PLOS SUSTAINABILITY AND TRANSFORMATION

RESEARCH ARTICLE

# Using life cycle assessments to guide reduction in the carbon footprint of single-use lab consumables

**Isabella Ragazzi**[1]*, **Martin Farley**[2], **Kate Jeffery**[3], **Isabela Butnar**[1]

**1** Institute for Sustainable Resources, The Bartlett School of Environment, Energy and Resources, University College London, 14 Upper Woburn Pl, London WC1H 0NN, United Kingdom, **2** Sustainable UCL, University College London, London, United Kingdom, **3** School of Psychology & Neuroscience, College of Medical, Veterinary and Life Sciences, University of Glasgow, Glasgow G12 8QB, United Kingdom

* isabella.ragazzi.21@ucl.ac.uk

## Abstract

Scientific research pushes forward the boundaries of human knowledge, but often at a sizable environmental cost. The reliance of researchers on single-use plastics and disposable consumables has come under increased scrutiny as decarbonisation and environmental sustainability have become a growing priority. However, there has been very little exploration of the contribution of laboratory consumables to 'greenhouse gas' (GHG) carbon emissions. Carbon footprint exercises, if capturing consumables at all, typically rely on analyses of inventory spend which broadly aggregate plastic and chemical products, providing inaccurate data and thus limited insight as to how changes to procurement can reduce emissions.

This paper documents the first effort to quantify the carbon footprint of common, single-use lab consumables through emission factors derived from life cycle assessments (LCAs). A literature review of LCAs was conducted to develop emission factors for lab consumables, considering the emission hotspots along each product's life cycle to identify where emission reduction policies can be most effective. Results can be used as inputs for lab practitioners seeking to understand and mitigate their carbon footprint.

## Author summary

While science research has been integral to pushing forward the boundaries of knowledge, laboratory activities can be environmentally damaging and contribute to greenhouse gas (GHG) emissions. Some initiatives have estimated the direct emissions and electricity use of science facilities, but understanding the impact of single-use products in laboratory settings has not yet been explored. We present the first effort to determine the emissions associated with the manufacture, use, and disposal of common lab consumables. Our estimates are based on findings from a focused literature review of Life Cycle Assessments (LCAs), studies that assess the GHG emissions of each stage of a product or system's life. The LCA-based emission factors can be used to identify where the carbon impact can be

**Citation:** Ragazzi I, Farley M, Jeffery K, Butnar I (2023) Using life cycle assessments to guide reduction in the carbon footprint of single-use lab consumables. PLOS Sustain Transform 2(9): e0000080. https://doi.org/10.1371/journal.pstr.0000080

**Editor:** Jose Carlos Báez, Spanish Institute of Oceanography: Instituto Espanol de Oceanografia, SPAIN

**Received:** June 13, 2023

**Accepted:** September 11, 2023

**Published:** September 29, 2023

**Peer Review History:** PLOS recognizes the benefits of transparency in the peer review process; therefore, we enable the publication of all of the content of peer review and author responses alongside final, published articles. The editorial history of this article is available here: https://doi.org/10.1371/journal.pstr.0000080

**Data Availability Statement:** Data is provided in the Supporting Information (S1 Appendix).

**Funding:** This research was funded in whole by the Wellcome Trust [Grant number 204841/Z/16/Z], received by K.J. Funding was used to support the salary of the corresponding author, Isabella Ragazzi. The funders had no role in study design, data collection and analysis, decision to publish, or preparation of the manuscript. For the purpose of open access, the author has applied a CC BY public copyright licence to any Author Accepted Manuscript version arising from this submission.

**Competing interests:** There are no competing interests that could be perceived to bias this work.

reduced along the products' life cycles, so as to inform targeted actions to reduce emissions within the laboratory and in procurement decisions.

## Introduction

The United Nations Framework Convention on Climate Change in 1992 marked the first formal international recognition of the need to stabilise atmospheric concentrations of greenhouse gas emissions (GHGs). Despite numerous subsequent conferences and agreements, emissions reductions have failed to materialise. Global surface temperature has climbed by an average of 1.1˚C to 1.3˚C since the pre-industrial period, with impacts evident in more extreme and frequent weather events, wildfires, rising sea levels, altered rain patterns, and ocean heating [1]. The International Panel on Climate Change (IPCC) has clearly established that human activities are the main drivers of heating and have called for drastic reductions in GHG emissions to avoid irreversible environmental damage and loss of life [1].

While scientific research facilities allow us to better understand the climate crisis, as well as many other aspects of the natural world, they are also resource and energy intensive, and thus contribute to the damage as well [2]. Energy-intensive buildings and equipment, reliance on plastic and chemical consumables (manufacture, transport and disposal), infrastructure maintenance, animal housing, business travel and commuting, and data storage all create a significant environmental footprint [3]. Growing recognition amongst scientists and laboratory researchers of the incompatibility of current laboratory practice with carbon reduction targets has driven efforts to minimise laboratory emissions [3]. Actions such as closing the sashes of fume hoods, turning off equipment not in use, and raising the temperature of ultra-low temperature freezers have been identified as initial methods to reduce laboratory energy consumption [3].

Much more is needed, however. Identifying solutions to the reliance on single-use consumables has proved more elusive. Plastics and other disposable consumables are integral to laboratory activities due to their low costs, light weight, time savings, and sterility [4]. However, the environmental impact of these consumables is substantial, with one estimate holding that 20,500 institutions worldwide involved in biological, medical or agricultural research produced 5.5 million tonnes of plastic waste in 2014 [4].

Establishing the GHG emissions of these products is a more complex task. Environmentally extended input-output analysis (EEIO) of inventory spend data is a commonly used approach to estimate the carbon footprint of consumables [5–7]. However, EEIO relies on broad economic categorisations and provides insufficient detail for shaping purchasing decisions [5].

An alternative approach, Life Cycle Assessment (LCA), captures the environmental impacts of products and systems through a systematic, highly detailed methodology, and can provide data on which stages of a product or system's life are most impactful. However, few LCAs have been conducted on the consumables typically found in the lab. Thus, the goals of the present study are several. First, to identify direct or proxy emission factors for a variety of lab consumables through a focused literature review of LCA studies. Second, to inform on the emission hotspots of these materials. Finally, to quantify and explore opportunities to reduce emissions from lab consumables.

The following sections will review previous assessments of laboratory consumables, describe the method of developing emissions factors, detail findings from the literature review of proxy consumable LCAs, and suggest potential interventions which could reduce the laboratory footprint associated with single-use consumables.

### Review of carbon footprinting of lab consumables

Laboratory consumables have occasionally been captured in carbon footprinting exercises for universities. These studies typically document emissions relating to university procurement purchases. They sometimes specify categories of consumables such as plastic or paper, but do not offer further detail [5–8]. Many of these studies use EEIO analysis, which assigns emission factors capturing the upstream or "embodied" emissions of a category of goods or services based on monetary and occasionally physical flows between economic sectors [9]. EEIO analysis benefits from simplicity, ease of comparability, and reliance on publicly available data [5]. However, the assumed homogeneity across each sector does not offer detail into how changes in material use, production, or disposal could affect the carbon footprint of products.

In contrast, LCA serves as a reputable, internationally standardised method of estimating the environmental impacts of a given product, service, or system over its lifetime [10]. "Cradle-to-grave" LCAs typically cover the entire product life cycle, including the stages of raw material extraction, manufacturing, transport, distribution, use, and disposal [7]. For each stage, the material inputs and outputs are recorded, and their respective environmental loads are calculated.

LCAs can identify the GHG emissions generated in each stage of a product's life cycle, making them very useful in determining emissions hotspots, comparing similar products, and evaluating potential substitutes. However, LCAs are highly time consuming and data-intensive, which has limited their use in footprinting exercises for universities and large organisations [7]. Additionally, the boundaries of LCAs vary, which can lead to different estimates of environmental impacts for the same products. The validity of LCA studies also relies on high quality data and often is context-specific, thus making documentation of data assumptions very important.

Within the scientific research sector, LCAs on laboratory materials are scant. One study has evaluated 'cradle-to-gate' (resource extraction to factory gate) GHG emission factors of chemical solvents, drawing on LCAs and LCA databases for estimates [11]. A recently published database aggregates LCAs relating to healthcare products and processes [12]. Some LCAs accessible in the database, such as those on disposable personal protective equipment (PPE) and nitrile gloves, are transferable to laboratories [13–16]. However, LCAs on common single-use lab plastics do not yet exist.

Assessments of alternatives to single-use consumables are also sparse. Alves et al. [17] considers methods to reduce single-use plastic waste within a microbiology lab but does not calculate the emissions savings associated with the alternative approaches. Farley and Nicolet [18] compare disposable lab consumables with reusable alternatives but limit their assessment to four products.

The present study covers a wider scope of consumables and seeks to provide insight not only in terms of the total footprint of consumables, but also on alternative approaches to reduce emissions in production and disposal.

## Methodology

### Literature review

Emission factors were developed for a selection of laboratory consumables through a focused literature review of products created with the same primary material. The investigated products were those frequently disposed of in the lab. The primary material composition was identified through prior LCA studies of the products and publicly available information from suppliers, if the material composition was not immediately known.

**Literature search.** The following search terms were investigated:

- *Products*

  – *Plastic consumables*: pipette tips, microtubes, microplates, cell culture dishes and PCR plates, stripettes, flasks, deep-well plates, PCR tubes, cuvettes, weighing boats, plastic syringes, bijous, and falcon tubes

  – *Protective wear*: nitrile gloves, laboratory aprons, face masks, protective eyewear, and respirators

  –*Solvents and chemicals*: ethanol, isopropanol, methanol, acetone, and toluene

- Materials

  – *Plastics*: polyethylene terephthalate (PET), high density polyethylene (HDPE), low density polyethylene (LDPE), polycarbonate (PC), polystyrene (PS), and polypropylene (PP).

