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TOPICAL REVIEW

# The health impacts of waste-to-energy emissions: a systematic review of the literature

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## Abstract

Waste-to-energy (WtE) processes, or the combustion of refuse-derived fuel (RDF) for energy generation, has the potential to reduce landfill volume while providing a renewable energy source. We aimed to systematically review and summarise current evidence on the potential health effects (benefits and risks) of exposure to WtE/RDF-related combustion emissions.

We searched PubMed and Google Scholar using terms related to health and WtE/RDF combustion emissions, following PRISMA guidelines. Two authors independently screened titles, abstracts and then full-texts of original, peer-reviewed research articles published until 20th March 2020, plus their relevant references. Overall quality of included epidemiological studies were rated using an amended Navigation framework.

We found 19 articles from 269 search results that met our inclusion criteria, including two epidemiological studies, five environmental monitoring studies, seven health impact or risk assessments (HIA/HRA), and five life-cycle assessments. We found a dearth of health studies related to the impacts of exposure to WtE emissions. The limited evidence suggests that well-designed and operated WtE facilities using sorted feedstock (RDF) are critical to reduce potential adverse health (cancer and non-cancer) impacts, due to lower hazardous combustion-related emissions, compared to landfill or unsorted incineration. Poorly fed WtE facilities may emit concentrated toxins with serious potential health risks, such as dioxins/furans and heavy metals; these toxins may remain problematic in bottom ash as a combustion by-product. Most modelling studies estimate that electricity (per unit) generated from WtE generally emits less health-relevant air pollutants (also less greenhouse gases) than from combustion of fossil fuels (e.g. coal). Some modelled estimates vary due to model sensitivity for type of waste processed, model inputs used, and facility operational conditions.

We conclude that rigorous assessment (e.g. HRA including sensitivity analyses) of WtE facility/technological characteristics and refuse type used is necessary when planning/proposing facilities to protect human health as the technology is adopted worldwide.

| | |
|---|---|
| **ADCO** | address concentration |
| **$CO_2$ eq.** | carbon dioxide equivalent |
| **CR** | cancer risk |
| **CRP** | carcinogenic risk potential |
| **DALYs** | disability adjusted life years |
| **EBD** | environmental burden of disease |
| **GWP** | global warming potential |
| **HAPs** | hazardous air pollutants |
| **HCl** | hydrogen chloride |
| **HI** | hazard index |
| **HQ** | hazard quotient |
| **HIA/HRA** | health impact/risk assessment |
| **IARC** | International Agency for Research on Cancer |
| **ILCR** | increased lifetime cancer risk |
| **LCA** | life cycle assessment |
| **MSW** | municipal solid waste |
| **NHL** | non-Hodgkin's lymphoma |
| **$NO_x$** | nitrogen oxides |
| **OR** | odds ratio |
| **PAH** | polyaromatic hydrocarbons |
| **PM** | particulate matter |
| **PNC** | particle number concentration |
| **PCDD/Fs** | polychlorinated dibenzo-p-dioxins and dibenzofurans |
| **Rc** | excess cancer risk |
| **RDF** | refuse-derived fuel |
| RR | relative risk |
| **$SO_x$** | sulfur oxides |
| **$SO_2$** | sulfur dioxide |
| **UFPs** | ultrafine particles |
| **UK** | United Kingdom |
| **WtE** | waste-to-energy |
| **YLL** | years of life lost |

## 1. Introduction

Global waste generation has been estimated to double in the decade from 2015 to 2025, from 3 to over 6 million tonnes of waste per day; this rate is expected to continue into the next century, when the estimate increases to 11 million tonnes per day (World Energy Council 2016). In parallel, the world is facing an energy sustainability crisis. Heightened electricity consumption increases energy demand, while conversely, greenhouse-gas emissions must be curbed to mitigate climate change. Sustainable energy and waste management requires policies that promote a 'circular economy', balancing product life cycles (from production to disposal), and that minimise adverse economic, environmental, and societal impacts (Beyene *et al* 2018, IEA Bioenergy 2018). A circular economy reuses and recycles goods, where possible, restoring and regenerating products, components and materials to be at their highest utility and value at all times (IEA Bioenergy 2018). The process of waste-to-energy (WtE; also known as 'energy-from-waste') supports a circular economy by reducing landfill volume from municipal solid waste (MSW) by up to 80%, while also generating energy such as through combustion for turbine-driven electricity (Beyene *et al* 2018, U.S. Energy Information Administration 2018).

Combustion of MSW is the most established method of energy recovery through WtE worldwide, accounting for nearly 90% of the WtE sector (Clean Energy Finance Corporation 2015, World Energy Council 2016). MSW includes domestic, commercial and institutional waste such as plastics, rubbers, wood, metals and paper, which may be combustible or recyclable. The combustible component of MSW is known as refuse-derived fuel (RDF), and is used in a thermal process (incineration, pyrolysis, or gasification) to generate electricity, or heat, fuel gases, and solids as primary recovery products (Beyene *et al* 2018). From a health perspective, the WtE process may have advantages compared to waste management practices that solely rely on landfill sites that are associated with contamination of the air (e.g. volatile organic compounds) alongside water and soils (Vrijheid 2000). However, the WtE process may emit higher concentrations of carbon dioxide ($CO_2$), sulfur dioxide ($SO_2$), and nitrogen oxides ($NO_x$) per unit electricity produced compared to other forms of energy such as natural gas or renewables (O'Brien 2006). The WtE process involves the combustion of RDF components for which emissions may also include persistent organic pollutants such as dioxins (Albores *et al* 2016). This concern is offset, to some extent, in modern, well-run WtE plants that emit lower concentrations of these pollutants compared to coal and oil-fired power plants or traditional incineration of MSW (US EPA 2016). Hence, WtE processes may have both beneficial and adverse impacts on the emission of airborne toxins, and consequently on health, relative to alternative waste disposal and energy generation processes.

The WtE sector is already well established in Europe and provides up to 8% of electricity and up to 15% of domestic heating needs (World Energy Council 2016, Zafar 2018). As of 2008, 475 European WtE plants processed an average of 59 million tonnes of MSW creating revenue of US$4.5 billion each year (Zafar 2018). In Scandinavia, Denmark repurposes 54% of its MSW as RDF (Zafar 2018). Meanwhile Sweden, which has employed WtE since the 1940s, is aiming to match the repurposing of 99% of its local MSW (two million tonnes annually) with an equivalent amount of imported MSW as RDF (Fredén 2018). In 2012, approximately 600 WtE plants across 35 different countries were estimated to combust 130 million tonnes of MSW (Hoornweg and Bhada-Tata 2012), with the sector growing at a compounded annual rate of nearly 10% (World Energy Council 2016). Outside of Europe, the process is being adopted with eagerness, using the established Waste Incineration Directive (WID 2000/76/EC) of the European Commission as a guide for monitoring and regulating WtE emissions (Clean Energy Finance Corporation 2016). In 2016, the USA alone operated

71 WtE plants generating approximately 14 billion KWh of electricity from 30 million tonnes of RDF (U.S. Energy Information Administration 2018). In the Asia-Pacific region, China is the fastest growing adopter of WtE, recently planning 125 new plants to double national capacity (World Energy Council 2016, Zafar 2018). China, one of the major importers of MSW, has restricted imports of certain materials (e.g. plastics, paper) to reduce local widespread environmental contamination (Retamal *et al* 2019), challenging major exporters of MSW such as Australia (Cheng and Hu 2010, Downes and Dominish 2018). Responding to this challenge, Australia has estimated that a national shift towards WtE presents an opportunity to repurpose 20+ million tonnes of MSW otherwise going to landfill annually and avoid 9 million tonnes of $CO_2$ (equivalent) emissions by replacing fossil-fuel combustion while meeting 2% of national baseload electricity demand (Clean Energy Finance Corporation 2016). Hence, it is timely to consider the place of WtE in the energy transitions landscape and, in particular, to consider its impact on air quality and health.

