoids impacts from production, use, disposal, and pollution of plastic. For definitions of the plastic waste management strategies assessed here, see the supplementary information.



varied by impact category. For awareness and education, no negative impacts were identified. For GHG emissions, 41% of impacts were positive, whereas for air pollution, only 14% were positive. Overall, impacts that are easier to measure, or are of particular interest, such as GHG emissions or financial impacts, are better researched than other impacts, such as long-term sustainability (Fig. 3). Impacts to environmental and social justice were prevalent, but many of these studies focused on changes in employment, which was coded as a social impact. Fewer studies assessed changes in equity, or distributions of benefits and harms. The presence and measured direction of the impacts depend on several factors, including study design, methods, and context.

Direction of impacts

In this section, we discuss the impacts by PWMS in the order of the waste hierarchy (Fig. 1). For each PWMS, we briefly summarize the state of the literature and implications by impact category. For each impact category, we also state the total number of studies that evaluated the impact, and the percentage of all studies that evaluated that impact. Note that studies could evaluate multiple impacts.

Bans, levies, and taxes

We reviewed 16 studies that evaluated the impacts of bans, levies, and (or) taxes. These studies evaluated a total of 18 impacts. Five were positive, six were negative, five were neutral, and two depended on the context of implementation (Fig. 4). Impacts to long-term sustainability and financial impacts were the most evaluated.

Awareness and education

Three studies (19% of the studies evaluated) reviewed awareness and education outcomes. Martinho et al. (2017) found that a tax had no effect on the perception of marine litter or the impact of plastic bags on the environment and health, even in coastal communities. In contrast, Sharp et al. (2010) found that a ban reduced people's plastic bag consumption and approval for the ban increased postimplementation. The level of consumer education prior to implementation influences how the policy affects consumer awareness (Sharp et al. 2010).

Environmental and social justice

Two studies (13%) reported impacts to environmental and social justice. A study in Zimbabwe reported negative impacts, as retailers and consumers reported being inadequately consulted regarding the ban, leading to resistance to policy implementation (Chitotombe 2014). In Mali, Traore (2013) reports that impacts to gender equality depend on how women are incorporated into plastic ban policy.

Financial

Four studies (25%) reported financial impacts. Two studies reported positive impacts for retailers from avoiding purchase and storage of plastic bags (Convery et al. 2007), and extra revenue from selling reusable bags (Miller 2012). However, one study found negative impacts driven by disproportionately high administrative burdens of levies and taxes for small retailers as compared to large businesses (Killian 2003), while a review found a reduction in competitiveness for small retailers (Oosterhuis et al. 2014).

In another review, Killian (2005) found that Ireland's levy provided landfill cost savings to municipalities by minimizing waste, and South Africa's levy benefitted the tourism industry by reducing pollution, suggesting that the impact of financial burdens may depend on the beneficiary evaluated.

GHG emissions

One study (6%) reviewed the impact of bans, levies, and taxes on GHG emissions. Here, Leeuw (2020) found single-use

Fig. 2. Heat maps present the total number and type of positive (*a*), negative (*b*) and dependent (*c*) impacts identified for each PWMS in the literature review. Neutral impacts are excluded from the heatmap, as few were identified. The bar chart (*d*) shows the total number and direction of impacts for each PWMS. EPR, extended producer responsibility; PWMS, plastic waste management strategy.

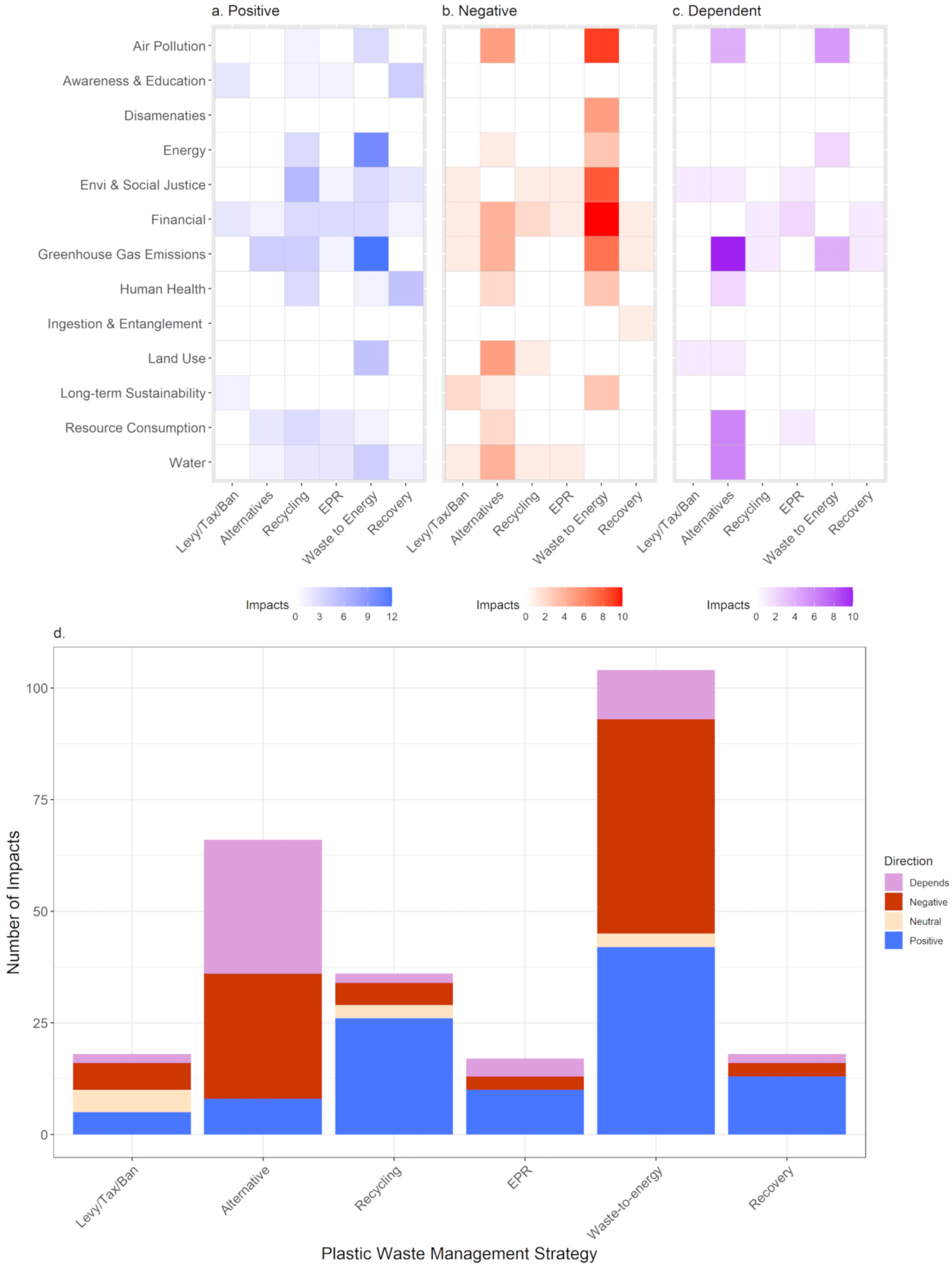
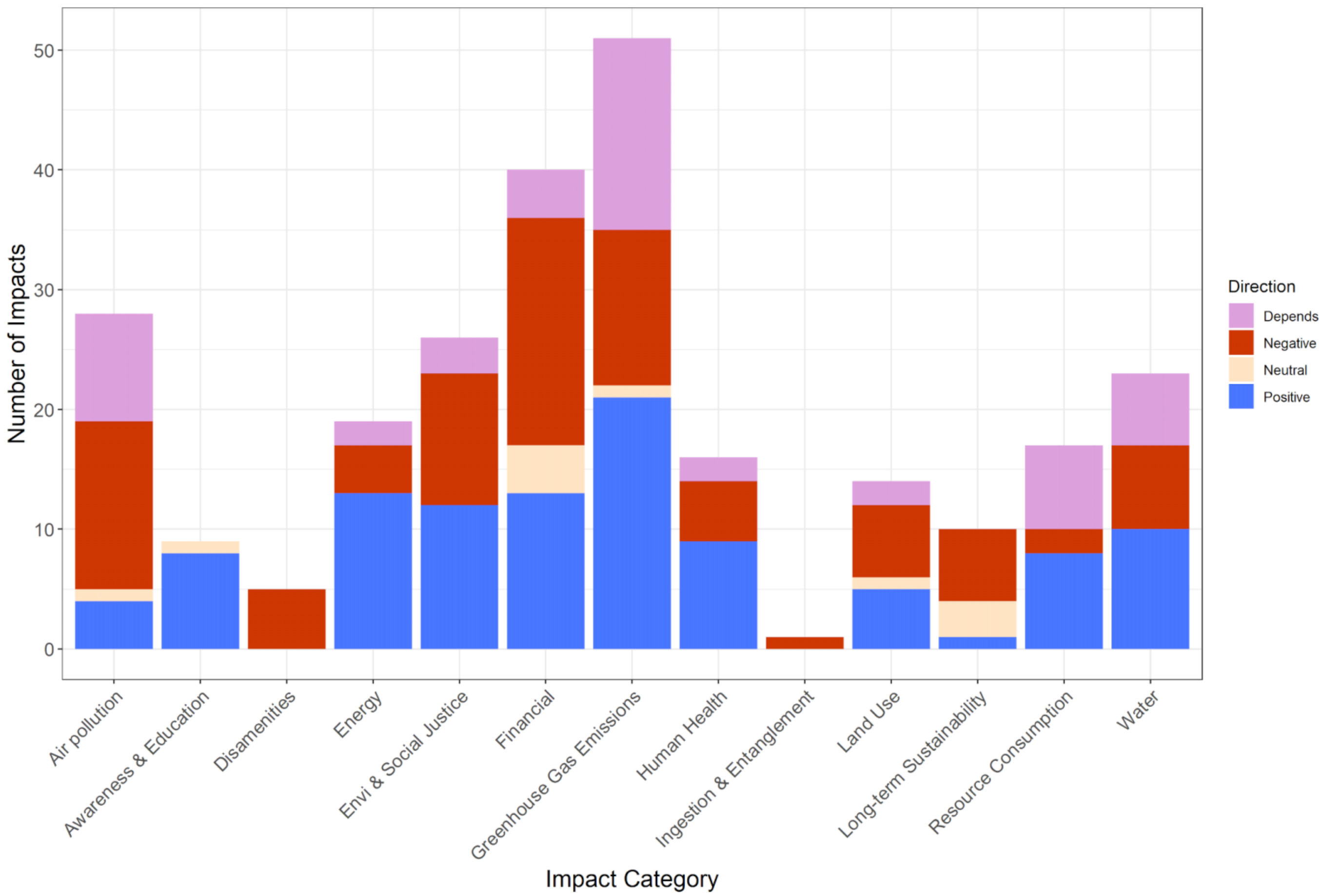


Fig. 3. Total number and direction of impacts by impact category, for all PWMS. PWMS, plastic waste management strategy.



plastic bags generally have lower GHG impacts compared to other materials when assessed through LCA, suggesting a ban would increase GHG emissions (Leeuw 2020).