Using these products and materials as keywords, forward and backward citation searching was conducted through Google Scholar and Scopus, as well as within relevant journals (*International Journal of Life Cycle Assessment, Journal of Cleaner Production, International Journal of Sustainability in Higher Education*) between October 2022 and February 2023. In the initial scan, 96 studies were identified as potentially relevant. Fig 1 illustrates the article review process, as well as subsequent stages of the research.

Studies that were not peer-reviewed and did not use LCA methodology were excluded, as were those that did not evaluate the global warming potential (GWP) of the assessed product. For plastics, studies that considered final products rather than solely polymer production were

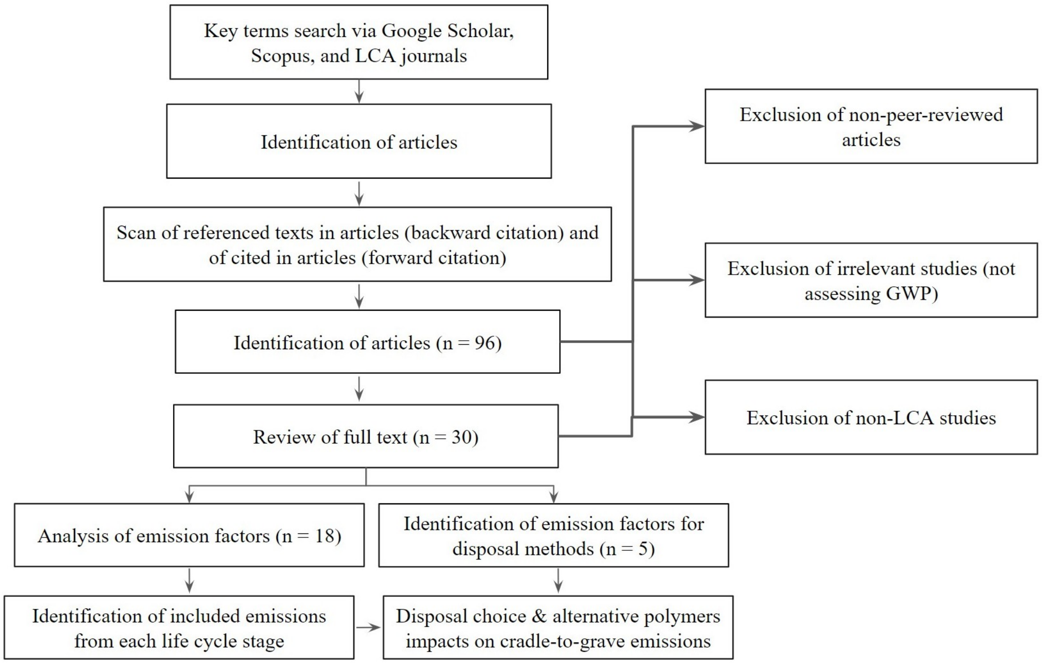

**Fig 1. Stages of the research process.** The figure documents the total number of studies initially recognised as viable (n = 96), the studies that were reviewed in full after excluding irrelevant studies (n = 30), and the studies that were ultimately used in developing cradle to gate emission factors (n = 18) and disposal methods (n = 5).

https://doi.org/10.1371/journal.pstr.0000080.g001

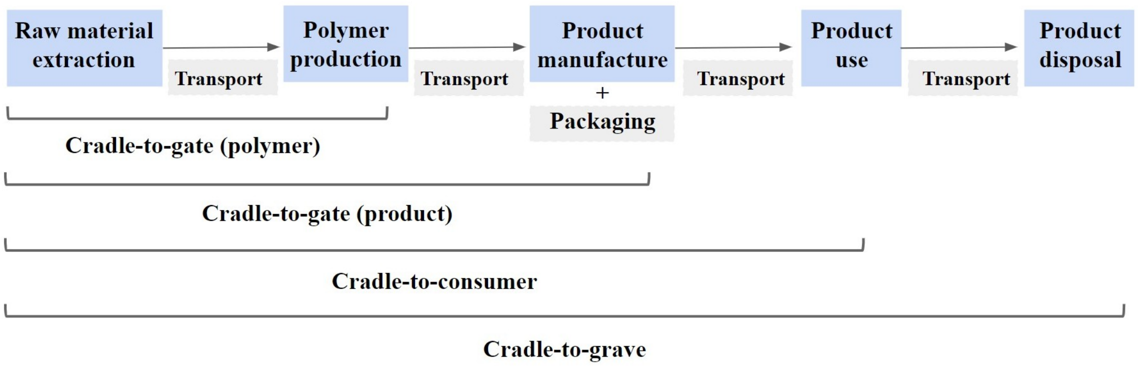

**Fig 2. Potential system boundaries of an LCA.** Stages that are typically included in LCAs are represented in blue. Transport and packaging are shown in grey, as they are only sometimes included in lab consumables LCAs.

https://doi.org/10.1371/journal.pstr.0000080.g002

prioritised. Additionally, only studies where it was possible to disaggregate end-of-life GWP and cradle-to-gate GWP were used.

Where LCAs specifically assessing lab consumables could not be identified, proxy cradle-to-gate emission factors were determined based on peer-reviewed LCAs of products made from the same materials.

### Harmonisation of LCA assumptions to enable comparability.

#### *Harmonisation of system boundaries*

LCAs vary in scope, meaning that they can include different stages of product life. Understanding the variations in system boundaries between LCAs is crucial when comparing different studies. Fig 2 shows a simplified representation of LCA boundaries for single use consumables. For instance, some LCAs only capture up to polymer production (cradle-to-gate (polymer)), while others may include product manufacture (cradle-to-gate (product)), and others still may include product use (cradle-to-consumer) and disposal (cradle-to-grave). Some LCAs include transport between the production stages and to consumers, while others exclude several or all transport stages. Given the discrepancies in boundaries, the published carbon footprints of lab consumables are not directly comparable. Hence a key contribution of this study is to determine the contribution of each stage to the full life cycle of lab products, which allows the identification of emission hotspots over the entire product life cycle.

#### *Comparisons of disposal and alternative raw materials*

A further literature review was conducted to include two topics of main interest to decision makers, namely 1.) disposal method and 2.) alternatives to virgin plastic polymers. To represent waste disposal, clinical waste incineration, pure plastic incineration, incineration with energy recovery, landfills, and closed-loop recycling were considered. End-of-life emission factors were taken from the literature if available, and the United Kingdom's Department for Environment, Food and Rural Affairs (DEFRA) if not. These new emission factors were considered to enable comparability between different consumables, given that each LCA took a different approach to product disposal. For instance, the LCAs on plastic materials often considered a mix of landfill, incineration, and recycling, or varying levels of incineration with energy recovery. Incineration with energy recovery was taken from DEFRA in order to harmonise assumptions on the level of recoverable energy.

As alternatives to virgin plastic materials several options were considered: a PP polymer derived from used cooking oil (UCO-PP), recycled PC, and recycled PET polymers.

# Results

## Overall findings

In total, 18 peer-reviewed studies were utilised to develop emission factors [11, 13–15, 19–32]. The studies were published from 2009 to 2023, with the greatest quantity published in 2021 (4 studies). Table 1 shows the number and scope of studies identified for each material.

LCAs specifically assessing lab consumables were only identified for nitrile gloves, protective wear, and solvents. For the remaining plastic consumables in Table 1, proxy cradle-to-gate emission factors were derived based on peer-reviewed LCAs of products made from the same polymers. An LCA assessing waste-based feedstock material for PP and several considering recycled PC and recycled PET polymers were also included.

The LCAs identified for PC, PE, and HDPE cover cradle-to-gate polymer emissions, inclusive of raw material extraction, transport to feedstock production, and feedstock production. For the other materials, LCA boundaries extend to capture transport to product manufacture and product manufacture. However, they vary in their inclusion of packaging emissions and transport to consumers.

