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MODEL DEVELOPMENT FOR OPTIMIZING EMISSIONS AND CARBON CREDIT FROM INTEGRATED WASTE AND WASTEWATER MANAGEMENT

by
AMANI HABIB MAALOUF

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AMANI HABIB MAALOUF

Approved by:

Dr. Mutasem El-Fadel, Professor — Advisor
Civil and Environmental Engineering, AUB

Dr. Salah Sadek, Professor — Committee Chair
Civil and Environmental Engineering, AUB

Dr. May Massoud, Associate Professor — Member of Committee
Environmental Health, AUB

Dr. Agamuthu Periathamby, Professor — External Examiner
Institute of Biological Sciences, University of Malaya, Malaysia

Dr. Francesco Di Maria, Professor — External Examiner
Department of Engineering, University of Perugia, Italy

Dr. Pascal Saikaly, Professor — External Examiner
Biological and Environmental Science and Engineering, KAUST, KSA

Date of dissertation defense: May 2, 2019

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PREFACE

The work presented in this PhD thesis was conducted at the Department of Civil and Environmental Engineering of the American University of Beirut (AUB) under the supervision of Professor Mutasem El-Fadel. This study was supported through a joint funding from the Lebanese National Council for Scientific Research (NCSR-L) through the CNRS-L/AUB Doctoral Scholarship Award and the American University of Beirut.

The PhD thesis is based on several publications listed below and referred to in the thesis as needed. Other relevant publications produced during the residency of the candidate are listed below as well.

Papers in Refereed International Journal

1. Maalouf, A., El-Fadel, M. (2017). Effect of a food waste disposer policy on solid waste and wastewater management with economic implications of environmental externalities. *Waste Management*, 69, 455-462. doi: https://doi.org/10.1016/j.wasman.2017.08.008
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AN ABSTRACT OF THE DISSERTATION OF

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This study examines the variability in estimating aggregated and disaggregated emissions from the solid waste sector using worldwide adopted methods for country accounting, life cycle assessment modelling, and corporate reporting. Disaggregation of emissions was conducted by source (waste management process from collection to disposal), gas (CO_2, CH_4, and N_2O) or type (direct and indirect) to identify processes contributing most to the potential variability in estimated emissions. While similar operational data were introduced in all methods, significant variability in estimated emissions were evident across methods. The variability in aggregated emissions ranged from 3 to 65% that dropped to 2 and 17% when default parameters were standardized across methods. At the disaggregated level, a wider variability was discerned reaching several folds depending on the source, gas or type of emissions. The observed variability can be attributed to differences between methods in approaches and default parameters. These differences can affect emissions mitigation measures / reduction targets or influence investments in carbon credit to meet countries' Nationally Determined Contributions under the Paris Agreement.

This research presents a novel comprehensive model developed to assess the carbon footprint of integrated solid waste management systems including the diversion at source of the food waste component into the wastewater/sludge management systems using household food waste disposers. In addition to the current state of practice in developed economies, the model includes emissions from waste management processes still practiced in developing economies (such as open dumping, open burning, poorly operated landfills with flaring systems and auxiliary fuel needed to satisfy the low heating value (LHV) during incineration) commonly not considered in most life cycle assessment (LCA)-based models. It can disaggregate emissions by source (from collection to final disposal), or type (direct-operating, indirect-upstream, indirect-downstream), or gas (CH_4, CO_2, N_2O) and offers users the flexibility to select processes or modify input parameters while examining their impact on uncertainty in model simulations. Equally important is a clarity in deriving and applying emission factors used to quantify emissions from waste management systems.