Despite the growing global interest in WtE, the public health implications of combusting RDF remains little studied. There has been no previous systematic literature review of the health impacts associated with WtE, although several reviews on municipal waste incineration have been published. In 2019, a systematic review on the evidence of health effects from waste incineration (2002 to 2017) was published in response to several new incinerators proposed for use within Australia (Tait *et al* 2020). The literature review, which did not include WtE facilities, concluded that the available evidence likely underestimated the health effects of exposure to incineration emissions due to most studies being of low quality and only examining a limited subset of potential exposure and disease pathways (Tait *et al* 2020). Other earlier reviews on the health impacts or risks of incineration and resulting emissions have focused on hazardous (industrial) or unsorted (municipal) solid waste, rather than sorted RDF for WtE. These reviews concluded that the evidence is insufficient to support an association between a specific waste incineration process and adverse health effects (Vrijheid 2000, Hu and Shy 2001, Giusti 2009, Porta *et al* 2009, Cordioli *et al* 2013). Associations between exposure to emissions and health outcomes such as increased risk of lung/throat cancer or ischaemic heart disease (Hu and Shy 2001), as well as non-Hodgkin's lymphoma and soft-tissue sarcoma (Giusti 2009), have been reported, however, the findings are inconsistent. The reason for this has been suggested to be due to poor methods of exposure characterisation which have relied on distance from source or self-reporting exposure, rather than measured or modelled pollutant concentrations (Hu and Shy 2001, Cordioli *et al* 2013, Hoek *et al* 2018, Tait *et al* 2020). More consistent associations have been reported between exposure to emissions and elevated biomarkers of organic chemicals or heavy metals in urine and blood (Hu and Shy 2001).

In their review on MSW incineration without energy recovery (i.e. not WtE), Tait *et al* (2020), recommended future studies be conducted on the health impacts of WtE, including studying content and volume of feedstock (waste), combustion specifications, consideration of multiple exposure pathways, reporting of a larger array of health outcomes, and controlling for potential confounding factors (Tait *et al* 2020). Other reviews have suggested that previous limitations of incineration studies could be addressed by large, prospective, multi-site cohort studies with personal measurements of exposure, based on knowledge of biological pathways and toxicological effects of specific compounds (Giusti 2009, Porta *et al* 2009, Hoek *et al* 2018), however such studies can be expensive and sample size (of the study population) can be a limiting factor. Clearly, the expanded interest in WtE facilities yet current lack of evidence on health impacts with their operation requires stringent oversight to safeguard environmental and health outcomes.

Our aim was to conduct a systematic review on the potential health effects associated with exposure to airborne emissions from WtE processes (including RDF combustion). The primary motivation for the current review was the perceived lack of data on the potential for health impacts of WtE processes and emissions, and the increasing growth in demand in regions where WtE has not yet been adopted on a widespread basis. As WtE has been promoted worldwide as a potentially sustainable form of both waste management and electricity generation, we considered it timely to ascertain the extent and breadth of evidence from published studies with health-related data or information associated with airborne emissions from WtE processes. The different types of study designs for studies included in our review include epidemiological, environmental monitoring, health risk assessments/health impact assessments, and life-cycle analyses (detailed below in Methods).

## 2. Methods

### 2.1. Literature search strategy

We conducted a systematic search of PubMed and Google Scholar, supplemented by a hand search of bibliographies of the articles included for full text screening. We used PubMed as the primary database source given our review was focused on health outcomes and PubMed is considered to be the most comprehensive health database. We used Google Scholar as a secondary source to identify relevant literature that PubMed does not catalogue, as done previously for hazardous waste reviews (Cordioli *et al* 2013).

Search terms and the Boolean operators (string) that we used were as follows:

> "air" AND "health" AND "energy" AND "waste"
> AND "energy from waste" OR "waste to energy" OR "incineration" OR "refuse derive$ fuel"
> AND "air pollution" OR "air quality" OR "emission"

'Incineration' was chosen as it is the industrial term that represents 'combustion' and 'burning'. In addition, 'air pollution', 'air quality' or 'emission' terms were used to avoid pollutants or hazards associated with other emissions. Two investigators (TCH, CC) independently screened titles, abstracts and full-texts for inclusion or exclusion of articles. Where there was variation between the two investigators, this was resolved by reviewing the the full-text article a second time until agreement was reached.

The inclusion criteria used for selection of eligible articles were as follows:

(a) Published in English.
(b) Published up to and including the 20th March 2020.
(c) Included an abstract and be full-text accessible.
(d) Reported original research.
(e) Published in a peer-reviewed journal.
(f) Related to anthropogenic waste, municipal solid waste, air pollution emissions, and relevant to human health.

Exclusion criteria included articles that related to:

(a) Hospital or medical waste, composting of waste, or agricultural waste.
(b) Combustion of biomass fuel for cooking and heating in low-income settings.
(c) Review papers.

### 2.2. Literature review and synthesis

We followed the approach (criteria) suggested by the PRISMA guidelines for performing and reporting the flow of a literature review process (e.g. figure 1) (Moher *et al* 2009). We synthesised study findings by grouping the articles by different study designs (methods). We used the following groupings: epidemiological (examining direct associations between exposure and health risk); environmental monitoring (emissions or exposure assessments or modelling); health risk assessment (focused, standard methodology to estimate risks related to a single or a mix of pollutants; applying health risk estimates from epidemiological studies to quantify the health burden due to the exposure of interest in a defined population), or health impact assessment (broader methodology that assesses the public health impacts to inform decision making; often including HRA methods or other health risk findings) (Gulis 2017); and, life-cycle analyses (LCA; quantifying carbon-related impacts and indirect health impacts, with some LCAs also addressing direct health impacts). We used a standardised series of tables to summarise the studies and to list exposure assessment methods, health outcomes, summary results, and risk of bias. We provided an overall quality rating for epidemiological studies, similar to the *Navigation* framework previously developed (Woodruff and Sutton 2014) and demonstrated (Johnson *et al* 2014). The *Navigation* framework was developed in recognition that usual quality frameworks used for reviewing health studies, such as Cochrane, do not necessarily translate well to studies of environmental exposures, due to the nature of the exposure and difficulty in conducting randomised trials. As there were few relevant epidemiological studies, the criteria were amended slightly to ensure relevance depending on the study design. For example, we included mention of sensitivity analyses in modelling studies or explicit statements about assumptions used in the analyses. However, we did not critically appraise or scrutinise the assumptions or the software used in the LCA models as this was beyond the scope of our study.

## 3. Results and discussion

### 3.1. Literature search results

The PubMed literature search identified 258 relevant primary records (articles) for review. The Google Scholar search identified 11 unique records relevant for review. As such, the complete search gave a combined total number of 269 unique records for consideration.

After two investigators independently reviewed the titles of these 269 records, 137 records were identified to be appropriate for abstract screening (which removed 74 records). Sixty-three records were subsequently identified as eligible for full-text review, leading to the exclusion of 46 records. Finally, 17 full-texts were selected for our review synthesis, plus two of their references to give a total of 19 full-texts to be synthesised (figure 1).

The 19 included articles all related to combustion of MSW as RDF or in WtE facilities or processes. MSW incineration studies were included if they presented information or data related to emissions that were relevant to WtE processes, such as incineration of RDF. All included articles were published in the past 15 years, reflecting the increasing interest in WtE. Most studies comprised health impact or risk assessments/risk modelling (n = 7), followed by life-cycle assessments (n = 5), environmental monitoring studies (n = 5), with only two being epidemiological studies.