Fig 3 shows each material's cradle-to-gate emission factor, as reported in the literature (kg polymer or kg product as identified in Table 1). In cases where multiple LCAs were identified

**Table 1. Literature sources identified for each lab material.** Emission factors were identified for polyethylene terephthalate (PET), high density polyethylene (HDPE), low density polyethylene (LDPE), polycarbonate (PC), polystyrene (PS), polypropylene (PP), recycled polycarbonate (rPC), recycled polyethylene terephthalate (rPET), used cooking oil-based polypropylene, various solvents, and protective wear.

| Material or product | Sources | Scope |
|---|---|---|
| HDPE | Harding et al., 2007 [24]; Hottle et al., 2017 [25] | Polymer only |
| LDPE | Rizan et al., 2021 [13] | Final product (apron) |
| PC | Kumar et al., 2021 [14]; Zhou et al., 2023 [31] | Polymer only |
| PP | Gao et al., 2022 [22]; Maga et al., 2019 [26]; Mannheim and Simenfalvi [27]; Moretti et al., 2021 [28] | Final product (single-use cups, straws, trays, generic PP product) |
| PET | Benavides et al., 2018 [20]; Dormer et al., 2013 [21]; Gironi, 2011 [23]; Moretti et al., 2021 [28] | Final product (bottles, trays, single-use cups, bottles) |
| PS | Razza et al., 2009 [30]; Zampori and Dotelli, 2014 [32] | Final product (cutlery, trays) |
| PE | Aryan and Samadder, 2019 [19] | Polymer only |
| rPC | Zhou et al., 2023 [31] | Polymer only |
| rPET | Dormer et al., 2013 [21] | Final product (trays) |
| UCO-PP | Moretti et al., 2020 [29] | Polymer only |
| Solvents | Khoo, 2018 [11] | Final product (solvents) |
| Nitrile gloves | Rizan et al., 2021 [13]; Jamal et al., 2021 [15] | Final product (nitrile gloves) |
| Face shield | Rizan et al., 2021 [13] | Final product (face shield) |
| Disposable gown | Rizan et al., 2021 [13] | Final product (disposable gown) |
| Mask | Rizan et al., 2021 [13] | Final product (type II mask, type II-R mask) |
| Respirator | Rizan et al., 2021 [13] | Final product (duckbill respirator, cup respirator) |

https://doi.org/10.1371/journal.pstr.0000080.t001

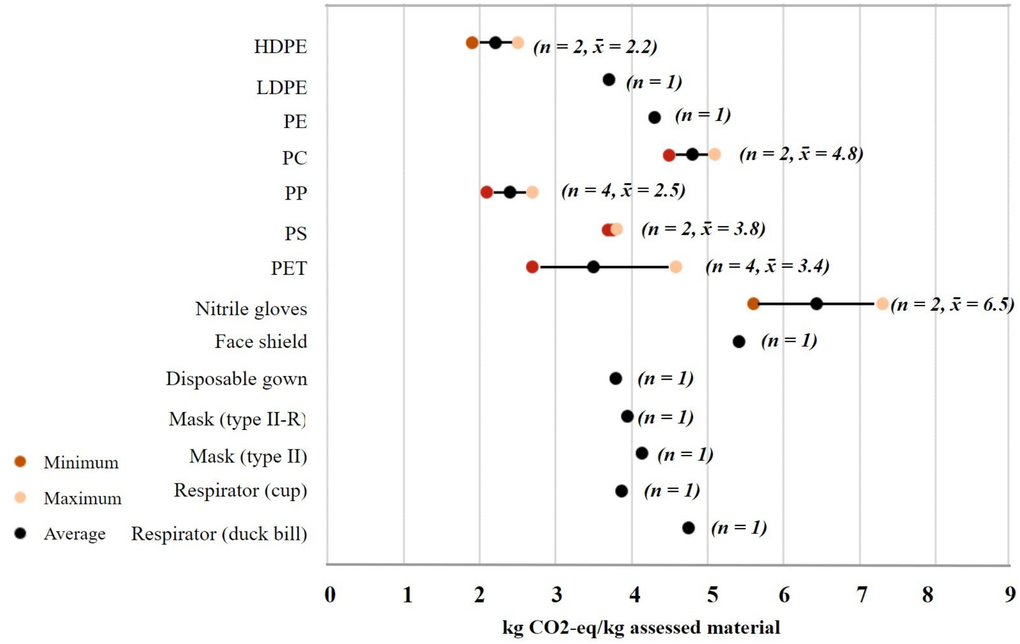

**Fig 3. Potential system boundaries of an LCA.** Stages that are typically included in LCAs are represented in blue. Transport and packaging are shown in grey, as they are only sometimes included in lab consumables LCAs.

https://doi.org/10.1371/journal.pstr.0000080.g003

for a single material, the minimum and maximum reported emission factors are shown, as well as an averaged value across all studies. Emission factors reported by each study can be found in the S1 Appendix.

Nitrile gloves were found to have the highest carbon impact of all assessed materials per kilogramme of material. PC was found to generate the most emissions of all plastics, even though the identified LCAs only capture PC polymer production. PP, one of the most common plastic materials in the lab, and HDPE had the lowest emission factors.

Differences between LCAs of the same material demonstrate the impact of the energy mix used in production. The most significant driver of the range in emission factors for all materials is the power generation mix of their respective production and manufacturing facilities. For instance, PET's wider range is largely driven by differences between United States and European production data. Similarly, Jamal et al [15] assume higher emissions from raw material and manufacturing of nitrile gloves than Rizan et al. [13], which could be driven by production in China as compared to Malaysia. Countries of production for each study are documented in the S1 Appendix.

The variation in the emission factor of different types of plastic suggests opportunities to reduce emissions through substitution of materials. For instance, as shown in Fig 4, deep-well plates can be made of PS or PP. However, the results indicate that PP products generate 33% less emissions than PS products.

Comparisons of products serving the same function must consider mass as well as material type. For instance, cup respirators were found to have lower emissions per kilogramme than duck-bill respirators. However, as illustrated in Fig 4, Rizan et al. [13] found cup respirators generate higher emissions than duck-bill due to their higher mass, which is almost double that of duck-bill respirators.

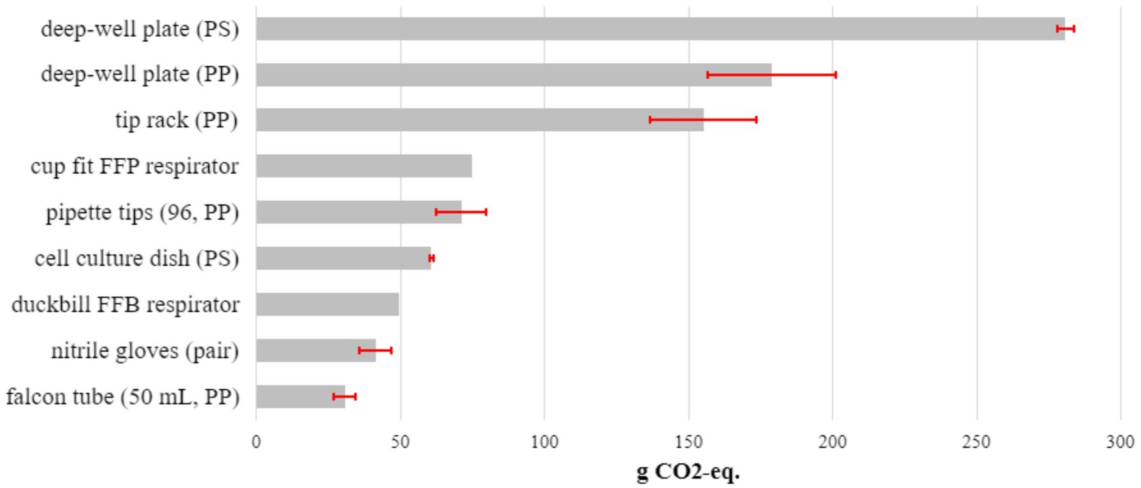

**Fig 4. Cradle-to-gate emissions from lab consumables ($gCO_2eq$).** The selection of consumables shown consist of polystyrene (PS) and polypropylene (PP) products, as well as personal protective equipment (PPE). Where multiple LCAs for a given material were found, the emissions estimates are based on the averaged emission factor. The red error bars reflect product emissions as calculated with the minimum and maximum emission factors found in the literature.

https://doi.org/10.1371/journal.pstr.0000080.g004

## Emission hotspots

Emission hotspots give insight into where the greatest emissions reductions can be achieved. Although the LCAs considered different materials and products, there were similarities in terms of the life cycle stages in which emissions were concentrated. In general, as summarised in Fig 5, the polymer and raw material production stage contributed the most to overall emissions, followed by end-of-life when incineration was the disposal choice. Product manufacture played an important role for nitrile gloves and some plastics, while transport and packaging were found to be less significant. A table documenting the emissions associated with each phase of product life, as reported in the assessed LCAs, is available in the S1 Appendix.

In the following the GHG hotspots are further discussed per type of laboratory consumable. The influence of disposal choice and alternative feedstocks are discussed separately for plastic consumables, given their prominent contribution to the overall cradle-to-grave GHG emissions of consumables.

**Gloves.** Two studies assessing nitrile gloves from cradle-to-grave were identified in the initial literature review, with emission factors detailed in the S1 Appendix. Both studies ranked raw material production as the most emissions-intensive life cycle stage for nitrile gloves, responsible for 35 and 38% of cradle-to-grave emissions, followed by glove manufacture (approximately 30%), and incineration (the rest to 100%). Rizan et al. [13] and Jamal et al. [15] assume the same emissions from disposal, so variations in their cradle-to-grave GWP estimates reflect differences in supplier-specific energy sources. Jamal et al.'s [15] carbon impact estimate is 20% higher than Rizan et al. [13] due to greater emissions in nitrile production and glove manufacture. Both studies utilised Ecoinvent data to quantify the emissions related to electricity use, but Jamal et al. [15] captures glove production in Malaysia, while Rizan et al. [13] captures glove production in China.

Transport and packaging, captured by both studies, were found to have negligible impacts on overall emissions, even when gloves produced in Asia were used in Europe. Transport contributed 3 to 3.3% and packaging contributed less than 2% to the overall emissions factor over the gloves' life cycle.