Land use

One study (6%) examined the land-use impact of bag bans, finding that the alternative replacing plastic bags influenced the outcome (Leeuw 2020). For instance, if paper bags are the alternative, land use could increase for paper production.

Long-term sustainability

Six studies (38%) evaluated the impact on long-term sustainability. An assessment of an upcycling initiative in Indonesia that operates on a levy system found that co-benefits of this initiative, such as woman's empowerment, made the initiative more likely to have long-term success (Bebasari 2019). A plastic bag ban in Australia reduced consumption of single-use polyethylene bags, but these reductions were offset by increased use of thicker "reusable" plastic bags, thus a neutral impact overall (Macintosh et al. 2020).

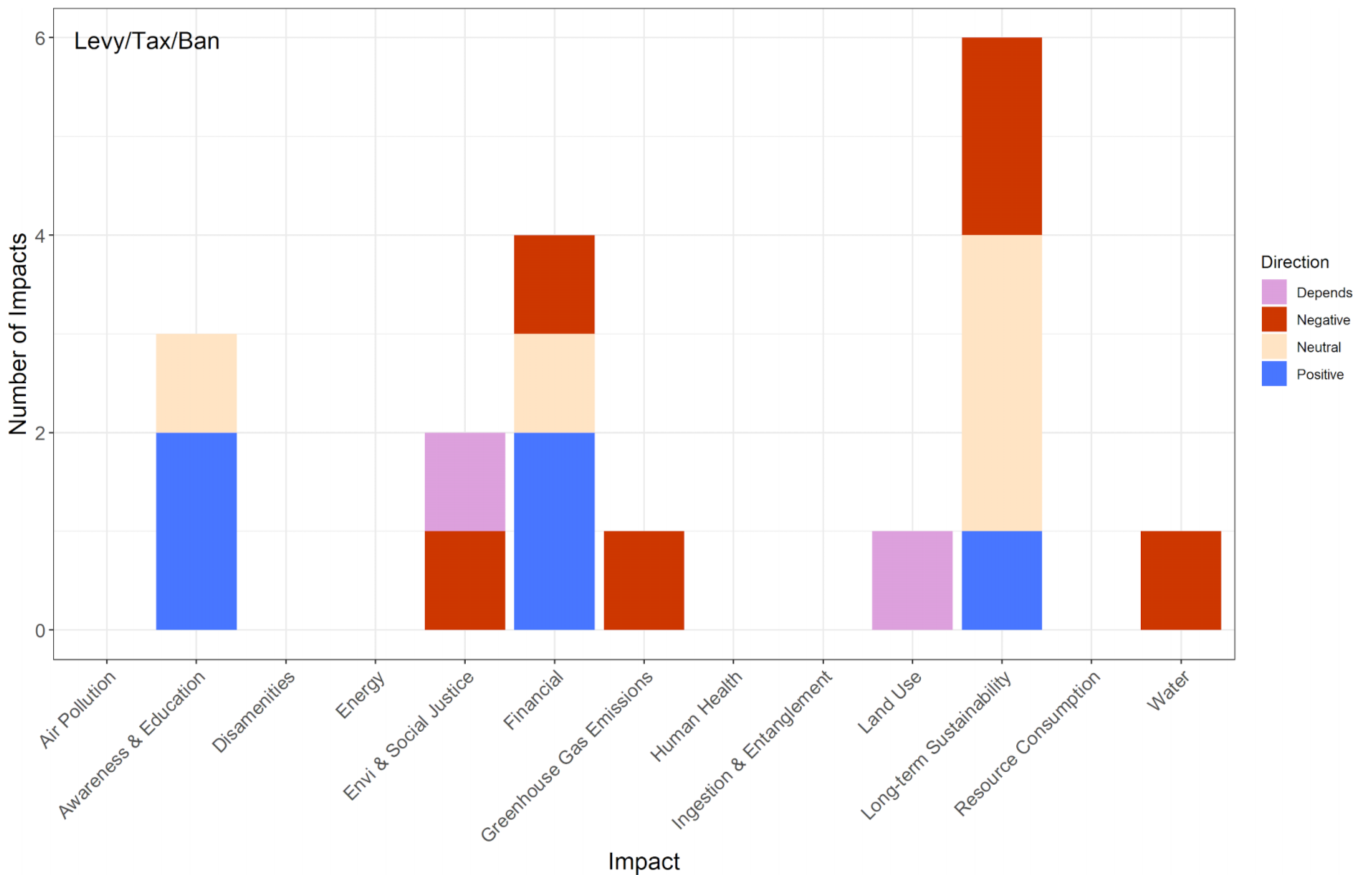
Water

One study (6%) found a ban could increase water use and (or) pollution if alternative products requiring more water, such as paper or cotton, are used (Leeuw 2020).

Bans, levies, and taxes summary

Most of the literature on bans, taxes, and levies applied to plastic bags specifically, which influenced the type of impacts identified. The impacts were influenced by the comparative scenario and the enforcement capacity of the country or region of implementation. Most negative impacts related to administrative costs or the increase in alternative products if plastics were taxed or banned (see the "Alternative products" section). Weakly enforced bans or taxes that do not adequately increase over time are likely to be ineffective and burden non-governmental stakeholders (Convery et al. 2007; Dikgang and Visser 2012). The efficacy of a plastic bag ban in India reduced within one year due to lack of enforcement, suggesting that in countries with limited enforcement capacity, education and awareness campaigns, and increased availability of substitutes should be pursued over bans (Gupta 2011). Similarly, Jiang (2016) reported that reward programs that encourage consumers to use reusable bags in exchange for loyalty points may be more cost effective in the long term than levies or taxes. However, most studies evaluated high income (HI) countries, where enforcement may be more effective. Further studies on the impacts of bans, levies, and taxes in low-middle income (LMI) and low-income (LI) countries, as well as on different plastic products, are needed. Studies also suggested the need to pair levies, taxes, and bans with awareness campaigns, and appropriate, sustainable alternatives.

Fig. 4. Impacts identified for bans, levies, and taxes. Total number and direction of impacts are shown for each impact category.



Alternative products

We reviewed 21 studies evaluating impacts of alternative products. In total, 66 impacts were measured, 8 were positive, 28 negative, and 30 context dependent (Fig. 5). Alternative products were classified into three categories: non-bio-based products ($n = 2$), bio-based nonplastics ($n = 4$), and bio-based plastics ($n = 12$). Non-bio-based products include aluminium and glass. Bio-based non-plastics are products like paper or cotton. Bio-based plastics are polymers with similar properties to plastics but are derived from renewable sources, not fossil fuels. Most studies were LCAs ($n = 17$). Thus, the most identified impacts were impacts that are easily assessed as LCA impact categories, namely, GHG emissions, water pollution and consumption, air pollution, and resource consumption. Overall, the direction of impact depended on the context of implementation.

Air pollution

In the 9 alternative product studies (43%) evaluating air pollution, the direction of impact depended on the alternative and pollutant considered. Paper and cardboard had higher photochemical oxidation, ozone layer depletion, and acidification potential (Abejón et al. 2020; Lewis et al. 2010). However, the direction of the latter could change if the ratio of plastic replaced to paper was high (Sevitz et al. 2003). One study found glass had higher photochemical oxidation, acidification, and ozone depletion potential (Humbert et al. 2009), while another found glass had lower potential in all three categories, provided glass was recycled (Accorsi et al. 2015).

Bio-based plastics also had mixed impacts. One study found lower photochemical oxidation potential (Lewis et al. 2010), whereas another found an increase (Khoo et al. 2010). Bio-based plastics were found to increase acidification potential (Changwichan et al. 2018; Sadeleer 2018), primarily due to agricultural production (Koch and Mihalyi 2018). However, Changwichan et al. (2018) found that acidification potential could decrease for certain polymers from certain feedstocks.

Energy

One study (5%) found that a distribution system using cardboard crates utilized more energy than using reusable plastic containers (Abejón et al. 2020).