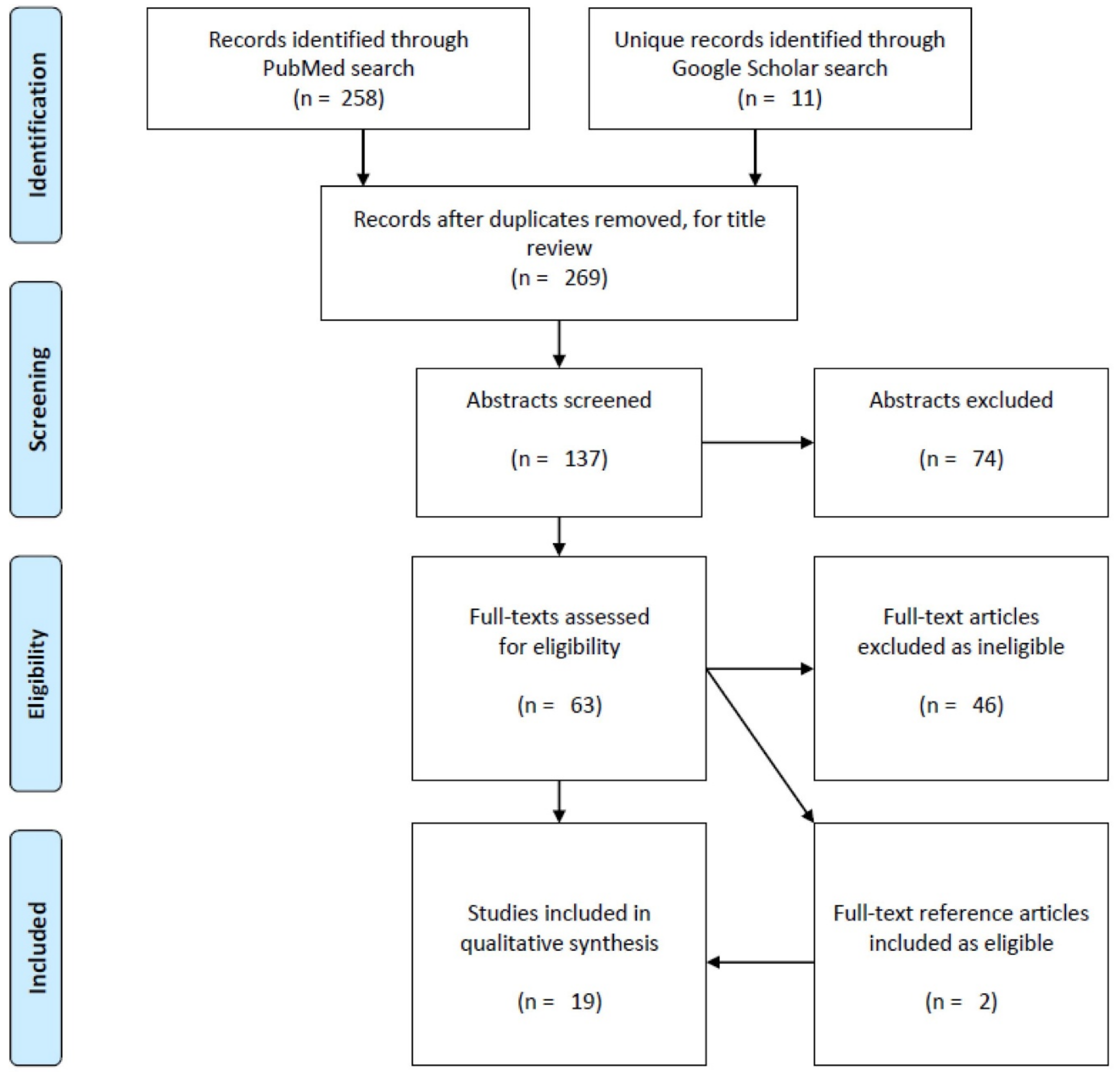


**Figure 1.** Systematic literature review flow diagram.

### 3.2. Synthesis and discussion of findings

To our knowledge, this is the first systematic literature review focused primarily on studies of the health effects associated with WtE-related air emissions. We found that while implementation of WtE technologies is increasing, the majority of incineration-health studies to date do not specifically address the combustion of sorted waste (RDF) for WtE (shown to be different than MSW due to waste composition characteristics by environmental monitoring studies—reported on below). Previous reviews have focused on the health impacts of waste incinerators (Cordioli *et al* 2013, Tait *et al* 2020) the economic implications of WtE technologies (Beyene *et al* 2018), exposure assessment methods in epidemiological studies of industrially contaminated sites (Hoek *et al* 2018) or waste incinerators, and the health impacts of general waste management practices (Giusti 2009). There are numerous epidemiological (e.g. cohort) studies on the health effects of other waste management risks including landfill leaching, sewage contamination and ionising radiation, yet few on air pollution emissions from RDF combustion. Due to the small number (n = 2) of epidemiological studies that directly measured health outcomes associated with WtE processes we believed it was not appropriate to meta-analyse the evidence for WtE health effects. However, we reviewed studies of environmental monitoring and health risk assessments in order to contribute to the evidence base for decision making. The following synthesis details the contributions to this evidence base, from studies detailing process emissions to health risk assessments.

### 3.3. Epidemiological studies of health outcomes

The direct health effects of exposure to emissions from combustion of RDF for WtE have been little studied. This is likely to be partly due to the difficulty of quantifying population health effects from generally inaccurate or low levels of exposure (Vrijheid 2000). This is despite previous recommendations that large prospective cohort studies with direct exposure and biomarker measurements be preferentially funded and performed (Giusti 2009).

We found only two epidemiological studies relevant to exposures to WtE facilities or RDF emissions. One epidemiological before/after cohort study was performed in Italy among 380 individuals residing near a new WtE facility, with exposure assessed before and one year after operation began (Ruggieri *et al* 2019). In this biomonitoring study, chromium (but not other heavy metal) concentrations were higher in the urine of participants predicted to be exposed to WtE emissions compared to unexposed but otherwise comparable participants (Ruggieri *et al* 2019). However, this finding was applicable in both the baseline and follow-up year, and so the result cannot be directly attributed to operation of the WtE facility. Interestingly, concentrations of other heavy metals were higher in the control subjects, and so were attributed to other sources of personal exposure such as fish intake (arsenic) and tobacco smoke (cadmium) (Ruggieri *et al* 2019). Hence, residing near the WtE plant was not associated with greater exposure to heavy metals. We considered the study to be of good quality, having used dispersion modelling to assign exposures and conducting before and after health outcome measurements in an 'exposed' and 'unexposed' group. Validation of the emissions modelling by environmental sampling could have improved exposure assessment. There is further follow-up planned for this cohort which is expected to provide additional data (Ruggieri *et al* 2019).

A recently published birth cohort study was conducted in Taiwan and investigated childhood social development in children residing near an incinerator (Lung *et al* 2020). The study of nearly 20 000 subjects (for which approximately five percent were considered exposed), reported a transitory negative effect on childhood social development, for children living within 3 km of a MSW incinerator, although this effect was apparent at six months but not evident at 18 months. A limitation of this study was considered to be the coarse exposure assessment applied to subjects which has the potential to lead to exposure misclassification. Exposure assessment ('whether there were incinerators within 3 km of their place of residence') and health outcome reporting were both coarse and subjective, with both being self-reported by parents (Lung *et al* 2020).

We conclude that the results from the two epidemiological studies provide little evidence of an adverse impact of WtE air emissions on health outcomes. See table 1 for further details of included epidemiological studies.

### 3.4. Environmental monitoring

While studies of emissions inventory profiles do not include health outcomes, they may provide valuable information on the potential pathways and hazards posed by incineration of MSW components comprising RDF, with the potential for carcinogenic or toxic emissions relevant to WtE processes. Our review found five articles which reported on emissions testing and environmental monitoring of WtE facilities. In general, we found that the articles related to emissions monitoring predominantly fell into three categories: (1) the first related to estimating pollutant emissions of concern; (2) the second related to the need for monitoring to ensure efficacy of treatment technologies in removing/reducing pollutants; and (3) the third related to the need for appropriate monitoring to determine the influence of the feedstock on pollutant formation.