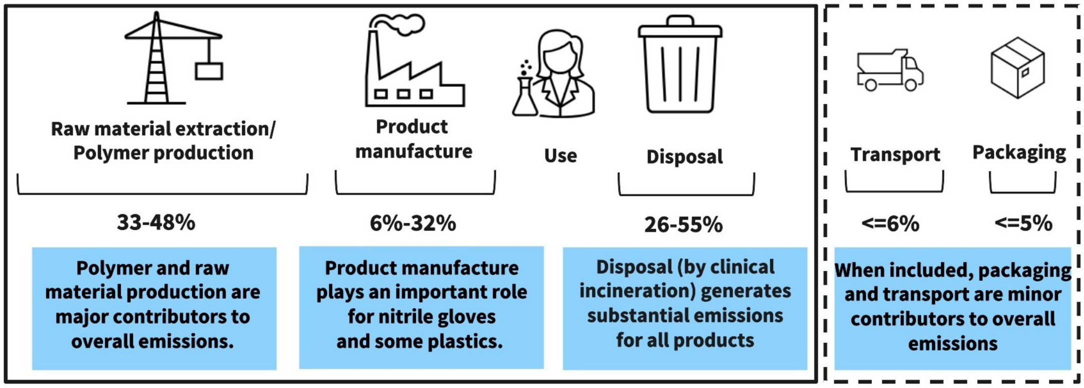

**Fig 5. Emission hotspots in the life cycle of consumables.** This figure illustrates the emission hotspots identified in studies assessing the supply chains of lab consumables or proxy single-use products. The figure does not include results for surgical masks and respirators as they generate more substantial packaging (up to 11% of total) and transport (up to 10% of total) emissions.

https://doi.org/10.1371/journal.pstr.0000080.g005

**Other protective wear.** Emission factors for other protective wear–laboratory aprons, face masks, face shields, disposable gowns, and respirators–were taken from Rizan et al. [13] (detailed in the S1 Appendix). All PPE was assumed to be clinically incinerated. For these products, production of materials (raw material extraction, production, and transport to PPE manufacturer) and disposal contributed most to overall emissions. Manufacturing, transport, and packaging contributed minimally in comparison. Electricity consumption in manufacturing was responsible for 2 to 8% of overall emissions. Packaging contributed between 0.5% and 4% of overall emissions for aprons, face shields, and disposable gowns, while transport contributed between 4% and 5%.

For duck-bill respirators and surgical masks, packaging had a higher emissions contribution, responsible for between 8 and 11% of overall emissions. This was due in large part to the individual wrapping of duck-bill respirators in LDPE film and the packaging weigh in relation to surgical mask weight. Transport-related emissions were higher than other protective wear but did not exceeded 10% of overall emissions.

**Plastics.** No LCAs for plastic lab consumables were identified, so LCAs of other plastic products were reviewed to develop proxy emission factors for products made from seven commonly used plastic materials: PP, PC, PS, PET, PE, HDPE, and LDPE. Emission factors for plastics are detailed in Tables 2 and 3.

Due to the wide variation in disposal methods for plastics, this section will report only on the cradle-to-gate emissions covering the stages of raw material and feedstock production, product manufacture, and transport and packaging as available. End-of-life emissions are

**Table 2. Cradle-to-gate LCAs of plastic polymers.**

| Source | Material | Production location | Stages (included) | Cradle-to-gate kg$CO_2$ eq / kg polymer |
|---|---|---|---|---|
| Harding et al. (2007) [24] | HDPE | Europe | Raw material and polymer production | 2.5 |
| Hottle et al. (2017) [25] | HDPE | United States | Raw material and resin production | 1.9 |
| Zhou et al. (2023) [31] | PC | China | Raw material prod., transport to PC production site, PC prod. | 5.1 |
| Kumar et al. (2021) [14] | PC | India | Raw material and polymer production | 4.5 |
| Aryan et al. (2019) [19] | PE | India | Production of virgin polymer | 4.3 |

https://doi.org/10.1371/journal.pstr.0000080.t002

**Table 3. Identified LCAs of plastic products.**

| Material | Source | Production location | Product | Stages (included in the emission factors) $kgCO_2$ eq / kg final product* |
|---|---|---|---|---|
| PS | Razza et al. (2009) [30] | Italy/Switzerland | Cutlery | Raw material production, polymer production, product manufacture, disposal (incineration with recovery) 3.8; n/a |
| PS | Zampori and Dotelli (2014) [32] | Italy | Tray | Raw material production, tray production, transport to user 3.7; 3.9 |
| PP | Moretti et al. (2021) [28] | Europe | Single-use cups | Raw material production, polymer transport, thermoforming, cups distribution, disposal (mixed) 2.6; 2.8 |
| PP | Mannheim and Simenfalvi (2020) [27] | Europe/Hungary | Generic PP product | Raw material production and transport, injection moulding, transport to users, disposal (incineration with recovery) n/a; 2.2 |
| PP | Maga et al. (2019) [26] | Italy/Germany | Meat tray | Raw material production and transport, extrusion, thermoforming, disposal (mixed) 2.7; n/a |
| PP | Gao and Wan (2022) [22] | United States | Single-use straws | Raw material production, transport, straw extrusion, and distribution, disposal (incineration) 2.1; 2.1 |
| PET | Benavides et al. (2018) [20] | United States | Bottles | Feedstock, conversion, transport to manufacture, bottle manufacture. disposal (landfill) 4.6; n/a |
| PET | Dormer et al. (2013) [21] | Europe/UK | Trays | Raw materials, plastic forming, transport, disposal (mixed) 3.4; 3.4 |
| PET | Moretti et al. (2021) [28] | Europe | Single-use cups | Polymer, polymer transport, thermoforming, distribution, disposal (mixed) 2.7; 2.9 |
| PET | Gironi and Piemonte (2011) [23] | Europe | Bottles | Polymer production, polymer transport, bottle manufacture, disposal (incineration with energy recovery) 3.1; n/a |
| LDPE | Rizan et al. (2020) [13] | China/Thailand | Apron | LDPE film production, electricity, transport to users (in UK), disposal (clinical waste incineration) 3.7; 4.0 |

*Cradle-to-gate without transport to the consumer; cradle-to-gate with transport to the consumer.

https://doi.org/10.1371/journal.pstr.0000080.t003

discussed separately in the sensitivity analysis, which considers how overall emissions change depending on the disposal choice.

LCA studies that include product manufacture as well as polymer production were found for PP, PS, PET, and LDPE. These studies assessed different products, documented in Table 3.

Polymer production was found to be the main contributor to cradle-to-gate emissions for PS, PET, and PP. Razza et al. [30] found that 61% of cradle-to-gate emissions for PS cutlery derived from polymer production. Polymer production made up 61 to 88% of PET products' cradle-to-gate emissions and 62 to 82% of PP products' cradle-to-gate emissions [20–23, 26, 28].

Product manufacture was always the second highest contributor to cradle-to-gate emissions. Manufacture was found to contribute approximately 30% of cradle-to-gate emissions in an LCA of PS cutlery [30], in assessments of PP trays and cups [26, 28], and in one LCA of PET bottles [20]. For the remaining studies, manufacture contributed between 12 and 17% of cradle-to-consumer emissions.

Although each study varied in its inclusion of transport and packaging emissions, any time these were included, their contribution was always small, ranging from 1% to 8% of emissions (packaging) and less than or equal to 5% (transport to user) of cradle-to-consumer emissions.

**Solvents.** Only one paper was identified that assessed emission factors for solvents. Khoo et al. [11] conducted a literature review of cradle-to-gate emission factors for different solvents, assigning each factor a reliability ranking. In the present assessment, the emission factors assigned the highest reliability ranking were used (Table 4). For all solvents except methanol and cyclopentyl methyl ether (CPME), the highest reliability ranking was given to the value provided by Ecoinvent. For methanol, the value was taken from the NREL database and for CPME it is from Kin et al. [33]. The emission factor for CPME is the only value that falls below

**Table 4. Solvent emission factors.**

| Solvent | Scope | $kgCO_2eq$ / kg solvent |
|---|---|---|
| acetone | cradle-to-gate | 2.3 |
| ethanol | cradle-to-gate | 2.1 |
| ethyl acetate | cradle-to-gate | 1.5 |
| formic acid | cradle-to-gate | 2.5 |
| isopropanol | cradle-to-gate | 1.7 |
| methanol | cradle-to-gate | 1.1 |
| toluene | cradle-to-gate | 1.6 |
| dichloromethane (DME) | cradle-to-gate | 5.0 |
| cyclopentyl methyl ether (CPME) | cradle-to-gate | 1.8 |
| N-Methyl-2-pyrrolidone (NMP) | cradle-to-gate | 4.8 |
| heptane | cradle-to-gate | 1.1 |

https://doi.org/10.1371/journal.pstr.0000080.t004

Khoo et al.'s [11] recommended reliability rank and should be approached with caution. Ecoinvent and NREL are highly trusted databases and generally representative of manufacturing of these chemicals from a global perspective. End-of-life treatments of solvents were not assessed, though they may have significant additional impacts if evaporated into the atmosphere and should be considered in future studies [36].

## End of life

Identified LCAs of plastic products that covered cradle-to-grave emissions found disposal to contribute between 3.8% (landfilled PET trays as reported by Dormer et al. [21]) and 60% (incinerated PP straws as documented by Gao and Wan [22]) of overall emissions. The original cradle-to-grave emission factors are available in the S1 Appendix. However, since most LCAs either did not capture disposal in the system boundary or assumed varying percentages of waste going to different waste treatments, it is not possible to directly compare emissions from disposal across all LCAs. Instead, this study utilised emissions factors primarily from DEFRA [34], as well as from several LCAs that considered only one type of end-of-life [19, 25, 35]. Table 5 shows the emission factors used for each disposal method.

To determine total cradle-to-grave emissions, the cradle-to-gate and disposal emission factors for each material were summed. Where a material had multiple LCA emissions estimates, the averaged cradle-to-gate emission factor was used.