Environmental and social justice

In 1 study (5%), Orset et al. (2017) found that wealthier consumers were more likely to purchase products labeled as environmentally friendly, such as alternatives. Although not included in the formal analysis (see the "Methods" section), a review article suggested that bio-based plastics contribute to job creation through the expansion of the bioeconomy. However, the feedstocks are often cultivated in regions with low employment protections and weak governance; thus,

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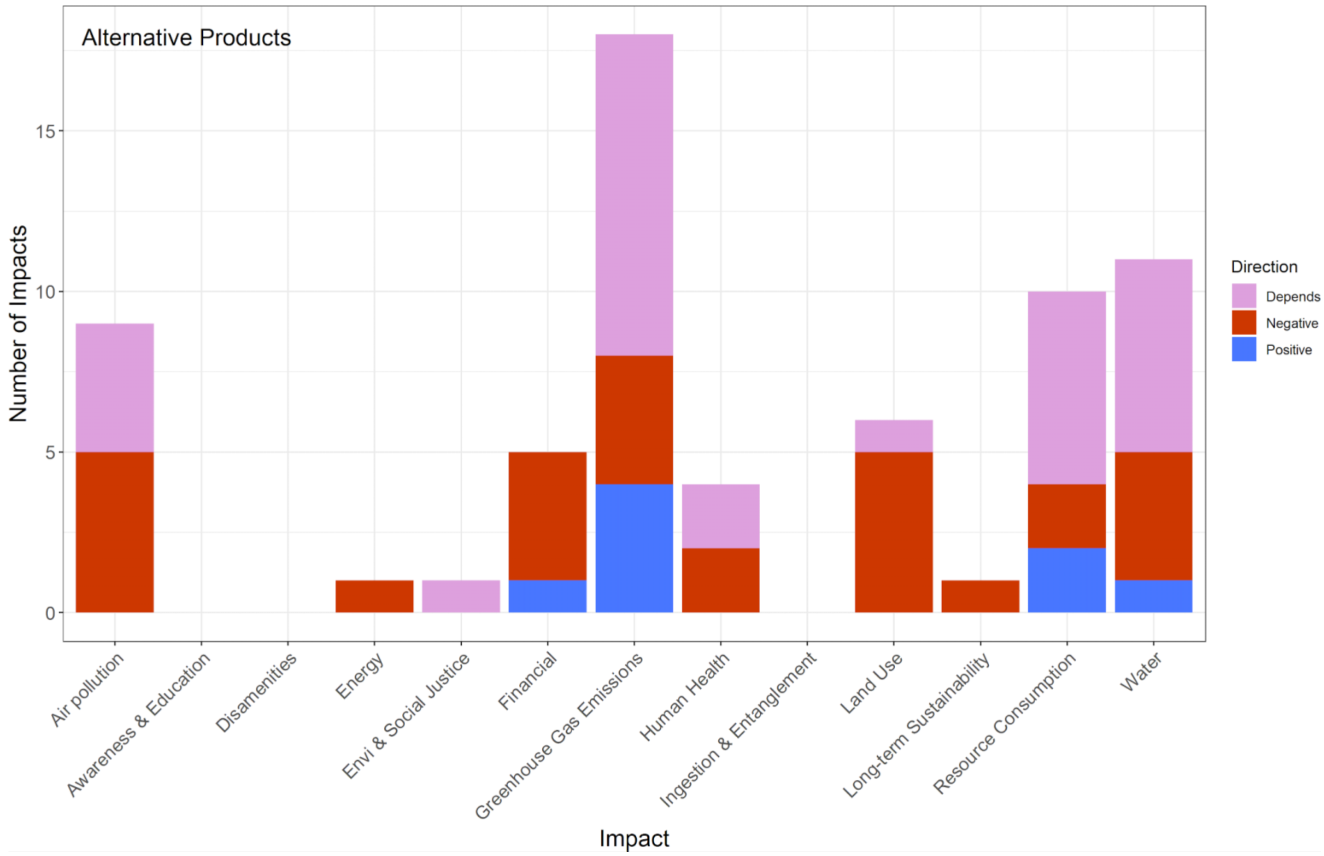
Fig. 5. Impacts identified for alternative products. Total number and direction of impacts are shown for each impact category. The direction of impact strongly depended on the context. Negative impacts were more common than positive impacts overall.



feedstock production could exploit workers or marginalized groups (Spierling et al. 2018).

Financial

Five studies (24%) reported financial impacts. Four studies found negative effects. Three reported that the system of production was more expensive, both for reusable plastic products (Accorsi et al. 2014) and bio-based plastics (Blanc et al. 2019; Changwichan et al. 2018). Escobar et al. (2018) used a global economic model to evaluate a production target for bio-based plastics, and found decreased gross domestic product (GDP), and increased feedstock cost, including for food commodities. However, Lindstrand and Thunell (2017) reported that paper was cheaper than plastic when the externalities of plastic pollution are internalized.

GHG emissions

Eighteen studies (86%) reported GHG impacts. Paper and cardboard had higher GHG emissions than plastic in four studies (Abejón et al. 2020; James and Grant 2005; Lewis et al. 2010; Sevitz et al. 2003). Glass had both positive and negative impacts, depending on recycling rates for the glass and the plastics being replaced (Accorsi et al. 2015; Humbert et al. 2009). Emissions associated with glass were higher due to transportation of the heavier material further distances (Humbert et al. 2009).

For bio-based plastics, several studies found increased GHG emissions (Dilkes-Hoffman et al. 2018; Khoo et al. 2010; Lewis et al. 2010). Bio-based plastic production could lead to increased GHG emissions from land-use change (Escobar et al. 2018). Other studies showed reduced GHG emissions (Blanc et al. 2019; Sadeleer 2018; Spierling et al. 2018). When specific applications such as bags were evaluated, the lowest impact on GHG emissions was reusable plastic bags, provided they were used enough times and the reusable product did not involve significant changes to the delivery system (Accorsi et al. 2015; James and Grant 2005; Lewis et al. 2010). Still, other studies showed mixed impacts, depending on the specific biopolymer produced, the system of production, the agricultural production of the bio-based feedstock, and end-of-life assumptions about both the bio-based plastic and the conventional plastic being replaced (Changwichan et al. 2018; Cheroennet et al. 2017; Dilkes-Hoffman et al. 2018; James and Grant 2005; Khoo et al. 2010; Vogli et al. 2020).

Human health

Four studies (19%) identified impacts associated with human health. Studies found that paper bags (Sevitz et al. 2003) and glass jars (Humbert et al. 2009) would have negative impacts when compared to plastic in LCA categories of respiratory effects and carcinogens. This impact depended on where the respective products were produced. Generally, bio-based plastics were found to have higher toxicity associated with production than conventional plastics (Changwichan et al. 2018; Sadeleer 2018).

Land use

In total, six studies (29%) reported land use impacts. Of these, five studies found higher land use associated with alternative products. Paper and bio-based plastics require land to produce feedstocks, resulting in greater land use (Changwichan et al. 2018; Escobar et al. 2018; Sadeleer 2018; Sevitz et al. 2003). One study found higher land use associated with glass packaging (Humbert et al. 2009).

Long-term sustainability

One study (5%) reported impacts to long-term sustainability. Nazareth et al. (2019) found that the biodegradability of certain bio-based plastics in the natural environment may not be as effective as claimed.

Resource consumption

Ten studies (48%) reported impacts to resource consumption. Several studies demonstrated lower fossil fuel consumption for bio-based plastics (Blanc et al. 2019; James and Grant 2005; Lewis et al. 2010; Sadeleer 2018). Changwichan et al. (2018) found that fossil fuel consumption depends on the feedstock (e.g., sugarcane vs. cassava), the polymer made, and the end-of-life assumptions for both products. Similarly, Vogli et al. (2020) found that resource savings associated with bio-based plastics depend on the polymers being compared (e.g., comparing bio-based polyhydroxyalkanoate (PHA) or polylactic acid (PLA) to conventional polymers).

Paper and cardboard had higher fossil fuel consumption in some studies (Abejón et al. 2020; James and Grant 2005; Lewis et al. 2010). The magnitude and direction of impact, however, depend on the ratio of paper to plastic for substitution (Sevitz et al. 2003). For glass, one study found increased fossil fuel consumption as the glass container was heavier. However, in this study, production of plastic occurred in a distant geographic location, potentially negating any reductions in GHGs from transporting the lighter material (Humbert et al. 2009). Accorsi et al. (2015) reported similar mixed results, with the impact depending on assumptions on the end-of-life management and energy production in the system.

Water

Eleven studies (52%) reported impacts to water. For bio-based plastics, one study found increased water consumption in production, but this impact was negated if the bio-based packaging decreased food spoilage, thereby implicitly saving water (Dilkes-Hoffman et al. 2018). Bio-based plastics typically had higher water deprivation potential, but the direction of impact depended on the bio-polymer produced and where the feedstock was grown (Cheroennet et al. 2017).

Three studies found increased eutrophication potential from production of paper and cardboard (Abejón et al. 2020; Lewis et al. 2010; Sevitz et al. 2003), although this may depend on the ratio of paper to plastic (Sevitz et al. 2003). Humbert et al. (2009) found that glass had higher eutrophication potential and aquatic ecotoxicity than plastic, whereas Accorsi et al. (2015) found lower eutrophication potential, unless significant amounts of plastic were recycled. Three studies that evaluated bio-based plastics all found increased eutrophication potential (Changwichan et al. 2018; Koch and Mihalyi 2018; Lewis et al. 2010). This was primarily from fertilizers used for growing feedstocks (Changwichan et al. 2018; Koch and Mihalyi 2018). Sadeleer (2018) found net lower water pollution with bio-based plastics despite higher production impacts, due to less plastic incineration.

Alternative products summary

Although the direction of impacts varied depending on the context, and the specific alternative evaluated, several patterns emerged. Reusable plastics typically had positive impacts, provided they were used enough (James and Grant 2005; Lewis et al. 2010), although they may increase costs (Accorsi et al. 2014). Overall, alternative products tended to be more expensive. Bio-based products require more land and have impacts associated with agricultural production (Blanc et al. 2019; Changwichan et al. 2018; Escobar et al. 2018).

The degradability of products claiming to be biodegradable in the marine environment is not clear; biodegradability may be less than advertised (Nazareth et al. 2019). Many of the "compostable" bio-based plastics require industrial composting infrastructure, which is not available in many locations (UNEP 2017), eliminating this benefit of "compostable" plastics compared to conventional plastics.

Generalizing the impacts of alternative products compared to plastics is challenging. First, the direction and magnitude of impacts depend on the specific products being compared. Studies comparing final products (e.g., cartons) may find different results than studies comparing polymers (e.g., polypropylene vs. polylactic acid). Second, the context of substitution matters. The energy used in production, transportation of materials, and end-of-life processing all affect impacts, meaning the results of one study, may not be applicable in another country or region where the assumptions made do not apply. Third, the functionality of materials may be different, for example, if packaging affects food waste, GHG emissions from food production would be impacted (Dilkes-Hoffman et al. 2018). Fourth, the bio-based plastics industry is relatively young (Dilkes-Hoffman et al. 2018); production techniques may be improved.