Of greatest concern for health is the combustion of plastic MSW (composed of hydrocarbon/oil-products), which is concentrated in RDF for WtE, and which emits organic and chlorinated/fluorinated compounds (e.g. dioxins), polychlorinated biphenyls, furans, chlorophenols, and mono- and polycyclic aromatic hydrocarbons (Karunathilake *et al* 2016). Notwithstanding, two environmental monitoring studies reported that after WtE upgrades to an Italian incinerator facility which included stricter emission-control measures primarily aimed at reducing dioxin emissions, particulate matter (PM) emissions also declined (Buonanno *et al* 2010, 2011). This indicates that controlling emissions for critical contaminants such as dioxins and furans may also have a beneficial effect of leading to a reduction in PM, a standard air pollutant. The health risk of toxics predominantly relate to cancer, neurological and adverse birth outcomes and are considered to pose a greater risk to health than the standard regulated air pollutants such as PM and gaseous compounds. However, exposure to even low levels of PM is not benign and many epidemiological studies point to a range of risks associated with PM including increased risk of mortality, cardiovascular morbidity, lung cancer and more (Hime *et al* 2018). Thus changes to existing treatment facilities that improve emissions controls for both types of pollutants are beneficial from the standpoint of exposure minimisation.

The review also reports on articles which compared or discussed monitoring campaigns and/or trials of varying technologies. In one study a two-stage dry treatment system was shown to remove harmful acid gases (hydrogen chloride, $SO_2$) from WtE emissions even with a widely-varying (potentially highly chlorinated) waste stream (Dal Pozzo *et al* 2016). This is an example of a monitoring program which can help provide evidence of efficacy of treatment technologies. Two articles reported on the influence of feedstock on pollutant concentration emissions.

The articles indicated that rather than combusting RDF directly for electricity generation, experimentation suggested that mixing certain proportions of RDF components (e.g. certain plastics, wood chips) with traditional fuels (e.g. coal) for combustion and

**Table 1.** Epidemiological studies of health outcomes of waste-to-energy/RDF processes.

| Author/s, Year, Country | Design | Study population | Exposure and confounding assessment | Results | Risk of bias[a] | Overall assessment [b] |
|---|---|---|---|---|---|---|
| Lung *et al* 2020, Taiwan | Design: Longitudinal study of effects of living near an incinerator. Outcome: Child development at 6, 18, 36, 66 months [Developmental index (Taiwan Birth Cohort Study Developmental Instrument)] | N = 19 516 children (Taiwan Birth Cohort Study) | Self-reported proximity (living within 3 km of incinerator). Confounding: Self-reported breastfeeding, urbanisation | Reduced development at 6 mths only ($\beta = -0.06$, $p = 0.013$) No effect at other timepoints | Low. Exposure: + Outcome: ++ Confounding: + Conflict: + (*none*) | Study quality: Moderate Health concern: Moderate |
| Ruggieri *et al* 2019, Italy | Design: Before (2013)/1-yr after (2014) after Gerbido (Turin) WtE plant operational (one of largest in Europe). Outcome: Urine (morning spot samples) analysis of 18 metals | N = 380 adults; 186 exposed; 194 unexposed. Response rate at follow-up high (approx. 96%). | Dispersion modelled predictions of total metal content (mg $m^{-3}$ $yr^{-1}$) conducted at baseline to determine exposed (0.014–0.11)/ unexposed (0–0.007). Incl. of unexposed (control) group. Confounding: Self-reported diet, sociodemographics, etc. | There were reduced levels of most metals in 2014 (post) compared to 2013 (pre) in both WtE-exposed and unexposed groups Exception: Increased Cr in exposed; Reduced Mn in unexposed | Low. Exposure: ++ Outcome: ++ (*Appropriate lab analyses QA/QC*) Confounding: ++ Conflict: + (*none*) | Study quality: Moderate Health concern: Low |

[a]Exposure assessment; Outcome assessment; Control for confounding; Study sample size; Conflict of interest statement
[b]Low, moderate or high potential for concern based on quality and outcomes of study

to replace electricity for industrial heating applications (e.g. cement kilns), has the potential to reduce sector or total emissions of health-relevant chemicals (e.g. dioxins, mercury) (Chen *et al* 2014, Richards and Agranovski 2017).

We conclude from the results of the environmental monitoring studies that there is a need for regulation of the feedstock used (e.g. removing food waste) for RDF and WtE facilities to maximise complete combustion and minimise carcinogenic/-contaminant emissions (e.g. volatile organic compounds), more so than the treatment technology used. See table 2 for further details of included environmental monitoring studies.

### 3.5. Health risk/impact assessment studies

We found seven studies comprising HRAs or HIAs of WtE facilities or RDF emissions. In table 3 we outline the health outcome assessed in a majority of the HRAs, including the hazard index (HI), hazard quotient (HQ), lifetime cancer risk (LCR) and other indices (4th column). These represent indices where cancer and non-cancer risks are considered for various chemicals of concern (3rd column, table 3), e.g. heavy metals, VOCs, organic compounds such as dioxins and furans, and so on. Some of the HRAs/HIAs also considered air pollutant emissions such as $NO_x$, PM, and sulfur oxides ($SO_x$). The risk of exposure is based on modelled estimates of the chemicals/pollutants emissions from each WtE facility or alternative waste disposal method. Some of the studies have used proprietary software which includes the exposure-response functions for the chemical/pollutant of concern, which we list in table 3 (3rd column).

These studies generally showed that the risk to or impact on health from exposure to WtE and RDF incineration emissions are not substantially elevated above ‘background’ risk levels (Mindell 2005, Roberts and Chen 2006, Krajčovičová and Eschenroeder 2007, Rovira *et al* 2010, Ollson and Whitfield Aslund *et al* 2014). They also point to lower emissions from well-run WtE facilities compared to landfill (Paladino and Massabò 2017) and traditional incineration (Krajčovičová and Eschenroeder 2007) or when RDF is substituted for fossil fuel for incineration (Rovira *et al* 2010).

Six of the HRA studies estimated that exposure to WtE emissions was unlikely to increase incremental LCR or HQ for cancer risk (Roberts and Chen 2006, Krajčovičová and Eschenroeder 2007, Rovira *et al* 2010, Ollson and Knopper *et al* 2014, Li *et al* 2015, Paladino and Massabò 2017). Two HRAs reported lower cancer risk for exposure to WtE emissions compared with incineration emissions (Karunathilake *et al* 2016) or as substitution of RDF for fossil fuels in cement production (Rovira *et al* 2010). One HRA estimated that cancer risk from exposure (all pathways) to WtE emissions (mainly dioxin) would be lower than for exposure to landfill emissions, and estimated that agricultural (milk and meat) product ingestion was a more important exposure pathway than for inhalation of WtE emissions (Paladino and Massabò 2017).

A health risk assessment conducted in Slovakia compared a traditional open-air (uncontained) MSW incinerator with a modern WtE plant, and found that the former increased the cancer risk 10–80 times above the background level, while the WtE plant presented a less than one-in-a-million excess risk of cancer (Krajčovičová and Eschenroeder 2007). That HRA estimated a substantially decreased cancer risk when MSW is sorted for RDF and its incineration emissions are properly controlled (contained) as advocated in modern, well-run WtE. In China, a more recent HRA estimated that under normal conditions, operational levels of emissions from WtE are unlikely to cause adverse health (incremental lifetime cancer) risks among nearby residents, with risks estimated for lifelong exposure through direct inhalation of ambient emissions and landfilling of bottom/solid ash residues (Li *et al* 2015). The exception to this was risk of chromium exposure which slightly exceeded the tolerance value (Li *et al* 2015). However, Li *et al* (2015) reported that all scenarios tested were sensitive to the model inputs and estimated that during abnormal operation (e.g. malfunction of control systems) the WtE facility could also carry an elevated risk due to inhalation of acid gas (hydrogen chloride).