Clinical waste incineration, a common disposal method for laboratory consumables, has a substantial carbon impact. As shown in Fig 6, clinical waste incineration can contribute between 28 and 54% of total product emissions. Clinical waste incineration assumes a mixture of materials in disposal. The emissions from incineration of pure plastic material (identified for PET, PE, and PP) generates even higher emissions, raising total emissions by an additional 5 to 13%, depending on the material. Landfilled plastics have near zero emissions (only from transport and handling), as landfilled petrochemical plastics and PPE materials are chemically inert. However, landfilling is typically avoided in Europe and the UK due to risks of leakages to the environment and zero waste to landfill initiatives [37, 38].

Figs 7, 8, and 9 compare the total emissions of different disposal methods for the most common plastic materials in the lab–PP, PC, and PS–and demonstrate the impact of disposal methods on total footprint. Incineration with energy recovery can reduce the emissions related to disposal by between 12 and 32% compared to clinical incineration. However, closed-loop recycling is the only disposal method that reduces lifetime emissions. This holds true even

**Table 5. (GHG emissions by end-of-life option (kgCO$_2$eq / kg waste.)).**

| Material | Landfill[a] | Clinical waste incineration[b] | Autoclave decontamination | Incineration (100% plastic)[c] | Incineration (energy recovery)[d] | Recycling (closed-loop)[e] |
|---|---|---|---|---|---|---|
| HDPE | 0.14 | 2.57 | 0.34 | - | 1.06 | -1.07 |
| LDPE | 0.15 | 2.57 | 0.34 | - | 1.06 | -1.09 |
| Nitrile | 0.03 | 2.57 | 0.34 | - | 1.20 | - |
| PET | 0.12 | 2.57 | 0.34 | 3.34 | 1.83 | -1.92 |
| PC | 0.03 | 2.57 | 0.34 | - | 1.20 | -1.17 |
| PE | 0.03 | 2.57 | 0.34 | 2.94 | 1.06 | -1.17 |
| PP | 0.07 | 2.57 | 0.34 | 3.21 | 1.36 | -0.91 |
| PS | 0.03 | 2.57 | - | - | 1.07 | - |
| UCO-PP | 0.07 | 2.57 | 0.34 | 3.21 | 1.27 | 0 |
| rPC | 0.03 | 2.57 | 0.34 | - | 1.20 | 0 |
| rPET (100%) | 0.12 | 2.57 | 0.34 | 3.34 | 1.83 | 0 |

[a)] Hottle et al. (2017) [25] for HDPE, LDPE, and PET. The same emissions assumptions as PET are made for rPET (100%), Gao and Wan (2022) [22] for PP. The same emissions assumptions are made for UCO-PP. The DEFRA (2011) [34] value for average plastics was utilised for nitrile, PC, PS, and PE. Values for PC and PET were used for rPC and rPET.

[b.)] Rizan et al. (2021) [13] for clinical waste incineration (via Ecoinvent's estimate for hazardous waste incineration) and autoclave steam decontamination. PS cannot be autoclaved as the temperature exceeds its melting point.

[c.)] Aryan et al. (2019)n [19] for PET and PE. The same factor is applied to rPET. Gao and Wan (2022) [22] for PP. The same factor is applied to UCO-PP.

[d.)] All values for incineration with energy recovery are taken from DEFRA (2011)n [34]. The rPC and rPET factors are assumed to be the same as their virgin polymer counterparts. The value for average plastics was assigned to nitrile.

[e.)] Emission factors for HDPE, LDPE, and PET are taken from Hottle et al. (2017) [25]. All other values are taken from DEFRA. Circularity is assumed for rPET, rPC, and UCO-PP so no recycling credits are assumed. PS is assumed to not be recyclable.

https://doi.org/10.1371/journal.pstr.0000080.t005

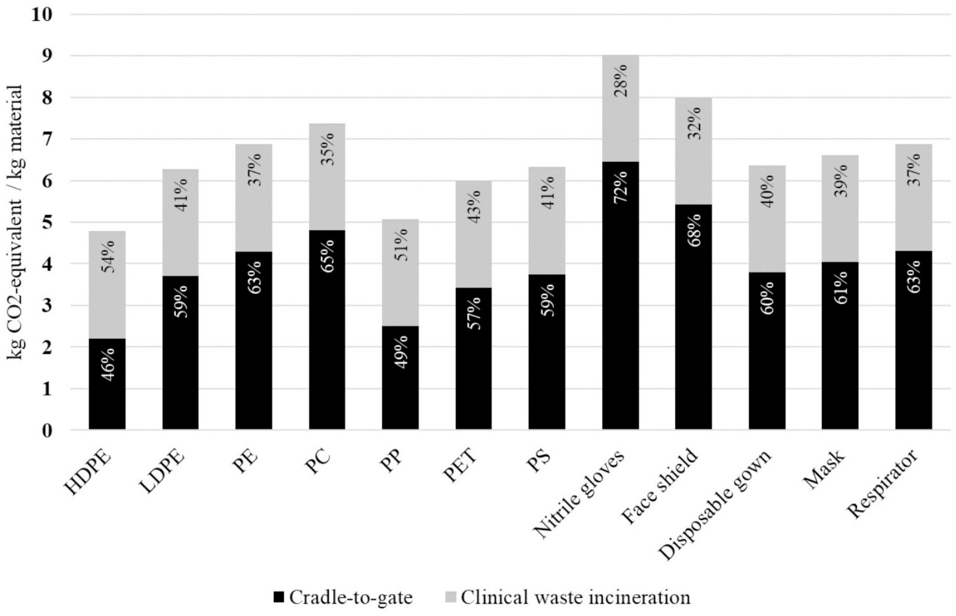

**Fig 6. Total life cycle emissions for each material, assuming disposal by clinical waste incineration.**

https://doi.org/10.1371/journal.pstr.0000080.g006

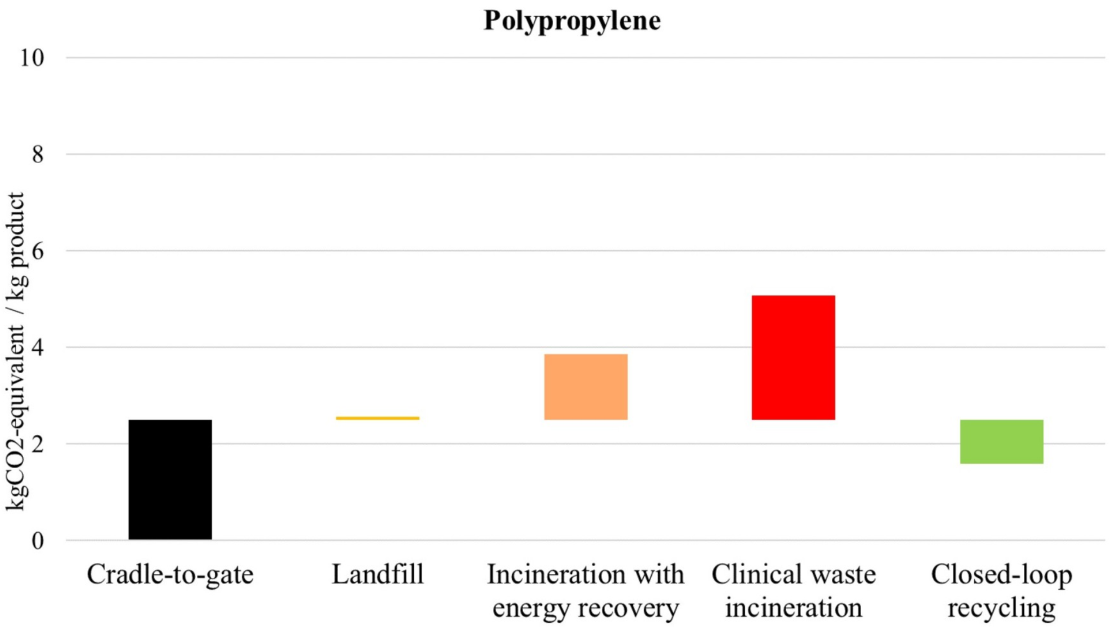

**Fig 7. Cradle-to-grave life cycle emissions based on end-of-life treatment for polypropylene products.**

https://doi.org/10.1371/journal.pstr.0000080.g007

when products are autoclaved prior to recycling. Closed-loop recycling credits the recycling institution with the reduction in emissions of manufacture of the new product, thus attributing "negative emissions" to the recycling institution.

By recycling instead of incinerating plastic products (PP, PC, and PET), lifetime emissions can be reduced by 51% (PC) to over 70% (PP and PET). However, closed-loop recycling is not currently possible for nitrile gloves, PS products, or protective wear.