Finally, most studies were LCAs, which typically assessed a common set of impact categories—usually easy-to-assess impacts, such as GHG emissions—in a set of specific circumstances. The outcomes of these studies are thus difficult to generalize to other circumstances. Many LCAs consider the cradle-to-factory-gate production of alternative products, neglecting the end-of-life phase and the impacts of plastic pollution (UNEP 2017). See supplementary materials for more discussion on LCAs.

Recycling

We reviewed 23 papers evaluating the impacts of recycling. Over half ($n = 15$) were conducted in HI countries, 5 were conducted in upper-middle income (UMI) countries, 1 study was conducted in an LMI country, and 2 papers studied multiple countries. These studies found 36 impacts in 10 different impact categories. Twenty-six impacts were positive, five were negative, three were neutral, and two were context dependent (Fig. 6). Financial impacts were the most studied, with the direction of impact varying. Impacts to environmental and social justice were commonly evaluated, with most impacts being positive, related to economic opportunities. Impacts GHG emissions tended to be positive, often through avoided emission of producing virgin plastic or from avoided impacts of incineration.

Air pollution

One study (4%) reported air pollution impacts, finding that recycling generated less air pollution than landfill and incineration (Arena et al. 2003; Perugini et al. 2005). However, there are still toxic emissions from recycling facilities (Vélez and Vélez 2017).

Awareness and education

One study (4%) found that children of formalized recyclers had improved education rates compared to informal recyclers, due to reliability of income (Aparcana and Salhofer 2013).

Energy

Three studies (13%) evaluated impacts to energy. In England and Switzerland, increasing plastic recycling rates had energy-saving benefits by reducing production of virgin products (Krivtsov et al. 2004). Similarly, Huysman et al. (2015) found that closed-loop recycling was more energy and resource efficient than incineration and landfilling. However, the energy impacts of recycling are dependent on the LCA methodology used to evaluate different PWMS (Lazarevic et al. 2010). Neither of these studies assess the energy demand of building or operating recycling facilities, limiting assessment of total energy impacts.

Environmental and social justice

Although numerous studies mentioned impacts to justice, 7 studies (30%) specifically addressed environmental and social justice implications of recycling, suggesting a need for increased study of justice-related impacts across countries. An upcycling initiative in Indonesia led to women acquiring negotiation skills, becoming more active in the community, and gaining confidence (Bebasari 2019). This correlated with poverty reduction and improved well-being for those involved and the community. Taylor (2008) found that community recycling groups can increase local engagement and community cohesion.

An assessment of building a waste management center on an UMI island found that job creation and the sale of recyclables and compost would lead to increased revenue, raising overall quality of life (Jameel 2013). Likewise, a study on waste collectors in an UMI country found that recycling promotes a solidarity economy through creating jobs and educational opportunities for marginalized people, whereas incineration was characterized as profit driven and less labor intensive (Gutberlet 2012).

Financial

Nine studies (39%) examined the financial impacts of recycling. Multiple studies in HI countries found that recycling could reduce costs for local municipalities compared to landfilling (Lavee 2007; Vélez and Vélez 2017; da Cruz et al. 2014), especially when landfill space is limited (Lavee 2007). However, in Germany, recycling was more expensive than landfilling (Wollny et al. 2002). Compared to multistream recycling, single stream has higher costs and may not increase recovery rates (Lantz 2008). Single stream recycling can, however, allow for economies of scale, thereby increasing cost efficiency.

Financial impacts of recycling were affected by the comparison scenario: some studies compared the financial cost between different recycling systems, whereas others compared recycling to different PWMS (Lantz 2008; Wollny et al. 2002). Inclusion of other scenarios, like incineration and biological treatment of waste (Eriksson et al. 2005) influenced the financial impacts of recycling.

The responsibility for investment in recycling also influenced financial impacts (da Cruz et al. 2014). When industry is not paying it's share, a "free rider" dynamic can emerge where the public is burdened with recycling costs. Efficiency of the entire recycling process (e.g., collection, separation, and transport) influenced cost. Consumer behaviour, such as cleaning and sorting of plastics (Vélez and Vélez 2017), can cause inefficiencies that impact financial viability (Ferreira et al. 2012). One study examining household sorting of recycling found higher sorting among wealthier residents, a possible indication that poorer areas lack the time or resources to separate recycling (Briguglio et al. 2016).

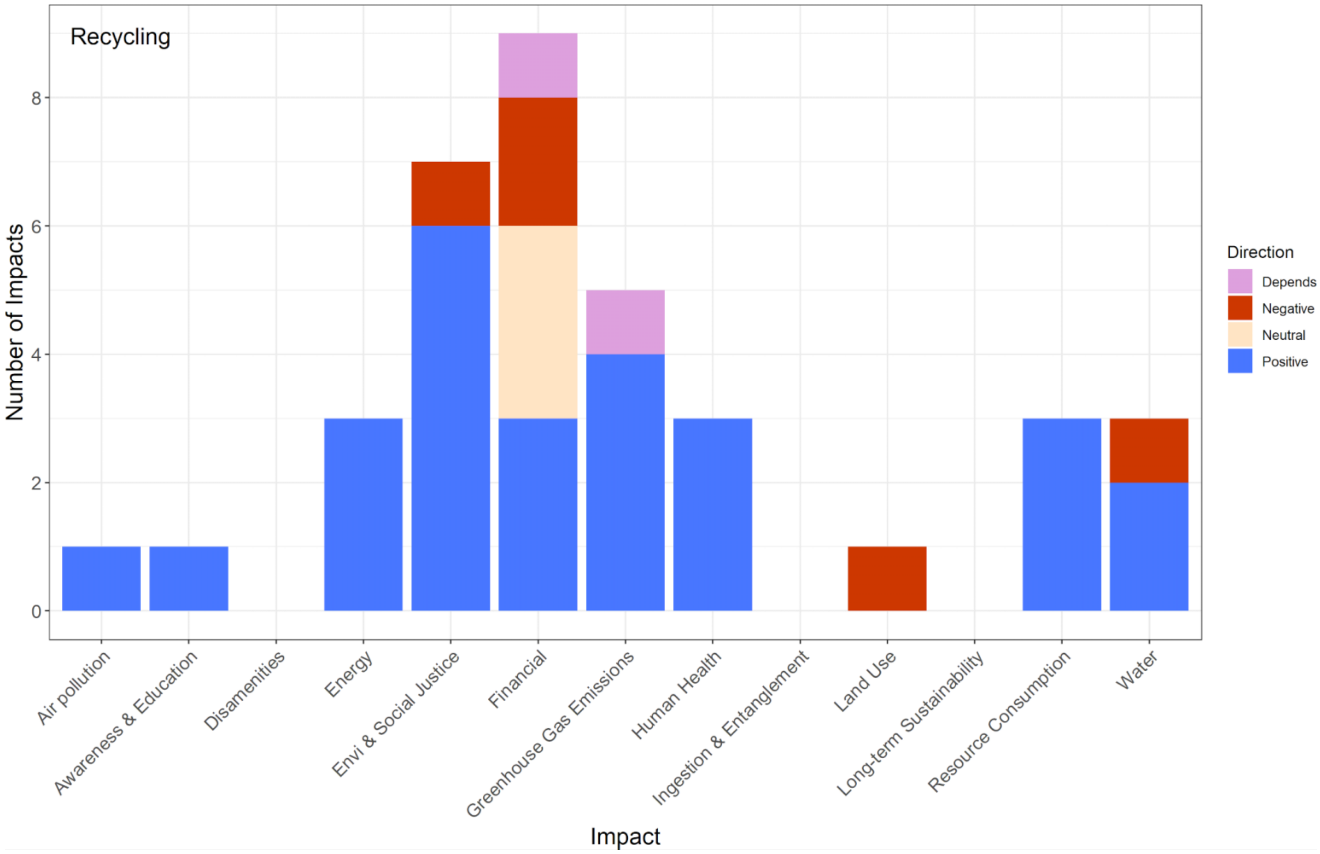
Information and decision-making asymmetries also affect the direction of impacts. Given that the recycling chain is separate from the production of virgin plastics, one firm could manufacture products that increase the cost of recycling for the downstream processor (e.g., multi-layered plastic) (Nicolli et al. 2012). These asymmetries can be alleviated through technological and design innovation. (Nicolli et al. 2012).

GHG emissions

Five studies (22%) reviewed GHG emissions, evaluating the impacts in two ways: the avoided GHGs emissions from producing virgin plastics because of recycling and the GHG intensity of recycling against different PWMS. All studies were conducted in HI countries or in multiple countries of different income statuses; findings are limited in their applicability to emerging economies.

Recycling had lower GHG impacts than other PWMS (Arena et al. 2003; Björklund and Finnveden 2005; Tabata et al. 2011). In a review, Vélez and Vélez (2017) found that recycling emits fewer GHGs than producing virgin plastic. This impact is most notable in high emission industries, like construction and demolition waste (Horsburgh 2013). Park and Gupta (2015) found that recycling is less GHG intensive than WTE in terms of avoided emissions. However, Lazarevic et al. (2010) found that specific features of recycling could alter GHG emissions, such as methods for mechanical recycling or substitution rates between recycled and virgin plastics.

Fig. 6. Impacts identified for recycling. Total number and direction of impacts are shown for each impact category. Most impacts of recycling were positive.



Mechanical recycling paired with feedstock recovery had lower emissions than standard recycling from the avoided emissions of producing virgin plastic polymers (Perugini et al. 2005). When assessed without these energy savings, recycling paired with feedstock recovery was more GHG intensive than mechanical recycling with landfill (Arena et al. 2003).

The scale of analysis affected the direction of impacts. Lower GHG emissions compared to incineration are a benefit to everyone, whereas incineration can provide local energy generation for residents (Park and Gupta 2015).

Human health

Three studies (13%) evaluated impacts on human health. Recycling systems powered by informal workers (predominantly in LMI and LI countries) can lead to socioeconomic improvement workers (Gall et al. 2020). A study on waste pickers found that the sale of recyclable materials allowed informal workers improved access to health programs and insurance (Aparcana and Salhofer 2013). However, a review of the recycling industry found that workers in this industry have a high risk of work-related health problems (Vélez and Vélez 2017).

Land use

Only one study (4%) evaluated land use, finding that building a new recycling facility requires land clearing, having a negative impact on land use (Jameel 2013).