Four HRAs assessed non-cancer risks (Mindell 2005, Roberts and Chen 2006, Ollson and Knopper *et al* 2014, Li *et al* 2015). Ollson *et al* (2014a) estimated that abnormal operation of a Canadian WtE facility could lead to infant consumption of breast milk contaminated with dioxins and furans (Ollson and Knopper *et al* 2014). Two of the HRAs (Li *et al* 2015, Karunathilake *et al* 2016) estimated no increased risk of non-cancer health effects from operation of their respective WtE facilities. Modelling studies of UK WtE plants estimated premature (total non-traumatic) deaths and respiratory-related hospital admissions to be less than or equal to one-in-a-million above background rates (Mindell 2005), and overall risk of dying to be 1 in 4 million for any year (Roberts and Chen 2006). It should be noted that these UK studies were either funded by the proponent company for the WtE facility, or written by previous employees of related boards/companies.

It is clear from these studies that the choice of scenarios and model inputs can influence the risk findings, and so it is important that sensitivity analyses be conducted. Of note, Li *et al* (2015) reported that all of the scenarios studied in their analyses (WtE, landfill, and material recovery and composting) were sensitive to the inputs used for the reference concentrations and the landfill gas collection rates. In sensitivity analyses, the HI for the WtE option increased the most, indicating that there needs to be careful selection of the reference criteria values and that sensitivity

**Table 2.** Environmental monitoring of waste-to-energy/RDF processes.

| Author, Year, Country | Study design and Outcome | Emission/Exposure metric | Results | Strength/Limitations[a] | Overall assessment[b] |
|---|---|---|---|---|---|
| Richards and Agranovski 2017 Australia | Design: Monitoring of stack emissions during the substitution of alternative fuels Outcome: Emission factors (kg/tonne) | Health-critical dl-PCB (dioxin-like polychlorinated biphenyl) congeners. 10 Australian cement plants during normal operation (baseline) and varied substitution fuel rates. | Reduced dl-PCB; Dioxins with substitution of coal for RDF | Exposure: + Sample: ++ Conflict:—(*unreported*) | Quality of evidence: Moderate Hazard: Decreased |
| Dal Pozzo *et al* 2016 Italy | Design: Simulated comparison of novel treatment and three benchmark alternatives Outcome: Optimal operating configuration for different reference waste compositions | Acid gas removal from waste and flue gas compositions. Total annual cost of operation (operating and capital costs) | Reduced acid gas emissions with flue gas with novel treatment. Facilities operating single stage technologies with high chlorine-content waste, the implementation of a lime injection line before the pre-dusting section is a retrofit that should be considered | Exposure: + Sample: + (*Monte Carlo sensitivity analysis performed to assess robustness of results*) Conflict:—(*unreported*) | Quality of evidence: Moderate Hazard: Decreased |
| Chen *et al* 2014 China | Design: Sampling of flue gases from five typical cement kilns during 12 runs Outcome: Levels and distribution of PCDD/Fs (dioxins/furans) | Emission concentrations of PCDD/Fs from cement kilns. I-TEQ values | Negligible difference in PCDDs (dioxins) when RDF combusted as supplemental fuel. PCDD/Fs: 2.3 to >40 ng $m^{-3}$ I-TEQ values: 9.3 ∼ 90.8 × 10–3 ng I-TEQ/$m^3$ | Exposure: + Sample: ++ Conflict:—(*unreported, except for national research funding*) | Quality of evidence: Moderate Hazard: Low |
| Buonanno *et al* 2011 Italy | Design: Experimental campaign to monitor ultra-fine particles (UFP) at an incinerator Outcome: Filter efficiency; particle size, concentrations, characterisation | UFPs; heavy metals. Instrumental Neutron Activation Analysis (INAA), Transmission Electron Microscopy (TEM) with Energy Dispersion Spectrometer (EDS). | Reduced particle number concentration with filter use. Pre-filter: 2 700 000. Post-filter: 2000 | Exposure: - Sample: ++ (*Advanced instrumentation for particle monitoring*) Conflict:—(*unreported*) | Quality of evidence: Strong Hazard: Decreased |
| Buonanno *et al* 2010 Italy | Design: Characterisation of emitted particle size distribution and chemical composition Outcome: Particle size, concentrations, compositions (*Annual mean values*) | Continuous measurement of environmental particles by SMPS/APS system over 12 months, as downwind receptor of a typical incinerator | Negligible particle number and mass concentration versus background (other sources) $8.6 \times 10^3 \pm 3.7 \times 10^2$ p/$cm^{-3}$ $31.1 \pm 9.0$ $gm^{-3}$ Based on chemical composition most elements attributed to long-range transport (natural/anthropogenic sources) "typical of a rural site" | Exposure: ++ Sample: ++ Conflict:—(*unreported*) | Quality of evidence: Strong Hazard: Low |

[a]Exposure assessment; Study sample size; Conflict of interest statement

[b]Low/decreased, moderate or high potential for concern based on quality and outcomes of study

**Table 3.** Health risk/impact assessment/modelling studies of waste-to-energy/RDF processes.

| Author, Year, Country | Design | Exposure pathways/receptors | Assessment outcomes | Results | Strengths/Limitations[a] | Overall assessment[b] |
|---|---|---|---|---|---|---|
| Paladino and Massabò 2017, Italy | HRA for human health due to air, surface water, groundwater and soil contamination. Compared alternative WtE management solutions to landfill | Dermal contact, Ingestion of fish, meat, milk, vegetables, water. Leachate: BTEX, TCE, MTBE, Naphtalene, Arsenic, Lead, Nickel, Mercury, Zinc, Ammonia. Inhalation Flue gas emissions: NOx, PM, SOx, HCl, Total VOC, Cadmium, Nickel, Arsenic, Mercury, Dioxins, Furans, PCBs, $CO_2$, CO | Hazard Index (HI)—*chemical Lifetime Average Daily Dose divided by reference dose [mg/(kg d)]*. Cancer Risk (CR). USEPA considers HI < 1, and CR < $1 \times 10^{-6}$ as acceptable | WTE has both reduced HI and CR vs Landfill. Decision matrix HI: <1; CR: <1. Exposure route importance rated by authors (contributing to both toxic and carcinogenic risk). Ingestion: 5/5; Inhalation: 3/5; Dermal: 2/5 | Exposure: + Sample: + Assumptions: + Conflict:—(*unreported*) | Study quality: Moderate Health concern: Low |
| Li *et al* 2015, China | HRA & LCA of waste management options: landfill; WtE; material recovery & compost (700 t $day^{-1}$) | WtE plant design parameters for $SO_2$, HCI, NOx, dioxins, CO, mercury, cadmium, chromium, lead and PM; modelling of life cycle inventory and air dispersion via integrated waste management (IWM-2) and Screening Air Dispersion Model (Version 3.0) (SCREEN3) | Hazard Index (HI) Hazard Quotient (HQ) Increased Lifetime Cancer Risk (ILCR); Non-cancer risks. | No adverse health effects under normal operating conditions. Increased HI (≤1) for WtE option using different reference concentrations. Slight exceedance of ILCR for chromium from WtE option where value slightly exceeds target value, but was within tolerance value. Highest HQ for HCl. Suggests that HRA should carefully select reference concentrations values. | Exposure: + Sample: + (*Use of pre-existing international HRA tools & guidelines [USEPA Risk Assessment Guidance for Superfund Vol I]*) Assumptions: + (*Sensitivity analyses*) Conflict: none | Study quality: Moderate Health concern: None found |
| Rovira *et al* 2010, Spain | HRA comparing 100% to 85% fossil fuel combustion (i.e. 20% RDF substitution) for cement production. Carcinogenic risks | Sampling of emission levels, environmental (air, soil, etc) concentrations, exposure concentration | Carcinogenic risks (unitless) derived from exposure to metals and PCDD/Fs for population living in vicinity of cement plant. | Reduced total carcinogenic risk from PCDD/Fs 100% fossil fuels: $3.84 \times 10^8$; 85% fossil fuels +15% RDF: 3.44 x $10^8$ | Exposure: + (*short-term monitoring—longer-term monitoring needed to ensure that further use of RDF will not change emission levels of PCDD/Fs*) Sample: + Assumptions: + Conflict: + (*funding from facility*) | Study quality: High Health concern: Low |