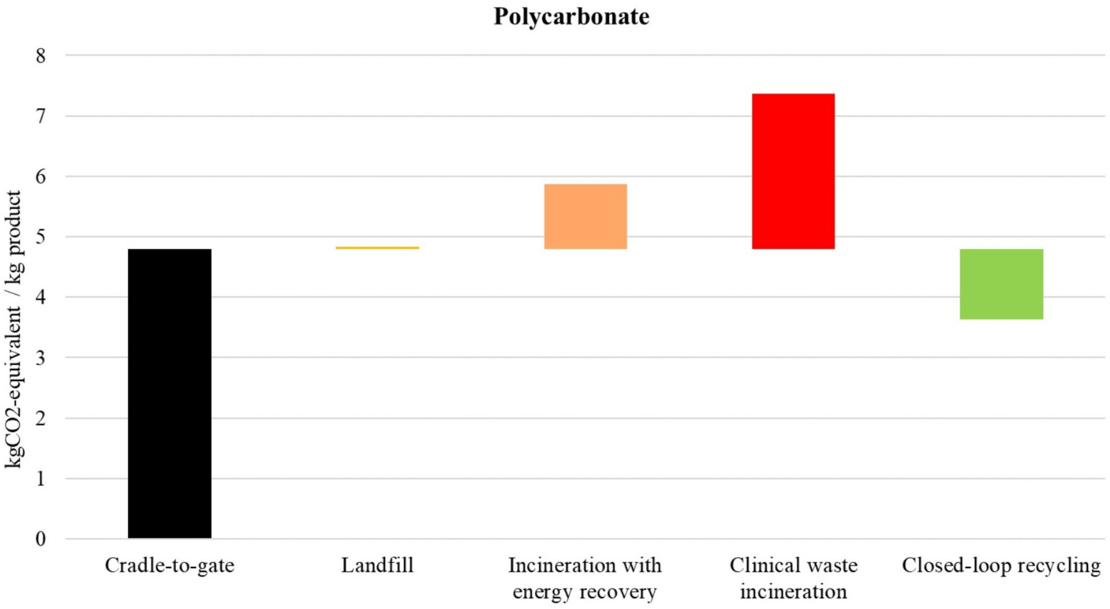

**Fig 8. Cradle-to-grave life cycle emissions based on end-of-life treatment for polycarbonate products.**

https://doi.org/10.1371/journal.pstr.0000080.g008

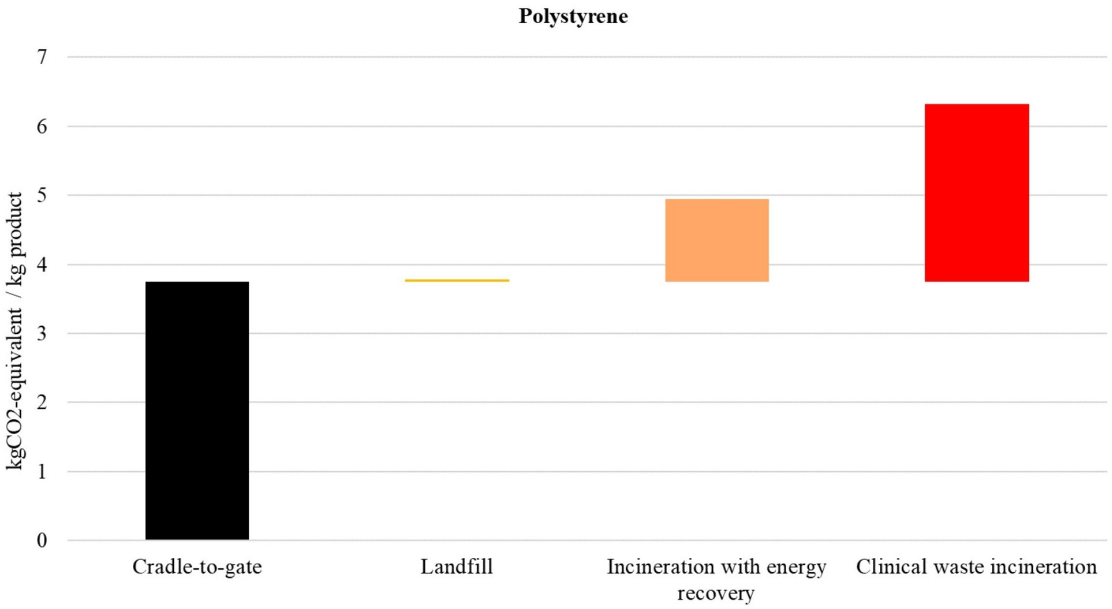

**Fig 9. Cradle-to-grave life cycle emissions based on end-of-life treatment for polystyrene products.**

https://doi.org/10.1371/journal.pstr.0000080.g009

## Alternative feedstocks

Using alternatives to virgin polymers in production can dramatically reduce emissions due to the high GWP of polymer production. Alternatives were found for PP, PC, and PET. UCO-PP is a novel source of PP. UCO-PP was found to generate 0.6 $kgCO_2$ eq/kg of polymer, decreasing emissions by 64% as compared to virgin PP polymer production. Use of UCO-PP could reduce cradle-to-consumer emissions by 48% and cradle-to-grave emissions by 24% as compared to virgin PP (assuming clinical waste incineration of both materials). PET and PC products made from recycled polymers have even more substantial reductions of 71 and 88% (cradle-to-gate) and 41 to 57% (cradle-to-grave), respectively. Recycled PS, PE, HDPE, and LDPE polymers were not identified.

Developing a fully circular system coupling recycled polymers in production with recycling at end of life may be able to get emissions from plastics closer to zero. Fig 10 shows the potential emission savings when both changes are made. Recycling virgin plastic products at end of life could reduce emissions by between 51 and 75% for the three assessed plastics (compared to clinical waste incineration). However, a circular approach of using recycled plastics in manufacture and recycling at end-of-life brings lifetime emissions down by 74 to 92%. To note, PC shows the greatest savings as compared to clinical incineration but likely overestimates emissions reduction for PC consumables because its emission factor only accounts for polymer production and excludes product manufacture.

## Limitations

Since this assessment relies heavily on proxy emission factors from LCAs done on other products, the emission factors serve as representative values only. This study attempted to avoid LCAs that excluded product manufacture, as these underestimate the total emissions of final products. In the cases where only polymer emissions were identified (HDPE, PE, and PC), emission factors should be interpreted with caution.

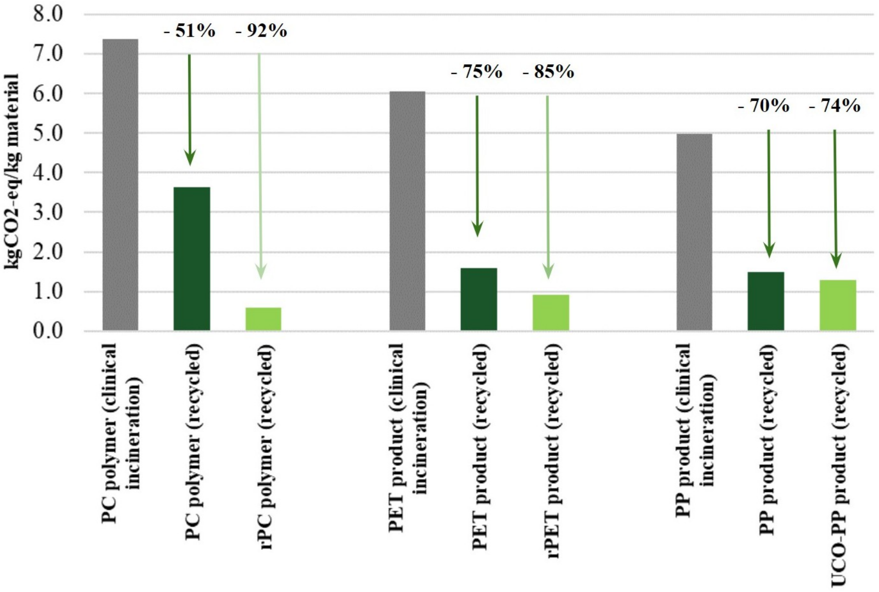

**Fig 10. Cradle-to-grave emissions from virgin polymers and alternatives.** Assessed materials include polyethylene terephthalate (PET), 100% recycled polyethylene terephthalate (rPC), polycarbonate (PC), recycled polycarbonate (rPC), polypropylene (PP), and used cooking oil-derived polypropylene (UCO-PP).

https://doi.org/10.1371/journal.pstr.0000080.g010

The LCAs identified also cover a range of countries with different energy mixes and do not always specify the assumptions made regarding fuel use. While most LCAs included the same stages (i.e. raw material production, manufacture, disposal), the boundaries of each stage and how they are streamlined varied (i.e. transport to production can be modelled separately or within polymer production; transport to consumers not always captured), which leads to issues in comparability. However, with proper documentation these issues can be moderated, and insights can still be developed.

## Discussion

The high reliance of science research on single-use materials has been noted as an area of concern when it comes to carbon emissions, but very little research has estimated the emissions associated with lab products [3]. This review assesses the available evidence to develop GHG emission factors that can be applied to lab consumables, based on LCA studies which provide substantial benefits in terms of transparency and detail.

LCAs of laboratory products provide users with the ability to make informed purchasing decisions regarding environmental impact, something that is currently lacking. The emission factors identified in this review illustrate that production of raw materials and incineration at end-of-life contribute most significantly to overall emissions of consumables. Additionally, product manufacture was found to play an important role for nitrile gloves and some plastics, while transport and packaging were found to be less significant. Transport, which is often emphasised in decarbonisation efforts, was found to contribute less than 6% of overall emissions for consumables.

In comparison to LCA, the more commonly used EEIO approach for assessing laboratory consumables, reliant on economic emission factors, does not provide the same level of detail. Spend data obscures where carbon emissions are originating from and so cannot provide insight into where changes can be most impactful. With EEIO analysis, the only way to see emissions reductions is through reduced expenditure.

Largely in response to the COVID-19 pandemic, several studies directly assessing protective wear were found, but a lack of LCAs directly assessing lab consumables, chemicals, and solvents is evident. Therefore, this study presents an initial estimate of GHG emission factors for common lab materials based largely on proxy values derived from LCAs of other plastic products. The LCA-based emission factors derived in this work can be used to estimate the carbon footprint of lab plastics, nitrile gloves, solvents, and protective wear.