Resource consumption

Three studies (13%) reported positive impacts of recycling on resource consumption, namely, by minimising the fossil fuel extraction necessary for virgin plastic production (Arena et al. 2003; Eriksson et al. 2005; Perugini et al. 2005).

Water

Three studies (13%) evaluated the impacts of recycling on water use or pollution. Two found that recycling decreases water use and pollution (Arena et al. 2003; Perugini et al. 2005). However, the creation of a new waste management facility in an UMI country was potentially impacting groundwater through pollution and dewatering. (Jameel 2013).

Recycling summary

The impacts of recycling were influenced by the study design, including whether the study evaluated operating an existing recycling plant or building a new plant. For new plants, there was increased land use, groundwater pollution, and job creation (Jameel 2013). For existing plants, recycling allowed for energy savings when avoiding the production of virgin plastic polymers, including through plastic feedstock recovery (Perugini et al. 2005). Impacts are affected by collection, sorting, and recycling rates, and the existing recycling infrastructure in place (Sadeleer 2018). Despite the above findings, across ecosystem quality, health, climate change, and toxicity, waste prevention was found to outperform any waste management intervention, including recycling (Cleary 2014).

Extended producer responsibility

We reviewed 12 studies that evaluated impacts of EPR. These studies included take-back schemes such as container deposit legislation (CDL) ($n = 3$) and polluter pays principles ($n = 7$). These studies evaluated a total of 17 impacts across 8 impact categories, 10 were positive, 3 were negative, and 4 were context dependent (Fig. 7). Financial impacts were evaluated most frequently.

Awareness and education

One study (10%) reported impacts to awareness and education. Pollution reflects poorly for a brand, which in turn affects consumer choices to less littered brand items (Roper and Parker 2013).

Environmental and social justice

Three studies (30%) reported environmental or social justice impacts, one positive, one negative, and one context dependent. Benefits of EPR schemes are the reduction of environmental impacts in places with poor waste management systems and (or) high plastic (or waste) imports, by placing the responsibility to manage plastic waste on the industry that produced it. Job creation increased across the collection, sorting, transport, and recycling sectors when EPR is implemented (Lavee 2010). In Denmark, Vigsø (2004) reported significant social costs to CDL because the value of collected material is relatively low, whereas if the material was incinerated or used for WTE plants, more money would be saved in social costs.

Financial

Six studies (60%) evaluated the financial impacts of EPR. Three were positive, one negative, and two context dependent. In Israel, the benefits of a national CDL exceeded the cost of implementation (Lavee 2010). Municipal authorities in Portugal, Belgium, and Italy gained revenue through “pay-as-you-throw” and EU “Green Dot” schemes; however, industry players are still not paying the net cost of waste management (Ferreira et al. 2017; Horsburgh 2013). Disposal fees are generally effective if there is a functioning recyclable market for collected materials (Calcott and Walls 2000). Similarly, savings can arise from the diversion of plastic waste from the waste stream (e.g., collection and landfill) (Ferreira et al. 2017; Lavee 2010). When the private sector controls recovery and recycling processes of EPR programs, the costs can be reduced through maximising efficiency rather than relying on public services, with the added benefit of creating a positive public image (Morden 2019; Roper and Parker 2013).

One study reported negative financial impacts of EPR, finding that when costs are disproportionately subsidized by the public and retail sectors, economic pressure increases on communities and small businesses that do not benefit from economies of scale (Fernie and Hart 2001).

Two studies found financial impacts to be context dependent. While there are financial gains for reducing the volumes of plastic waste being landfilled, EPR schemes may come at a substantial cost to the public sector (Sachs 2006). Effective EPR relies on economies of scale, where the costs associated with implementing the scheme to make products from the recycled materials are less than the costs of producing virgin materials in the first place (Calcott and Walls 2000; Jacobs and Subramanian 2009). With the price of feedstocks for virgin plastic (e.g., oil and fracked gas) remaining low, many EPR systems are rendered too expensive (Associated Press 2019). Similarly, transaction costs may be significant (e.g., collection, transportation, and recycling costs), leading to subsidization by the public sector (Calcott and Walls 2000; Ferreira et al. 2017). CDL can be an efficient option but often comes with high administrative and enforcement costs (Abbott and Sumaila 2019). EPR schemes that include mixed plastic products and (or) a variety of products mean greater levels of bureaucracy are needed, complicating legislation, enforcement, and monitoring of EPR schemes (Abbott and Sumaila 2019; Sachs 2006).

When EPR schemes are designed to be flexible in meeting obligations, via bargaining mechanisms across the private-public sector, such as incentives for “green” product design standards, they can be financially efficient (Abbott and Sumaila 2019; Sachs 2006).

GHG emissions

One study (10%) reported positive impacts to GHG emissions. Using an LCA, Singh and Cooper (2017) showed that an EPR programme for plastic bags in Sweden could yield significant reductions in carbon emissions; however, this was dependent on accessibility to deposit or collection infrastructure and environmental awareness.

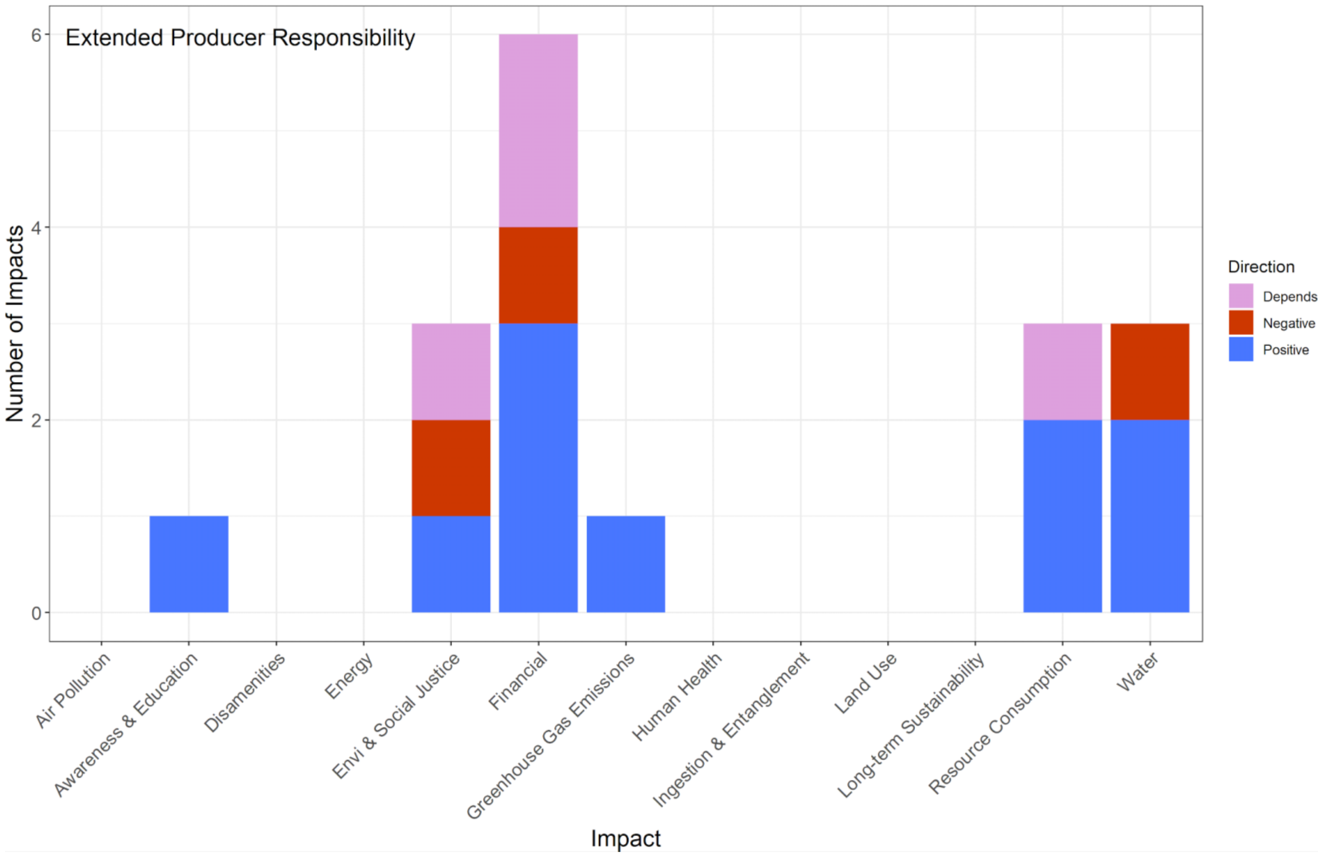
Resource consumption

Three studies (30%) reported impacts related to resource consumption of EPR schemes, two were positive and one context dependent. Singh and Cooper (2017) reported less demand for virgin feedstocks to make plastics due to the recovery of plastic materials through an EPR scheme using an LCA. Lavee (2010) showed less fossil fuel consumption with EPR because recycling is more energy efficient than virgin feedstock extractions. A reduction in fossil-based feedstock extraction was reported by Jacobs and Subramanian (2009). However, without a centralized supply of recyclable feedstock, virgin material demand may increase, lead to a dependency on other materials with higher environmental impacts (Morden 2019), or alternatively, may give “license” to continue unsustainable consumption practices (Sachs 2006).

Water

Three studies (30%) reported impacts to water, two positive and one negative. A reduced water footprint was associated with a modelled plastic bag take-back scheme (Singh and Cooper 2017). Alternatively, untreated materials collected through EPR schemes can negatively contribute to runoff, or potential acid rain from recycling processing plants (Jacobs and Subramanian 2009).

Fig. 7. Impacts identified for EPR. Total number and direction of impacts are shown for each impact category. Impacts were often positive, although many impacts depended on the context of implementation. EPR, extended producer responsibility.