**Table 3.** Health risk/impact assessment/modelling studies of waste-to-energy/RDF processes.

| Author, Year, Country | Design | Exposure pathways/receptors | Assessment outcomes | Results | Strengths/Limitations[a] | Overall assessment[b] |
|---|---|---|---|---|---|---|
| Ollson and Whitfield Aslund *et al* 2014,Canada | HRA for air, soil, water and biota contamination in area around WtE facility. Chemicals of potential concern (COPCs). Criteria Air Contaminants (e.g. HCl) and Chlorinated Polycyclic Aromatics (dioxins, furans) | Stack testing results for COPCs. Dispersion modelled (*CALPUFF*) prediction of ground-level COPC concentrations. Inhalation of vapours and PM emissions, ingestion and dermal exposure to soil and/or dust, food chain exposures | Concentration Ratio Hazard Quotient (HQ) Lifetime Cancer Risk (LCR) Non-cancer health | Highest concentrations of emissions and depositions located in area immediately surrounding facility (radius of ~10 km). Varied HQ. Varied LCR | Exposure: ++ Sample: + Assumptions: + Conflict: none | Study quality: High Health concern: None found |
| Krajčovičová and Eschenroeder 2007, Slovakia | HRA comparing municipal incineration to WtE combustion of solid waste. 3 prototype villages, representing ~one-third of rural population, formulated by agricultural land use and village population data | Direct exposure: Dispersion modelled (*CALPUFF*) Indirect exposure: *EMERAM* software (developed in this paper). Inhalation, soil ingestion, breastmilk ingestion, homegrown or commercial food consumption, water consumption | Hazard Index (HI); Incremental Lifetime Cancer Risk (ILCR) [cases per million population] | WtE has reduced ILCR vs incineration HIs Incineration: <1/million WtE: <1/million ILCRs Incineration: <400/million WtE: <1/million | Exposure: ++ Sample: ++ Assumptions: + Conflict:—(*unreported*) | Study quality: Moderate Health concern: Low |
| Roberts and Chen 2006, UK | HRA using US EPA HHRAP for Hazardous Waste Combustion Facilities and UK coefficients | Dispersion (ADMS3) modelled stack emissions ($SO_2$, $PM_{2.5}$) over 25 years in a population of 25 398 within 5.5 km of the stack. Concerns (quality of life) measured through interviews and discussion | Additional cancers/~25 k population. Additional deaths. Quality of life (anxiety, employment, noise, occupational risks, road accidents, and reduced use of landfill) | Negligible additional cancers and deaths due to WtE facility emissions Additional cancers: 0.02 Additional deaths: 0.48 ($SO_2$: 0.46, $PM_{2.5}$: 0.02). Negligible overall risk of premature death due to exposure to emissions: 1 in 4 million per population year | Exposure: ++ Sample: ++ Assumptions: + (*Authors noted methods and assumptions used are useful for purposes of illustration but are not epidemiological projections*) Conflict: + (*previous employment by facility-related board/company*) | Study quality: High Health concern: Low |
| Mindell 2005, UK | Development of computer modelling approach for linking air pollution—environmental effects to health impacts. N = 3 500 000 | Dispersion modelled concentrations of air pollution (e.g. $PM_{10}$) via GIS | Deaths. Respiratory disease hospital admissions. Events (vs background) | Negligible additional admissions and deaths vs background events Short-term 0.03 deaths/year 0.04 admissions/year Long-term 1.8–7.8 deaths/30-years | Exposure: ++ Sample: ++ Assumptions: + Conflict: + (*funding by company proposing to build facility*) | Study quality: High Health concern: Low |

[a]Exposure assessment; Study sample size; Appropriate assumptions; Conflict of interest statement
[b]Low, moderate or high potential for concern based on quality and outcomes of study

analyses are crucial for better understanding operational limitations of WtE facilities and to avoid abnormal operations or malfunction events.

Together, we conclude that the HRA results show that under normal operating conditions there is little to no evidence of an increased risk of cancer or non-cancer effects in humans, as WtE facilities are capable of lower emissions (except for a predicted potential higher emission of chromium) than existing waste management practices of landfill and traditional incineration. However, close attention is required to ensure operational limits are not exceeded, as such conditions are estimated to be associated with increased risk of dioxin exposure (one HRA) and potentially hydrogen chloride gas exposure (one HRA). This highlights the need for appropriate sensitivity analyses to be conducted during the HRA process, along with careful selection of reference health criteria and consideration of the fuel used for combustion. See table 3 for further details of included *health risk/impact assessment studies*.

### 3.6. Life-cycle analyses

A total examination of the environmental, social, and economic impact associated with all stages of a product's life, from raw material extraction to final product disposal (e.g. landfill or WtE), is termed a life-cycle analysis/assessment (LCA) (Muralikrishna and Manickam 2017). An LCA is distinct from a HRA in that an LCA considers the full life-cycle of a product, from production to disposal, while a HRA typically only considers one stage of the life-cycle while focusing on a health impact. For example, an LCA for WtE will consider the impacts of not only the resulting toxic contaminants in ash and air emissions, but also emissions which have a greenhouse gas impact such as carbon dioxide, as well as the impact of the fuel used for the WtE facility. Besides health impacts, LCAs can determine equitability of a product's environmental impact, and can determine if overall impact (both to health and the environment) of one waste management process is more favourable than another. For example, one may ask if exposure to atmospheric emissions from WtE is less harmful to health than from unsorted (mass) waste incineration or landfill leachate, also taking into account health impacts of climate change due to greenhouse gas emissions from each technology—however, none of the studies considered climate change in relation to health outcomes. Future LCA studies of new energy technologies could be important in estimating direct and immediate health impacts (due to a change in pollutant emissions) balanced with estimating the potential for indirect and delayed health impacts due to increased greenhouse gas emissions from climate change.

Nevertheless, our review reports the results of five LCAs. As with the HRAs reported above, two of the LCAs predict lower pollutant emissions from combustion of RDF (sorted for WtE), compared to incineration of unsorted MSW (still producing electricity). A Canadian LCA for a WtE facility estimated lower cancer and non-cancer health risk per unit of electricity generated with RDF than for unsorted MSW incineration (Karunathilake *et al* 2016). Similarly, a lower health risk was attributable to the sorting and use of relatively high calorie, low toxicity waste for RDF (e.g. wood, paper, plastics, textiles, and rubbers; sorted, treated, shredded, and combusted to produce approximately 4 MWh of energy per tonne) (Reza *et al* 2013) compared to the use of coal. Although Reza *et al* estimated lower heavy metal emissions for RDF used in WtE facilities, an exception was an estimated increase in lead emissions. This LCA was conducted with a focus of comparing environmental benefits of the two feedstocks for use in cement kilns.

Two of the LCAs estimated greater impacts from WtE processes compared with other waste management processes. Scipioni *et al* (2009) predicted lower emissions of respiratory related pollutants but greater potential for exposure to carcinogens, climate change pollutants (mainly $CO_2$) and radiation, when comparing dry and wet fly gas scrubbing, with and without WtE processing. Tan and Khoo's (2006) LCA analysis comparing landfill, WtE incineration, and recycling and composting stated that 'energy gained from incineration of waste materials is outweighed by the air pollution generated' and estimated that recycling and composting would result in the least ecosystem impact. The authors acknowledged the generally 'wetter' conditions of their MSW (in Singapore) which they suggested might be more suitable for composting. Although the article mentioned modelling of disability adjusted life years (DALYs; as a health measure) we could not see where these results were calculated or presented. In addition, the assumptions used in this LCA were not explicitly mentioned, so it is difficult to determine how a change in model inputs might influence these findings.

In Passarini et al's (2014) LCA which compared various upgrades of an incinerator to enable functioning as a WtE plant, they estimated that concentrations of heavy metals in the fly and bottom ash were the main contributor to carcinogen endpoints and these remained constant over time. However, they estimated decreases in carcinogens and particulate matter in airborne emissions during operation as a WtE facility. The LCA concluded that human health improvements were expected with WtE operations due to both the lowered emissions and the predicted improvements associated with greenhouse gas mitigation.