## Lessons learned

Several lessons from the present review can inform efforts to reduce emissions within the laboratory setting:

1. **Within the lab, disposal choice can reduce carbon impacts.** Clinical incineration was found to generate over half of the emissions of protective wear and plastics. By recycling instead of incinerating recyclable plastic products (PP, PC, and PET), lifetime emissions can be reduced by 50 to 74%. Recycling benefits hold even when considering the need to autoclave materials prior to recycling. However, closed-loop recycling is not possible for common consumables such as nitrile gloves, PS products, or personal protective wear. In these cases, reductions in use and substitution with alternative materials is needed.
2. **Targeting the energy sources for polymer production and product manufacture can significantly reduce the carbon footprint of materials.** Even when consumables cannot be recycled, emissions can be reduced by producing materials with a cleaner electricity mix. The manufacture of nitrile gloves was shown to contribute a third of total emissions, and to be substantially affected by the electricity generation mix. Jamal et al. [15] and Rizan et al. [13] varied in their emission factor estimates due to assumptions on the sources of electricity generation. Results from Rizan et al. [13] support the importance of the energy mix used in production, as they found that PPE production in the UK rather than from external sources could reduce emissions by 12%, with three quarters of this reduction attributed to the cleaner electricity mix in the UK.
3. **The greatest emissions reductions can be achieved through circular supply chains.** Combined, raw material production and clinical incineration were found to contribute 65% of the emissions associated with nitrile glove use and over 75% of the total emissions associated with plastic products. Eliminating incineration and opting for recycling where possible, using recycled and waste-based polymers, and reusing materials and products could thus have substantial impacts on the footprint of science research. Based on findings from rPET and rPC, when supply chains become fully circular, PET and PC emissions can be reduced by over 80%. Similarly, using waste-based PP and recycling at end-of-life could reduce PP emissions by 74%. While substantial reductions are possible, collaboration with suppliers is needed to reduce upstream emissions to reach net-zero emissions.
4. **Bio-based polymers may generate carbon savings but need to be considered with caution.** Used cooking oil was identified as an effective source of polypropylene, reducing the PP polymer emissions by over 60% as compared to virgin polymer production, and reducing overall product emissions by 24% (assuming clinical incineration). However, the conditions in which biopolymers are produced and disposed of are critical prior to assuming a reduction in emissions. Moretti et al. [29] evaluated a specific UCO-PP production plant in

the Netherlands reliant on locally sourced cooking oil. Currently, the UCO-PP from this site is only used for four microtube products, and the transferability of these results in other settings is unclear [39]. Moretti et al. [29] also note that UCO is a limited feedstock and the benefits from producing PP could be lower than utilising the feedstock for renewable diesel. Other biopolymers used for plastic products have been found to emit more GHGs even when excluding the land-use implications on scale-up [40]. Additionally, some have much higher emissions than their alternatives depending on the end-of-life treatment pursued [23]. Thus, biopolymer alternatives must be selected carefully, considering the context surrounding each option.

## Applications of derived emission factors

The emission factors identified in the present literature review can be utilised to produce carbon emission assessments that better capture Scope 3 emissions. Applying these factors in full LCAs of laboratories aids in identifying how changes in procurement and disposal could result in carbon emission reductions. This is something that is difficult if not impossible to achieve with spend-based emission factors [5] and available laboratory related LCAs, which lack consistent boundary definition [12]. For instance, applying the emissions factors derived in this study to assess the consumables' footprint of a theoretical researcher, we could identify practical ways to reduce emissions. The full details of our calculations are available in the S1 Appendix. We found that recycling and autoclaving all recyclable materials could reduce the overall footprint by 11%, and using waste-based PP rather than virgin PP would reduce emissions by a further 5%. Substituting recyclable options for nonrecyclable PS where possible can allow for further emissions savings. For instance, choosing to utilise PP deep-well plates instead of PS plates and recycling the consumables upon disposal could reduce the carbon footprint contribution of deep-well plates by over 70%. We found that approximately 60% of emissions were generated along product supply chains, requiring collaboration with suppliers on decreasing the emissions intensity of energy sources. Treatment of waste at the end of life of the consumables also makes a significant difference to the overall emissions. However, these actions would entail changes in procurement decisions and change in researcher behaviour, which could be informed by the type of data and analysis presented here.

## Further work

Fig 11 provides guidance for lab purchasers and researchers in the lab, based on the findings from the present literature review. Priorities for lab researchers should lie in reducing the volume of consumption, limiting the use of clinical incineration for disposal of lab materials, avoiding non-recyclable plastics, and opting for reuse where possible. Recycling offers emission reductions, with the present estimate for autoclaving suggesting this process will be less carbon-intensive than incineration.

The range in emission factors for the same materials demonstrates the importance of suppliers' electricity sources. For procurement, engaging in dialogue with suppliers on the emissions of their operations, particularly their electricity use and direct emissions, will be useful in understanding whether reductions in emissions are possible along the supply chain.

The literature review undertaken in this work identified substantial research gaps in terms of LCAs conducted on lab goods, but also demonstrates the importance of further research in areas including: investigating the recycling potential for PS, PP, HDPE, and LDPE, assessing the viability of repeated recycling of all plastic materials, determining the emissions associated

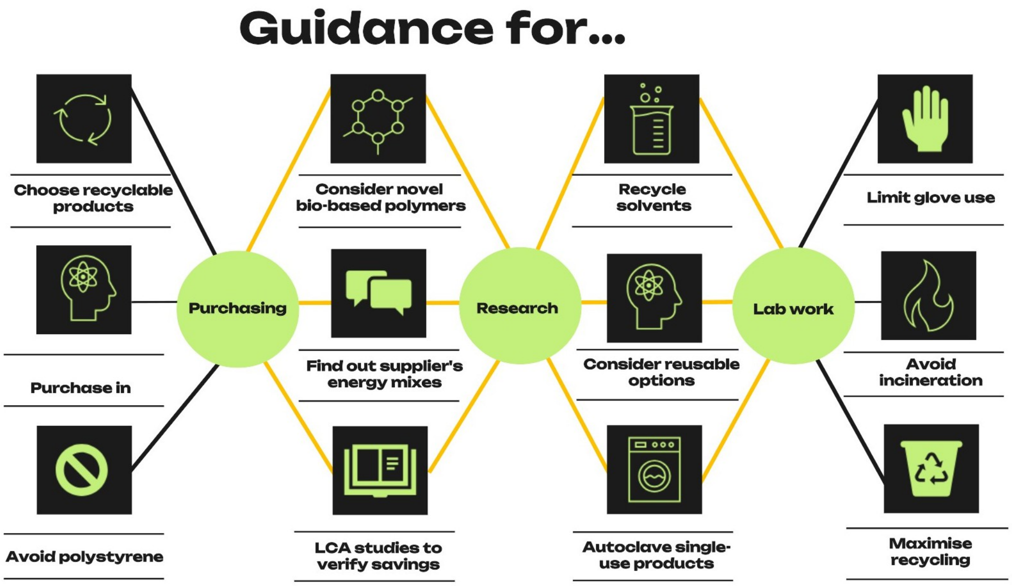

**Fig 11. Guidance for lab scientists.**

https://doi.org/10.1371/journal.pstr.0000080.g011

with autoclaving in a lab setting, and developing alternatives to use of PS. Opting for glass or durable plastic alternatives may be beneficial—While these products tend to have higher production emissions, if reused a sufficient number of times, they can generate considerable savings over the full life time of the reused consumable [41]. However, further work is needed to quantify the environmental and climate impacts of these options.

The present study illustrates the important role life cycle assessment can play in enabling emissions reductions in research. The emission factors developed in this analysis can be utilised by lab researchers to understand the carbon footprint of their laboratory consumables and identify immediate actions that can reduce emissions.

## Supporting information

**S1 Appendix. Supplemental Material.**
(DOCX)

## Acknowledgments

We would like to thank Richard Jackson, Director of Sustainability at University College London, for his support throughout the research process.