EPR summary

Effective examples of EPR schemes include those that address material design, labeling, reporting, recovery, and recycling (Oosterhuis et al. 2014; Sachs 2006). Generally, success occurs when the product and recovery system is designed and implemented by the producer, incentivizing efforts to achieve maximum efficiency (Morden 2019). However, such approaches are rare and require legislation that ensures costs for design and implementation are not deferred to consumers or governments (Fenton and Sinclair 1996; Sachs 2006). With many actors in the supply chain from raw material suppliers to retailers, identifying the "producer" can be challenging (Fenton and Sinclair 1996). Further, downstream EPR approaches may result in further diversion of responsibility for producers to provide more environmentally sustainable products (Calcott and Walls 2000). Alternatively, CDL can work efficiently in lower-income areas, where there is often a greater incentive to return the containers for cash (Schuyler et al. 2018). Although EPR is framed as a market-orientated environmental policy, establishing effective incentives and legal foundations is difficult; thus, enshrining EPR into legislation can increase the legitimacy of the programme for the public and industry (Fenton and Sinclair 1996).

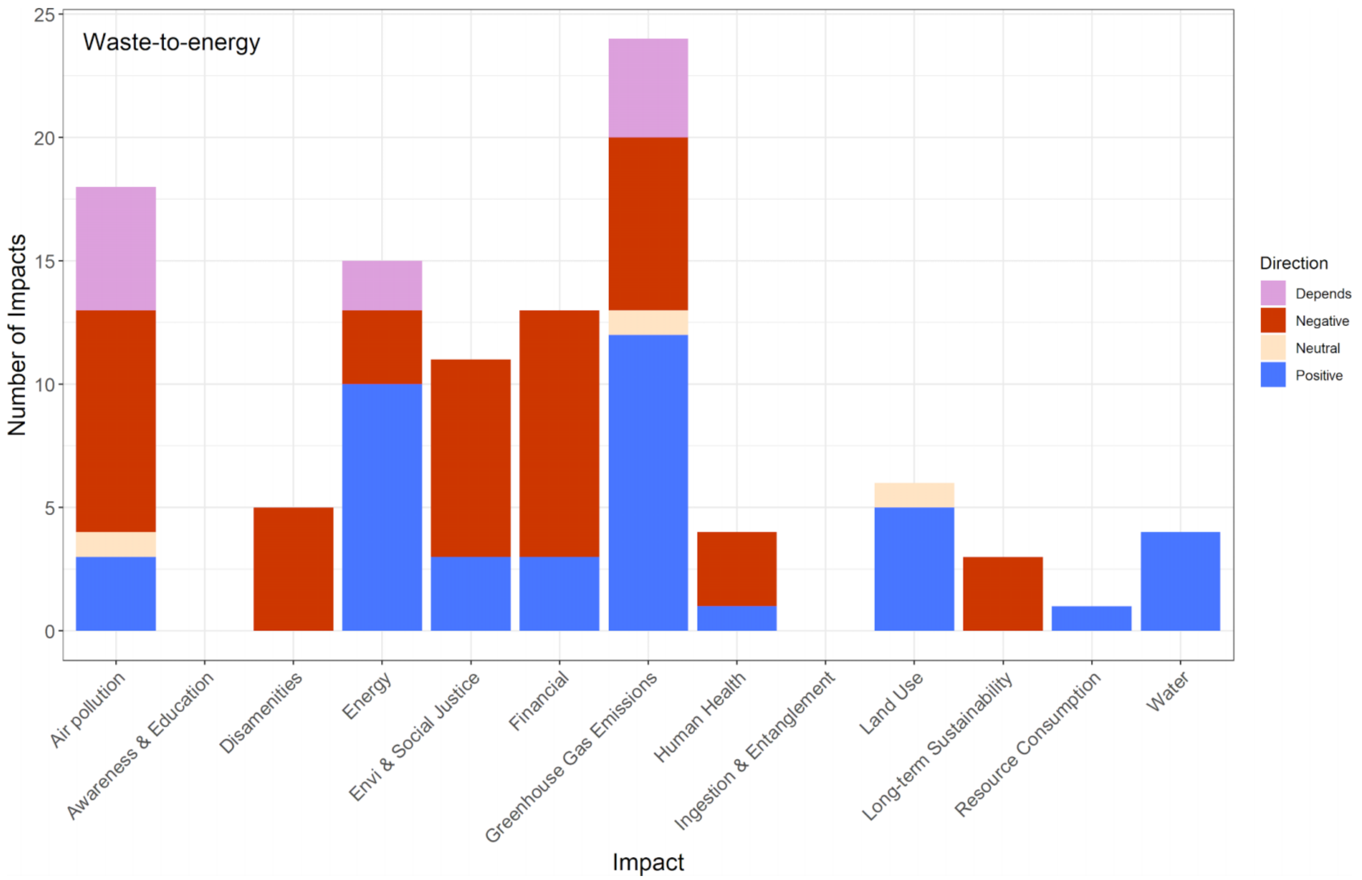
Waste-to-energy

Thirty-four studies evaluated impacts of WTE plants. Twenty-three studies evaluated incineration, one evaluated gasification and ten evaluated multiple types of WTE plants. Methods included LCAs ($n = 7$), contrast analyses ($n = 5$), surveys or interviews ($n = 4$), modelling and data analysis ($n = 11$), meta-analysis ($n = 1$), and mixed methods ($n = 6$). These studies evaluated 11 different impact categories and find a total of 104 impacts. Forty eight were negative, 42 were positive, 3 were neutral, and 11 depended on the context of implementation (Fig. 8). Impacts to GHG emissions were evaluated most frequently, with 50% finding positive impacts. Air pollution impacts were also commonly evaluated, with impacts found to be predominately negative or context dependent. Few studies evaluated the long-term sustainability of WTE; the studies that did all found negative impacts.

Air pollution

Eighteen studies (53%) evaluated impacts of WTE on air pollution. The four that found positive or neutral effects on air quality were conducted in HI countries with stringent regulations and advanced technology (Cucchiella et al. 2017; Haraguchi et al. 2019; Lahl and Zeschmar-Lahl 2018). Nine found WTE reduced air quality (Cucchiella et al. 2017;

Fig. 8. Impacts identified for WTE. Total number and direction of impacts are shown for each impact category. Air pollution and GHG emissions were the most identified impacts. The impacts to disamenities and long-term sustainability were consistently negative. Impacts to water and land use were positive or neutral. GHG, greenhouse gas; WTE, waste-to-energy.



Demaria and Schindler 2016; Islam and Jashimuddin 2017; Luthra 2017). These studies were conducted across income statuses and evaluated multiple air pollutants, including dioxins and furans (Mukherjee et al. 2020; Randhawa et al. 2020; Vanapalli et al. 2019), NOx, and SO_2 (DeAngelo 2004; Dijkgraaf and Vollebergh 2004; Mavrotas et al. 2015), particulate matter (Mavrotas et al. 2015; Randhawa et al. 2020), and heavy metals (Mavrotas et al. 2015). Generally, impacts depended on the air pollutant of interest. For instance, DeAngelo (2004) found the implementation of a WTE plant would produce more NOx but less particulate matter than hauling waste long-distance to landfills. Two studies comparing WTE to fossil fuels found WTE plants produced more dioxins and furans but less ozone depleting emissions (Leme et al. 2014; Thorneloe et al. 2007). Newer technologies (e.g., gasification) may be cleaner; however, these plants are often financially infeasible (Mukherjee et al. 2020).

Disamenities

Five WTE studies (15%) evaluated disamenities—smell, noise, and unsightliness. All five found WTE plants negatively affected local communities (de Bercegol and Gowda 2019; DeAngelo 2004; Kemal 2007; Lahl and Zeschmar-Lahl 2018; Mavrotas et al. 2015). Kroll (2013) found disamenities were avoided by placing the plant in a manufacturing district, far from residential communities.

Energy

Fifteen studies (44%) evaluated the impact of WTE on energy. The impact depended on the operating efficiency of the plant, the quality of municipal waste and the alternative sources of energy available (Chen and Chen 2013). Most studies that compared WTE to coal or landfill gas recovery found WTE was a cleaner energy alternative (DeAngelo 2004; Dijkgraaf and Vollebergh 2004; Kemal 2007; Kroll 2013; Leme et al. 2014; Thorneloe et al. 2007; Vanapalli et al. 2019). Some reviews noted well-placed plants could provide energy to remote or marginalized communities (AlQattan et al. 2018; Pan et al. 2015). However, three studies found WTE had negative impacts and two depended on the context of implementation. In some instances, plants did not produce enough reliable energy for the community (de Bercegol and Gowda 2019; Haraguchi et al. 2019; Kornberg 2019). Indeed, plants often use auxiliary energy sources to achieve full incineration, reducing their efficiency (de Bercegol and Gowda 2019; Kornberg 2019). Furthermore, plants are more efficient when running at capacity, which demands waste production be maintained, creating a technological lock-in situation.

Finally, none of these studies compared WTE to renewable energy sources (e.g., solar), which may influence the cleanliness and affordability of WTE.

Environmental and social justice

Eleven studies (32%) evaluated environmental and social justice with 10 focusing on labor. Three found WTE plants created jobs, predominantly for skilled laborers (Cucchiella et al. 2014; Hoang and Fogarassy 2020; Kemal 2007). However, the rest found WTE plants negatively compete with the informal waste sector (de Bercegol and Gowda 2019; Demaria and Schindler 2016; Gutberlet 2012; Kornberg 2019; Lahl and Zeschmar-Lahl 2018; Luthra 2017), which is more prevalent in lower-income communities.

Only one study directly assessed environmental justice and found stakeholders have different access to information and power over implementation (Behrsin 2020). Overall, the implications for environmental justice depend on the context of implementation, but in practice, WTE plants are more likely to be in marginalized communities; therefore, these communities are more likely to suffer the negative impacts of WTE (Mukherjee et al. 2020).

Financial

Thirteen studies (38%) evaluated the financial impacts of WTE. Ten of these found negative impacts. WTE was costly to the city (Luthra 2017; Thorneloe et al. 2007) and the public (Hoang and Fogarassy 2020). In some instances, cities must pay WTE companies if they do not produce enough waste to supply the plant (Chatterson 2018). Operational plants have shut down because they were not profitable (Peart 2016) or are not yet scalable (Münster and Lund 2010). Vanapalli et al. (2019) found WTE was particularly costly for developing nations, due to high capital costs and wages for skilled laborers. Alternatively, three studies, all conducted in the USA, found positive economic impacts of WTE, citing WTE as a source of economic development (DeAngelo 2004; Kroll 2013; Miranda and Hale 2005).