Most of the LCAs reviewed used accepted international methods for LCA, such as ISO standards along with the use of specialist software such as SimaPro and impact assessment methods such as Ecoindicator99. As with the HRA studies, some of the LCA studies highlighted the variability in calculated health risks to be dependent on the reference criteria and dose and other model inputs, thus indicating the necessity for sensitivity analyses to be conducted.

In general, we conclude that the predictions from the majority of LCA studies indicate that emissions from, and therefore health risks associated with, WtE plants are lower than for landfill and traditional incineration. However, an increased potential for health risk is highlighted for lead (Reza *et al*, 2013) and other heavy metals in the bottom and fly ash (Passarini *et al*, 2014) that may be emitted in later stages of the life cycle (following combustion of RDF for WtE). See table 4 for further details of included *life-cycle analyses*.

### 3.7. Implications

Our review indicates that there is a dearth of studies on the potential health impacts of WtE-related emissions, even in countries where WtE facilities have been in operation for some years (such as Sweden); however, some practical implications can be drawn from the limited research done. This has implications for the emerging WtE sector.

As a consequence of the lack of health studies related to WtE facilities, inference is often drawn from exposure studies to health-related emissions common to combustion of MSW. These studies might provide some indication of potential impacts from WtE process emissions, albeit newer technologies and tighter restrictions of feedstock appear to be implemented in WtE facilities. An example of this is exposure to dioxin emissions from older MSW incinerators, where past epidemiological studies have reported weak to moderate associations between dioxin emissions and an increased incidence of cancers including non-Hodgkin's lymphoma (Viel *et al* 2008) and sarcoma (Zambon *et al* 2007) among nearby residents and incinerator workers. These studies were conducted prior to the lowering of incinerator emission volumes through introduction of stricter regulations, and so the findings cannot be directly extrapolated to current WtE technologies. Furthermore, it should be acknowledged that due to the varying waste streams in different geographic regions and for different facilities, research evidence from one country may not accurately or wholly inform policy or practice in other countries/regions. Older studies have also tended to study the incineration of unsorted MSW.

The extent to which the existing evidence base reviewed here can support a causal association between exposures to airborne emissions from WtE facilities and adverse health impacts, is very limited. While the evidence base, as a whole, is weak and there is little evidence of effects under normal operating conditions of WtE plants, the review has highlighted some potential areas for further study. There is clearly a place for more studies of the potential for health impact from WtE facilities, using the various study types included in this review: epidemiological studies; HRAs; LCAs; and, environmental monitoring. However, given the cost of completing well designed and adequately powered epidemiological studies, and the difficulty in ensuring sufficient sample size or a non-exposed control group, it is likely that other methods such as health risk assessment, along with exposure modelling, with or without LCAs, will prove to be useful in assessing new WtE facilities.

Notwithstanding the above, there is a need for well-designed epidemiological studies of exposure to WtE emissions. Such studies could provide empirical data for subsequent HRAs and LCAs, but need to address issues of exposure misclassification potential which has occurred in the past (Forastiere *et al* 2011) such as using distance based measures as an exposure proxy. The collection of environmental monitoring data of environmental media, e.g. air and soil, in the vicinity of WtE facilities, along with emissions monitoring, would also facilitate validation of the exposure models used in epidemiological and HRA studies.

There is an argument to be made for more standardised exposure assessment methods and standardised measurement units, reference criteria and models, in studying the health impacts of WtE emissions, especially for more harmful components such as dioxins, given the variety of methods presented in the LCAs and HRAs reviewed for this paper. Further, we agree with previous researchers that modelling studies, such as HRAs and LCAs, should explicitly outline model input assumptions and associated uncertainties given their influence on model outcomes (Scipioni *et al* 2009).

Some of the studies included in this review have highlighted the need for special consideration of the feedstock used for RDF and WtE facilities, given that it is one of the critical issues affecting contaminant emissions, over and above the treatment technology used. For instance, the World Energy Council (2016) stated that dioxin (and other toxins such as furan) emissions from RDF can be reduced by nearly 100% with the implementation of regulatory emission-control strategies within the WtE sector, such as controlling the nature of the feedstock. This can result in emission volumes which are lower, per equivalent energy unit, than for coal or gas-powered power plants (World Energy Council 2016). Regulating the pre-sorting of waste for WtE processes can help to maximise complete combustion and minimise carcinogenic emissions (Reza *et al* 2013, Karunathilake *et al* 2016). Others state that food waste, for example, should be removed from the RDF stream as it yields little exportable energy in WtE processes (Diggelman and Ham 2003). Some researchers state that the higher calorific value of RDF results in more

**Table 4.** Life cycle analyses of waste-to-energy/RDF processes.

| Author, Year, Country | Methods (considerations) | Environmental outcome/s | Health outcome/s | Results | Strengths/Limitations[a] | Overall quality assessment[b] |
|---|---|---|---|---|---|---|
| Karunathilake *et al* 2016 Canada | LCA and HRA LCA based on 1 tonne MSW or RDF feedstock; used ReCiPe Endpoint impact assessment method (System boundaries: MSW collection to treatment) | SimaPro software used. Health impacts estimated per 1 GWh annual energy generation for MSW & RDF incineration technologies | Average daily dose (ADD); Hazard Quotient (HQ); Incremental Cancer Risk (Rc); Non-cancer risks. Exposure pathways limited to inhalation. Weight of evidence values from IARC & USEPA for screening cancer risk. IRIS USEPA database used for potency factors; toxicity data from CalEPA | HQ and Rc significantly lower for using RDF in WtE versus incineration ($5.94 \times 10^{-10}$ vs $1.65^{-9}$). Aggregated non-cancer HQ <1 for incineration ($4.52 \times 10^{-4}$) & WtE ($3.09 \times 10^{-5}$) for 1 GWh annual energy supply | Exposure: + (*Lifelong, 75 yrs.; limited to inhalation route of exposure; did not address risk from exposure to GHG*) Outcome: + (*non-cancer risks articulated*) Assumptions: ++ (*values fully referenced and justified and from established sources; input assumptions based on international published reference values of IARC, USEPA, etc*) Conflict: —(*unreported*) | Study quality: Moderate (*Conference paper with no indication of full-paper forthcoming*) Health concern: Low |
| Passarini *et al* 2014 Italy | LCA comparing impact of upgrades from an incinerator (mid-1970s) to WtE facility (1996–2011) (currently 150 000 t $yr^{-1}$; use of 6 scenarios to reflect the various upgrades. Adopted 1ton MSW as functional unit. Output/to (System boundaries.: MSW arrival to treatment) | *Midpoints:* respiratory inorganics & organics, ozone layer, climate change, acidification/eutrophication, ecotoxicity, minerals, fossil fuels. *Endpoints*: human health, ecosystem quality, resources. Used Ecoindicator99 impact assessment method 2.09. | Human health impact: Carcinogens, respiratory inorganics & organics. Ecoinvent database updated with existing emissions monitoring data for modelling scenarios. Used Pedigree Quality Matrix to test uncertainties for six main error sources: representativeness & independence of data; of temporal, geographical & technical corrections. | Improvement in human health indicator due to climate change improvements with facility upgrade (production of energy from WtE plant). Decrease in carcinogens and PM in emissions. No change in heavy metal (main carcinogenic) concentrations in fly & bottom ash. Decreased environmental impact with modelled energy recovery | Exposure: + (*emissions monitoring data updated standardised model*) Outcome: + (*Used LCA guidelines & ISO 14 040–14 944 [2006] framework*) Assumptions: —(*sensitivity analysis conducted using Monte Carlo analyses; results not reported*) Conflict: —(*unreported*) | Study quality: Low Health concern: Low |