## Author Contributions

**Conceptualization:** Martin Farley, Kate Jeffery, Isabela Butnar.

**Data curation:** Isabella Ragazzi.

**Formal analysis:** Isabella Ragazzi.

**Funding acquisition:** Martin Farley, Kate Jeffery, Isabela Butnar.

**Investigation:** Isabella Ragazzi.

**Methodology:** Isabella Ragazzi, Isabela Butnar.

**Project administration:** Martin Farley.

**Supervision:** Martin Farley, Isabela Butnar.

**Visualization:** Isabella Ragazzi.

**Writing – original draft:** Isabella Ragazzi.

**Writing – review & editing:** Isabella Ragazzi, Martin Farley, Isabela Butnar.

## References

1. Intergovernmental Panel on Climate Change. Climate Change 2022: Impacts, Adaptation, and Vulnerability. Summary for Policymakers. IPCC.2022.
2. Greever C, Ramirez-Aguilar K, Connelly J. Connections between laboratory research and climate change: what scientists and policy makers can do to reduce environmental impacts. FEBS Lett. 2020; 594(19):3079–85. https://doi.org/10.1002/1873-3468.13932 PMID: 32964436
3. Rae CL, Farley M, Jeffery KJ, Urai AE. Climate crisis and ecological emergency: Why they concern (neuro) scientists, and what we can do. Brain Neurosci Adv. 2022; 6:23982128221075430. https://doi.org/10.1177/23982128221075430 PMID: 35252586
4. Urbina MA, Watts AJ, Reardon EE. Labs should cut plastic waste too. Nature. 2015; 528(7583):479. https://doi.org/10.1038/528479c PMID: 26701046
5. Townsend J, Barrett J. Exploring the applications of carbon footprinting towards sustainability at a UK university: reporting and decision making. J Clean Prod. 2015;6; 107:164–76. https://doi.org/10.1016/j.jclepro.2013.11.004
6. Larsen HN, Pettersen J, Solli C, Hertwich EG. Investigating the Carbon Footprint of a University-The case of NTNU. J Clean Prod. 2013;1; 48:39–47. https://doi.org/10.1016/j.jclepro.2011.10.007
7. Thurston M, Eckelman MJ. Assessing greenhouse gas emissions from university purchases. Int J Sustain High Educ. 2011;11; 12(3):225–35. https://doi.org/10.1108/14676371111148018
8. Ozawa-Meida L, Brockway P, Letten K, Davies J, Fleming P. Measuring carbon performance in a UK University through a consumption-based carbon footprint: De Montfort University case study. J Clean Prod. 2013;1; 56:185–98. https://doi.org/10.1016/j.jclepro.2011.09.028
9. Kitzes J. An introduction to environmentally-extended input-output analysis. Resources. 2013;30; 2 (4):489–503. https://doi.org/10.3390/resources2040489
10. Muralikrishna IV, Manickam V. Life cycle assessment. Environ Manag. 2017; 11(1):57–75. https://doi.org/10.1016/B978-0-12-811989-1.00005-1
11. Khoo HH, Isoni V, Sharratt PN. LCI data selection criteria for a multidisciplinary research team: LCA applied to solvents and chemicals. Sustain Prod Consum. 2018;1; 16:68–87. https://doi.org/10.1016/j.spc.2018.06.002
12. Drew J, Christie SD, Rainham D, Rizan C. HealthcareLCA: an open-access living database of healthcare environmental impact assessments. Lancet Planet Health. 2022;1; 6(12):1000–12. https://doi.org/10.1016/S2542-5196(22)00257-1 PMID: 36495883
13. Rizan C, Reed M, Bhutta MF. Environmental impact of personal protective equipment distributed for use by health and social care services in England in the first six months of the COVID-19 pandemic. J R Soc Med. 2021; 114(5):250–63. https://doi.org/10.1177/01410768211001583 PMID: 33726611
14. Kumar H, Azad A, Gupta A, Sharma J, Bherwani H, Labhsetwar NK, Kumar R. COVID-19 Creating another problem? Sustainable solution for PPE disposal through LCA approach. Environ, Dev and Sustain. 2021; 23:9418–32. https://doi.org/10.1007/s10668-020-01033-0 PMID: 33071605
15. Jamal H, Lyne A, Ashley P, Duane B. Non-sterile examination gloves and sterile surgical gloves: which are more sustainable?. J Hosp Infect. 2021;1; 118:87–95. https://doi.org/10.1016/j.jhin.2021.10.001 PMID: 34655693
16. Patrawoot S, Tran T, Arunchaiya M, Somsongkul V, Chisti Y, Hansupalak N. Environmental impacts of examination gloves made of natural rubber and nitrile rubber, identified by life-cycle assessment. SPE Polym. 2021; 2(3):179–90. https://doi.org/10.1002/pls2.10036
17. Alves J, Sargison FA, Stawarz H, Fox WB, Huete SG, Hassan A, McTeir B, Pickering AC. A case report: insights into reducing plastic waste in a microbiology laboratory. Access microbiol. 2021; 3(3). https://doi.org/10.1099/acmi.0.000173 PMID: 34151149

**18.** Farley M, Nicolet BP. Re-use of laboratory utensils reduces CO2 equivalent footprint and running costs. PloS One. 2023;12; 18(4):e0283697. https://doi.org/10.1371/journal.pone.0283697 PMID: 37043455

**19.** Aryan Y, Yadav P, Samadder SR. Life Cycle Assessment of the existing and proposed plastic waste management options in India: A case study. J Clean Prod. 2019;20; 211:1268–83. https://doi.org/10.1016/j.jclepro.2018.11.236

**20.** Benavides PT, Dunn JB, Han J, Biddy M, Markham J. Exploring comparative energy and environmental benefits of virgin, recycled, and bio-derived PET bottles. ACS Sustain Chem & Eng. 2018;28; 6 (8):9725–33. https://doi.org/10.1021/acssuschemeng.8b00750

**21.** Dormer A, Finn DP, Ward P, Cullen J. Carbon footprint analysis in plastics manufacturing. J Clean Prod. 2013;15; 51:133–41. https://doi.org/10.1016/j.jclepro.2013.01.014

**22.** Gao AL, Wan Y. Life cycle assessment of environmental impact of disposable drinking straws: A trade-off analysis with marine litter in the United States. Sci Tot Environ. 2022;15; 817:153016. https://doi.org/10.1016/j.scitotenv.2022.153016 PMID: 35026269

**23.** Gironi F, Piemonte V. Life cycle assessment of polylactic acid and polyethylene terephthalate bottles for drinking water. Environ Prog Sustain Energy. 2011; 30(3):459–68. https://doi.org/10.1002/ep.10490

**24.** Harding KG, Dennis JS, Von Blottnitz H, Harrison ST. Environmental analysis of plastic production processes: Comparing petroleum-based polypropylene and polyethylene with biologically-based poly- $\beta$-hydroxybutyric acid using life cycle analysis. J Biotechnol. 2007; 130(1):57–66. https://doi.org/10.1016/j.jbiotec.2007.02.012 PMID: 17400318

**25.** Hottle TA, Bilec MM, Landis AE. Biopolymer production and end of life comparisons using life cycle assessment. Resour Conserv Recycl. 2017;1; 122:295–306. https://doi.org/10.1016/j.resconrec.2017.03.002

**26.** Maga D, Hiebel M, Aryan V. A comparative life cycle assessment of meat trays made of various packaging materials. Sustainability. 2019 Sep 26; 11(19):5324. https://doi.org/10.3390/su11195324

**27.** Mannheim V, Simenfalvi Z. Total life cycle of polypropylene products: Reducing environmental impacts in the manufacturing phase. Polymers. 2020 Aug 24; 12(9):1901. https://doi.org/10.3390/polym12091901 PMID: 32846916

**28.** Moretti C, Hamelin L, Jakobsen LG, Junginger MH, Steingrimsdottir MM, Høibye L, Shen L. Cradle-to-grave life cycle assessment of single-use cups made from PLA, PP and PET. Resour Conserv Recycl. 2021 Jun 1; 169:105508. https://doi.org/10.1016/j.resconrec.2021.105508

**29.** Moretti C, Junginger M, Shen L. Environmental life cycle assessment of polypropylene made from used cooking oil. Resour Conserv Recycl. 2020 Jun 1; 157:104750. https://doi.org/10.1016/j.resconrec.2020.104750

**30.** Razza F, Fieschi M, Degli Innocenti F, Bastioli C. Compostable cutlery and waste management: An LCA approach. Waste manag. 2009 Apr 1; 29(4):1424–33. https://doi.org/10.1016/j.wasman.2008.08.021 PMID: 18952413

**31.** Zhou X, Zhai Y, Ren K, Cheng Z, Shen X, Zhang T, Bai Y, Jia Y, Hong J. Life cycle assessment of polycarbonate production: Proposed optimization toward sustainability. Resour Conserv Recycl. 2023 Feb 1; 189:106765. https://doi.org/10.1016/j.resconrec.2022.106765

**32.** Zampori L, Dotelli G. Design of a sustainable packaging in the food sector by applying LCA. The Int J Life Cycle Assess. 2014 Jan; 19:206–17. https://doi.org/10.1007/s11367-013-0618-9

**33.** Kin I, Ohta G, Teraishi K, Watanabe K. "Solvents containing cycloakyl alkyl ethers and process for production of the ethers." US Patent 7,494,962. Feb 24, 2009.

**34.** Department for Environment, Food & Rural Affairs. 2011 Guidelines to Defra/DECC's GHG Conversion Factors for Company Reporting: Methodology Paper for Emission Factors. PB13625. DEFRA. London: DEFRA; 2011.

**35.** Rizan C, Bhutta MF, Reed M, Lillywhite R. The carbon footprint of waste streams in a UK hospital. J Clean Prod. 2021;1; 286:125446. https://doi.org/10.1016/j.jclepro.2020.125446

**36.** National Renewable Energy Laboratory. Solvent vapor recovery. DOE/GO-102000-0894. Golden, CO: NREL; 2000.

**37.** Cole C, Osmani M, Quddus M, Wheatley A, Kay K. Towards a zero waste strategy for an English local authority. Resour Conserv Recycl. 2014;1; 89:64–75. https://doi.org/10.1016/j.resconrec.2014.05.005

**38.** Yadav V, Sherly MA, Ranjan P, Tinoco RO, Boldrin A, Damgaard A, Laurent A. Framework for quantifying environmental losses of plastics from landfills. Resour Conserv Recycl. 2020; 1; 161:104914. https://doi.org/10.1016/j.resconrec.2020.104914

**39.** Eppendorf. Eppendorf Tubes Biobased. 2023 [Cited 22 May 2023]. Available from: https://www.eppendorf.com/gb-en/eShop-Products/Laboratory-Consumables/Tubes/Eppendorf-Tubes-BioBased-p-PF-4440301.

**40.** Nessi S, Sinkko T, Bulgheroni C, Garcia-Gutierrez P, Giuntoli J, Konti A, Sanye-Mengual E, Tonini D, Pant R, Marelli L. Comparative Life-Cycle Assessment (LCA) of Alternative Feedstock For Plastics Production. Draft report for Stakeholder Consultation. 2020:2–10.

**41.** Ferrara C, De Feo G, Picone V. LCA of glass versus pet mineral water bottles: An italian case study. Recycling. 2021;15; 6(3):50. https://doi.org/10.3390/recycling6030050