GHG emissions

Twenty-four WTE studies (71%) evaluated impacts on GHG emissions. Twelve reported decreased emissions, seven reported increased emissions, one found no effect, and four found context-dependent emissions impacts. An important factor in these differences was the comparison scenario. When WTE was found to reduce GHG emissions, it was typically compared to fossil fuel energy or landfill gas recovery (Cucchiella et al. 2017; Haraguchi et al. 2019; Lahl and Zeschmar-Lahl 2018). Two studies found WTE plants reduced GHG emissions by reducing the amount of fuel used for the transport of waste to landfills (Kemal 2007; Mavrotas et al. 2015). Vanapalli et al. (2019) concluded the WTE would have no influence on GHG emissions, since plastics are an oil product, which would eventually be burned for energy regardless. Finally, studies comparing WTE to recycling found WTE would increase GHG emissions (Eriksson et al. 2005; Park and Gupta 2015). Chatterson (2018) concluded that a WTE plant in Honolulu, Hawai`i increased GHG emissions because the demand for feedstock hindered implementation of zero-waste initiatives.

Four studies found that impacts depended on the context of implementation. This affected the quality of waste feedstock and technology available, which significantly alter GHG emissions from WTE plants (Chen and Chen 2013; Islam and Jashimuddin 2017). In India, WTE increased GHG emissions since auxiliary fuel is needed to fully combust waste with high-moisture content (de Bercegol and Gowda 2019; Kornberg 2019).

Human health

Four WTE studies (12%) evaluated impacts on human health. Two studies found WTE had negative effects on human health in India, primarily related to increased air pollution (Demaria and Schindler 2016; Randhawa et al. 2020). One found positive health benefits of WTE, due to reduced air pollution compared to fossil fuel-based energy production. This study was conducted in the USA, where stringent environmental standards are more likely met (Thorneloe et al. 2007). Finally, one study found increased health risks from the creation of hazardous chemical waste (Dijkgraaf and Vollebergh 2004).

Land use

Six WTE studies evaluated land use impacts (18%). Five found reduced land use from reducing the volume of waste sent to landfill (de Bercegol and Gowda 2019; DeAngelo 2004; Dijkgraaf and Vollebergh 2004; Hoang and Fogarassy 2020; Vanapalli et al. 2019). Chatterson (2018) was the only exception, indicating land use was not reduced because the landfill has remained in operation. However, they did not explore how landfill life expectancy may change future land use.

Long-term sustainability

All three WTE studies (9%) that evaluated long-term sustainability deemed WTE unsustainable (Chatterson 2018; Peart 2016; Vanapalli et al. 2019). This was primarily due to dependence on waste production, which has forced some countries to import waste to maintain feedstocks (Olovsson and Hein 2018).

Resource consumption

One study (3%) found that adding incineration to waste management made the management more resource efficient (Arena et al. 2003).

Water

Four WTE studies (12%) reported impacts to water. Those that considered impacts of WTE on water quality found that water pollution decreased with the implementation of WTE plants, by reducing leaching from landfilled waste (Leme et al. 2014; Mavrotas et al. 2015; Thorneloe et al. 2007).

WTE summary

Consistently positive effects were seen for two impact categories: land use and water, while negative effects were measured for disamenities and long-term sustainability. For all other categories, there was little consensus in the literature, with the context of implementation and comparison scenario influencing impacts. Waste with high percentages of organic matter and high moisture content is harder to combust, sometimes requiring auxiliary fuels. This influences energy production, GHG emissions, air pollution, and subsequently health impacts and disamenities to local communities (Chen and Chen 2013; Dijkgraaf and Vollebergh 2004; Mukherjee et al. 2020; Vanapalli et al. 2019). Stringent regulations and advanced technology reduce emissions of air pollutants, GHGs, and water pollutants from WTE plants and landfills—the destination for bottom and fly ash produced (AlQattan et al. 2018; Cucchiella et al. 2017; Lazarevic et al. 2010; Mukherjee et al. 2020). National management and larger centralized facilities may also be more efficient and cheaper than municipal management of WTE plants (Pan et al. 2015; Peart 2016).

Recovery

Eleven studies evaluated the impacts of coastal ($n = 7$) and ocean clean-ups ($n = 4$). A total of 18 impacts were identified; 13 were positive, 3 were negative, and 2 depended on the context of implementation (Fig. 9). Ninety-one percent of coastal clean-up impacts were positive (10 of 11), compared to only 43% of the ocean clean-up impacts (3 of 7). Two of the ocean recovery studies were assessments of The Ocean Cleanup, an organization developing technologies to remove plastic pollution from oceans. Positive impacts frequently related to community cohesion, education, and awareness through clean-ups. However, these impacts were compared to a non-intervention scenario, rather than comparing them to the value of awareness and education from other PWMS (e.g., recycling or EPR schemes). No studies assess the long-term sustainability of plastic recovery efforts.

Awareness and education

Four studies (36%) found that beach clean-ups increase public awareness (Lachmann 2016; Cecconi 2019). Busch (2019) found that after participating in a plastic recovery activity, people felt motivated to address the plastic pollution crisis in their lives. However, while those who participate in a beach clean-up are more likely to do it again, they can also become demotivated by the presence of litter (Lucrezi and Digun-Aweto 2020).

Environmental and social justice

Two studies (18%) reviewed implications on environmental and social justice. A case study in Mexico found that recovery initiatives increase woman's involvement in community sustainable development (Hanson 2017). Van Giezen and Wiegmans (2020) found that The Ocean Cleanup could be optimized to create jobs by diverting collected waste to be sorted and reused.

Financial

Three studies (27%) evaluated financial impacts. One consistently negative impact was the cost of implementation. The Ocean Cleanup was estimated to cost between €490 and €700 billion annually from 2020 to 2030 to achieve a 25% reduction in the level of plastic debris assessed in the world's oceans in 2010 (~80 million tons; Cordier and Uehara 2019; Geyer et al. 2017; Jambeck et al. 2015). There were concerns that ocean recovery could be an unaffordable ongoing investment for private and public actors. Additional costs (e.g., sea transportation and port handling costs), transport routes to the mainland, and the composition of waste influenced operating cost (van Giezen and Wiegmans 2020). However, operations could be optimized to yield high returns on recycled plastic (van Giezen and Wiegmans 2020).

Positive financial impacts were primarily associated with businesses using recovered plastic for new products. However, these studies did not consider the costs of recovery operations, which may be subsidized by the public. Furthermore, financial impacts are dependent on the comparative scenario. For example, Cordier and Uehara (2019) found that relying only on The Ocean Cleanup would be more expensive than preventative interventions, such as waste reduction or improved waste management.

A study found a higher willingness to pay for products that recognizably incorporated ocean plastics, suggesting brands could design promotional strategies based on ocean plastic recovery (Magnier et al. 2019).

GHG emissions

Two studies (18%) reviewed GHG emissions. One found high transport emissions for a remote beach clean-up; 120 passengers produced 268 tonnes of GHG emissions for a beach clean-up that removed 500 kg of debris (Lachmann 2016). The other found it depends on the context of implementation (van Giezen and Wiegmans 2020).

Human health

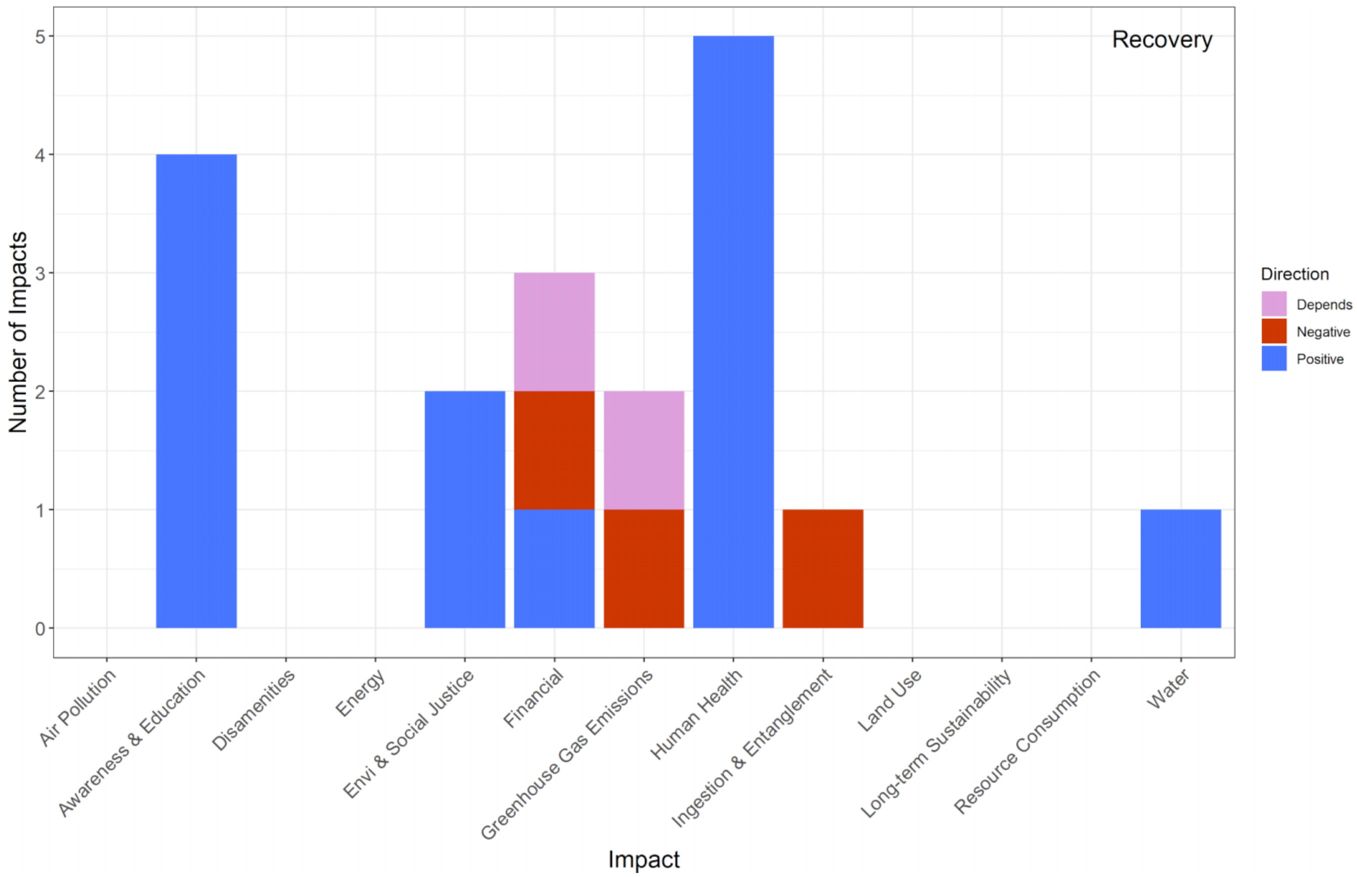
Five studies (45%) evaluated human health impacts. Hanson (2017) found coastal clean-ups improved water quality and sanitation, reducing water-borne diseases. Given that bioaccumulation of plastic in marine food webs can affect humans, recovery of waste can positively impact human health (Hanson 2017; Morrison et al. 2019).