**Table 4.** Continued.

| Author, Year, Country | Methods (considerations) | Environmental outcome/s | Health outcome/s | Results | Strengths/Limitations[a] | Overall quality assessment[b] |
|---|---|---|---|---|---|---|
| Reza *et al* 2013 Canada | LCA to determine potential environmental benefits & impacts of RDF production & use in cement kiln (req high temp incineration). Emission impacts estimated using GaBi LCA software. Cost-benefit analysis conducted. | Functional unit: production & use of 1ton RDF = 647–792 kg coal Global warming potential (GWP: $CO_2$ eq./t: $CO_2$ fossil, $CH_4$, $N_2O$) | Human toxicity (kg As-eq): carcinogen risk potential (CRP: As, Cd, Cr-Vl, Ni, benzo(a)pyrene, dioxin/-furan.) Other toxic emissions: e.g., Hg, Pb. | Reduced CRP by 3–4 fold (mg As-eq) for RDF versus coal use Reduced emissions of As-eq (~63 mg). Reduced concentration of heavy metals except Pb, where RDF increases emissions. Reduced (30% decrease in) PCDD (mcg) for RDF versus coal use | Exposure: + (standardised model; used software provided data or data from other plants, not supplemented by real monitoring data) Outcome: + (*Used LCA guidelines & ISO 14 040–14 944 [2006] framework*) Assumptions:—(*unclear whether sensitivity analysis conducted*) Conflict:—(*unreported*) | Study quality: Low-Moderate Health concern: Low |
| Scipioni *et al* 2009 Italy | LCA to analyse 3 systems: incineration (dry and wet flue gas cleaning) with and without WtE 1 tonne MSW inputs | Ecoindicator99 chosen for impact assessment SimaPro 7.1 Software (uses 6 different databases incl. Ecoinvent) | DALYs (due to carcinogens, respiratory organics & inorganics, climate change, radiation, ozone layer) | Reduced DALYs with dry versus wet MSW ($2.37 \times 10^{-05}$ vs 2.44E-05) For incineration alone, greatest human health impact due to climate change effects assoc with stack emissions (mainly $CO_2$); for WtE functioning, lower resp pollutants but greater potential exp to carcinogens, climate change impacts and radiation | Exposure: + (*used standard model (ISO 14 040–14 044 [2006] framework*) Outcome: + (*used Ecoinvent database; used Pedigree Quality Matrix and Monte Carlo quantitative analysis to test uncertainties*) Assumptions: + (*sensitivity analysis conducted using Monte Carlo analyses*) Conflict:—(*unreported*) | Study quality: Low-Moderate Health concern: Low |
| Tan and Khoo 2006 Singapore | LCA comparison of landfill, incineration, incl. for WtE, & recycling & composting (System boundaries: MSW generation to treatment) | SimaPro software used: three main damage functions (human health, ecosystem quality or ecotoxicity, resources). HAPs avoided | DALYs (avoided), but paper did not present this data or calculations | "Energy gained from incineration of waste materials is outweighed by the air pollution generated". Recycling & composting resulted in least impact. | Exposure:—(*not stated for human health*) Outcome:—(*paper referred to health outcomes DALYs, but did not present clear data on DALYs or health risk*) Assumptions:—(*not well articulated*) Conflict:—(unreported) | Study quality: *Weak* Health concern: Undetermined |

[a]Exposure assessment; Study sample size; Appropriate assumptions; Conflict of interest statement
[b]Low, moderate or high potential for concern based on quality and outcomes of study

complete (higher temperature) combustion, resulting in less emissions of other potentially toxic pollutants such as volatile organic compounds (Friege and Fendel 2011).

Although not strictly airborne emissions, there is a need for increased scrutiny of the use/disposal of bottom and fly ash given at least two studies estimated increased concentrations of chromium and dioxin in bottom/fly ash. Others have advocated that WtE residuals should be re-purposed (isolated) as construction 'filler' rather than go to landfill (Tan and Khoo 2006, Passarini *et al* 2014, Malakahmad *et al* 2017). While toxins such as dioxins and furans found in breast milk, or heavy metals found in urine, are not themselves measured health impacts, they could cause health impacts with accumulation over time. The WHO recognises that, due to the omnipresence of dioxins, the whole population has background exposure which is not expected to affect human health (such as the levels found in our included studies); however, as these toxins have a high toxicity, efforts need to be taken to reduce additional exposure such as from waste incineration (WHO 2020). As such, we can best prevent or reduce this exposure by continuing to measure directly at the source.

More broadly, LCAs including health impacts of MSW stream management should not only consider direct pollutant emissions, but also the potential effects of repurposing waste such as reducing and recycling, and the impact on greenhouse gas production and transport emissions (Giusti 2009). A fair and full LCA may show that the most benefit from RDF/WtE processes may come from fuel substitution for industrial processes such as cement manufacturing (Reza *et al* 2013, Richards and Agranovski 2017), within which combustion ash could be isolated to further reduce the environmental impact from landfill. While WtE may have a larger carbon footprint ($CO_2$ emissions) compared with recycling of materials (e.g. plastic) (Tan and Khoo 2006), it generally emits lower concentrations of greenhouse gases ($CO_2$, methane) than landfill (Giusti 2009, Clean Energy Finance Corporation 2015, Malakahmad *et al* 2017, Beyene *et al* 2018, Murray 2018, Orru *et al* 2019). Furthermore, WtE technology (e.g. dry treatment of flue gas) has the ability to offset traditional (fossil fuel) combustion for electricity generation and thereby potentially reduce total emissions of greenhouse gases or criteria air pollutants (Scipioni *et al* 2009). These are all important considerations from a broader public health perspective.

LCA methodology appears to be well suited to provide useful information for the planning and design stage of waste management facilities, as they allow identification of alternative processes and treatment requirements, and so can enable decisions on long term infrastructure investments which benefit health not only locally, and more broadly. We recommend that in regions where WtE has not yet been fully adopted, that LCA incorporating HRA, should be undertaken using local data inputs and with local conditions in mind.

As many regions of the world are needing to manage unprecedented volumes of waste, and at the same time are also experiencing slow implementation of cleaner/safer technologies, the risks related to waste management are likely to remain a challenge for years to come. Using RDF for WtE may address a gap in the circular economy for recovering energy from waste, and while seen as a renewable resource (Natural Resources Canada 2015), decision-makers should appropriately assess applications for new WtE facilities, taking a precautionary but not inhibitory approach, in light of the lack of rigorous health evidence.

### 3.8. Conclusion

We have found a dearth of well-conducted epidemiological studies investigating the health risks of exposure from WtE processes. The limited evidence from the two epidemiological studies, along with HRAs, LCAs and emissions monitoring studies suggests that the risks to human health from emissions of appropriately designed, properly managed (including feedstock), state-of-the-art WtE incineration plants are relatively lower compared to prevailing alternative waste management practices, including incineration of unsorted waste (without energy recovery) and land fill. Importantly, the waste management hierarchy recommends an emphasis on the reduction of material going to waste before it is re-purposed or recycled, as it is clear that the input waste stream can substantially influence pollutant emissions.

While WtE practice might be a reasonable option for mitigating waste management and energy security issues, its implementation requires proper design, operation, and emissions management (monitoring) and control, as well as ongoing environmental and health monitoring and surveillance to maximise both economic and environmental benefits while minimising health impacts or risks. With respect to planning and design of WtE facilities, it is important that health risk assessments supported by comprehensive exposure monitoring, and robust modelling (e.g. detailed emissions modelling plus atmospheric modelling and real population data) be conducted for proposed WtE facilities to ensure that protective measures are optimally designed and emissions criteria appropriately implemented. Furthermore, close attention to health data used and assumptions made for reference doses, exposure duration and frequency, and concentration-response functions, is needed. It is equally important for HRAs and LCAs to include sensitivity analyses to test such assumptions. Future reviews will be reliant on additional well conducted epidemiological studies or HRAs and LCAs and, exposure modelling and monitoring, to further our knowledge in this area.

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## Data availability statement

No new data were created or analysed in this study.

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