Two studies linked participation in coastal clean-ups to benefits in well-being. Wyles et al. (2017) found time spent near the sea was restorative. Another study found improved mental health among veterans participating in beach clean-ups (Liebengood 2020).

Ingestion and entanglement

Though studies recognized the need to reduce the harm of marine debris on wildlife, 1 study (9%) was critical of ocean recovery projects due to the impacts on ingestion and entanglement. In a case study evaluating The Ocean Cleanup, Morrison et al. (2019) expressed concern that the machinery deployed to recover ocean plastic could unintentionally remove surface-living and free-floating organisms vital for ecosystem function. The impacts of ingestion and entanglement mortality were influenced by how and where ocean recovery devices are deployed. Floating booms deployed near to or onshore were more effective at collecting debris and less harmful to wildlife than those deployed in the ocean (Morrison et al. 2019).

Fig. 9. Impacts identified for recovery. Total number and direction of impacts are shown for each impact category. Impacts were often positive, especially for awareness and education and human health.



Water

Hanson (2017), the only study (9%) to evaluate the impact of coastal clean-ups on water, found improved water quality.

Recovery summary

Overall, impacts for coastal recovery were positive, though impacts for ocean recovery were less consistent. In many studies, there was no alternative intervention presented, and the predominant alternative scenario was no plastic recovery. For example, even where negative impacts were recorded with regards to The Ocean Cleanup, there was a recognition that ingestion and entanglement mortality may be higher in the absence of any intervention. Overall, the effectiveness of coastal clean-ups depended on the framing of the issue and incentives offered to participants (even "fun" could be an incentive) (Lucrezi and Digun-Aweto 2020).

Discussion

We reviewed the literature evaluating the sustainability impacts of PWMS from 2000 to 2020. This review is timely for two reasons. First, it provides an initial evaluation of the research on PWMS and their impacts, illustrating the extent to which these additional impacts are considered in research on plastic pollution mitigation. Second, it reveals critical factors that should be considered when researching and deploying PWMS to avoid adverse outcomes for sustainability. Even if PWMS have negative impacts, the overall trade-off of utilizing PWMS may be positive if the benefits of reducing the impact of plastics is greater than the impacts of the PWMS themselves. Indeed, the presence of a negative impact should not a priori discredit a potential PWMS. Analysis that compares the benefits of reducing plastics and the impacts of implementing PWMS is necessary for informed decision-making that leads to long-term sustainability. Several factors limit generalization of impact direction and type, including study methods, context, and comparison scenarios. Additionally, there is a lack of research for many of these impacts and for many of the PWMS. This review should be considered a survey of the current literature on PWMS, and not a comprehensive evaluation of all potential impacts related to PWMS. Below we discuss some of the limitations, challenges, and opportunities of this study.

Methods

Each of the studies reviewed was conducted to assess specific impacts. Thus, certain impacts are more represented in the literature, and other impacts are neglected or ignored. The number of impacts identified in this review is not an indication of magnitude of impacts, but rather an indication of the amount of attention that impact has received in the

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literature. Impacts that are difficult to analyse are likely underrepresented in this review.

Additionally, consideration of a study's methods is important for understanding which impacts are identified and the direction of those impacts. For example, LCAs reveal different impacts than case studies. In LCAs, the total material requirements and impacts of a product are evaluated in specific impact categories chosen by the researcher—such as GHG emissions or non-renewable energy use. By totalling these impacts per unit of product, LCAs can estimate the impacts and requirements of each product process and compare these results (UNEP 2017). With this methodology, certain impacts (e.g., GHG emissions) are easier to estimate than others (e.g., social impacts). LCAs are also context-specific, evaluating the specific process or product chosen by the researcher, such as one factory, thereby preventing generalization (Antelava et al. 2019; Parra Paitan and Verburg 2019).

Alternatively, case studies provide a detailed examination of a specific management strategy or group of strategies. The researcher may focus on single impact or attempt to provide a more comprehensive analysis of impacts. Unlike LCAs, which follow a standardized methodology, case studies are more flexible; therefore, the impacts revealed depend on decisions made by the researcher. Further, the level of detail provided by case studies may be high, and context specific, limiting broad generalization or comparability (Yin 2015).

Context

The presence and measured direction of the impacts depends on the context of the study. Here, context refers to assumptions about the system being studied, such as the location of study, the plastic delivery system, and waste management system. In some instances, the income status of the country of implementation influenced the impacts identified. For example, the social impacts of WTE may depend on how the informal waste-picking sector is affected by competing with WTE. As informal waste picking is more prevalent in UMI, LMI, and LI countries and less important for HI economies, the context of the country will influence how WTE impacts the local economy (de Bercegol and Gowda 2019; Kornberg 2019).

Comparison scenario

Many studies depend on comparison scenarios to assess impacts. The scenario a PWMS is compared to influences the type and direction of impacts identified. This could be a comparison to a baseline (i.e., present or past scenario), a counterfactual (i.e., scenario that would have occurred without intervention), or an alternative scenario designed for comparison. For example, in models evaluating GHG emissions of WTE, WTE was often compared to coal-fired power plants or landfill gas—the historic baseline—but not compared to alternative energy sources (Thorneloe et al. 2007; Vanapalli et al. 2019). These may be the current baseline for WTE, but since plants operate for decades, it is possible that different energy sources will exist in the future, such as solar and wind, which would influence the relative impact of WTE as a PWMS.

Limitations, future research, and implications for policy

Our analysis identifies all impacts that were measured by studies without considering the magnitude or distribution of impacts, net impacts (i.e., trade-offs between all impacts of a given intervention), or impacts of PWMS on other interventions. In our analysis, studies that identified impacts for a given PWMS were coded equally, even if the magnitude of impact was different. For example, the development of a recycling facility may have a measured impact on land use, but this is smaller in scale and scope than the impact of growing feedstocks to produce alternative products (Escobar et al. 2018). Understanding the magnitude of impacts is necessary to compare PWMS and is ultimately important for decision making.

In addition to considering magnitude, future research should also consider the the net impacts of PWMS. The net impact depends on both the direct and indirect impacts. For example, when evaluating impacts to freshwater ecotoxicity, Sadeleer (2018) found that bioplastics can have higher production impacts but lower overall impacts. This is due to decreased incineration of conventional plastics when bioplastics are being used. According to Sadeleer (2018), incineration of plastics has higher impacts than the production impacts of bioplastics; therefore, there is a net benefit to using bioplastic, despite higher production impacts. In these instances, an intervention with negative impacts may be beneficial overall.

Importantly, we did not explore the distribution of impacts, which is critical to social and environmental justice. The impacts of plastic may affect different parties than the impacts of PWMS, and the impacts of different PWMS may also affect different parties. For example, a coastal community could benefit from the construction of an inland WTE plant due to decreased plastic pollution. However, the inland community would incur the negative impacts associated with proximity to a WTE plant. Consideration of the distribution of impacts is necessary for realizing the SDGs.

Interventions may also compete with one another. This matters when one intervention is preferred. For example, when plastics are recyclable, recycling is typically preferred over WTE, but several studies found that WTE will compete with, and likely disincentivize, plastic recycling (Baxter et al. 2016; Peart 2016; Vanapalli et al. 2019).

Few studies were directly comparable, due to differences in research design or context. Impact categories are also not directly commensurable, as they differ in type and are distributed differently. For many impacts, the sample of studies is too small to draw significant conclusions. Additionally, the literature tends to prioritize impacts that are easy to measure, often neglecting difficult-to-measure impacts, such as long-term sustainability or social justice impacts. Therefore, future research should consider interventions within their specific context of implementation, and qualitative impacts such as social justice, in addition to quantitative impacts to determine the magnitude and distribution of impacts among stakeholders.

Our review demonstrates that PWMS have impacts of their own, beyond their intended goal; consideration of these impacts is necessary for designing sustainable policy. However, broad generalizations regarding the impacts of PWMS are difficult based on the studies conducted to date. Current research has focused on reducing plastic pollution, not on the additional impacts that PWMS may incur. When studies consider these impacts, they generally focus on a subset of impacts of interest, such as GHG emissions and financial impacts. This greater representation in the literature does not necessarily reflect higher importance of these impacts over others. Policy-makers should consider each intervention and all possible impacts, within the specific context of implementation.

Conclusion

Plastic waste management strategies designed to manage or modify the current plastic system are not without their own impacts. This review evaluates the state of the literature regarding these impacts, based on categories informed by the SDGs. Our review demonstrates that there are many strategies to improve plastic waste management, but impacts of each strategy should be carefully evaluated in the specific context of implementation. Currently, there are significant gaps in the literature. Therefore, studies designed to evaluate and compare PWMS effectiveness to their impacts are needed to better inform plastic pollution mitigation policy. Accounting for the externalized, or indirect, impacts of PWMS is imperative for designing just, equitable, and sustainable policy for future plastics use in society.

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Supplementary material

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