The model was tested in the context of developed and developing economies to assess the impact of waste composition, management processes, energy consumption and other parameters on variations in emissions. The results demonstrated that best practices through material recycling, biological treatment, food waste diversion, and/or energy recovery can contribute to significant savings in emissions that ranged between 24 and 95%, depending on the tested systems. The carbon footprint of introducing a food waste disposer (FWD) policy was examined in the context of its implications on solid waste and

wastewater management with economic assessment of environmental externalities emphasizing potential carbon credit and increased sludge generation. For this purpose, the model was asdoped to test scenarios for a waste with high organic food content typical of developing economies. For such a waste composition, the results show that a FWD policy can reduce emissions by nearly ~42% depending on market penetration, fraction of food waste ground, as well as solid waste and wastewater management schemes, including potential energy recovery. In comparison to baseline, equivalent economic gains can reach ~28% when environmental externalities including sludge management and emissions variations are considered. The sensitivity analyses on processes with a wide range in costs showed an equivalent economic impact thus emphasizing the viability of a FWD policy although the variation in the cost of sludge management exhibited a significant impact on savings.

In closure, this study argues the benefits of the model application in providing guidelines for policy planning and decision making about process viability for investing in carbon credit.

Keywords: Solid waste management, Food waste disposer, Wastewater and sludge management, Emission accounting, Life cycle assessment, Carbon footprint, Carbon credit.

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ABBREVIATIONS

CCX	Chicago Climate Exchange
CDM	Clean development mechanism
CER	Certified emission reductions
CH_4	Methane
CO_2	Carbon dioxide
EFs	Emission factors
EPA	U.S. Environmental Protection Agency
ETS	Emission Trading Scheme
EU	European Union
FWD	Food waste disposer
GHG	Greenhouse gas
GUI	Graphical User Interface
GWFs	Global warming factors
GWP	Global warming potential
IPCC	Intergovernmental panel on climate change
ISO	International Organization for Standardization
LCA	Life cycle assessment
LCC	Life cycle cost
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LFG	Landfill gas
MSW	Municipal solid waste
$MTCO_2E$	Metric Tons of CO_2 equivalent
N_2O	Nitrous oxide
NDCs	Nationally Determined Contributions
S	Sludge
UNFCCC	United Nations framework convention on climate change
WBCSD	World Business Council for Sustainable Development
WRI	World Resources Institute
WW	Wastewater

This thesis is dedicated to my family for their love and support

CHAPTER 1

INTRODUCTION

1.1 Background

The waste sector contributes to greenhouse gas (GHG) emissions primarily in the form of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), and a few other gases with less significant quantities. These gases are emitted through various processes and components of the waste management cycle (from collection to material recovery, biological and thermal processes, and landfilling) and accounted for ~3% ($1446x10^6$ $MTCO_2E$) of worldwide GHG emissions in 2010 (Blanco et al., 2014). While relatively a smaller contributor to total GHG emissions, the waste sector is considered to present an appreciable potential towards emissions' reduction through selected technologies (Bogner et al., 2007; IFEU and Ökoinstitut, 2010) particularly in developing economies where emissions from waste can account for a larger percentage reaching 15% of total country emissions (Friedrich and Trois, 2011; IFEU and Ökoinstitut, 2010).

Several models have been developed to estimate emissions from the waste sector and assess environmental burdens associated with waste management processes (EPA/ICF, 2016; Fernández-Nava et al., 2014; Itoiz et al., 2013; Pires et al., 2011; El Hanandeh and El-Zein, 2010; Cherubini et al., 2008; 2010; EEA, 2003). In this context, emissions from waste management encompasses indirect upstream emissions arising from inputs of materials and energy (electricity & fuel), direct operational emissions from system operation such as onsite operating equipment and waste degradation, and indirect downstream emissions (or savings) related to energy generation, materials substitution, and carbon storage (Gentil et al., 2009).

Most studies assessing global warming factors (GWFs) for emission contribution associated with waste management have focused on individual processes with the majority of emissions' accounting models established with default input parameters that are not accessible or

adjustable (Assamoi and Lawryshyn, 2012; Laurent et al., 2014). Commonly used models also neglect certain upstream (fuel/energy and material provision) or downstream (avoided emissions from carbon storage and material recovery) processes. Equally important is the difficulty to disaggregate emissions using existing models based on scope of reporting whether for national inventorying (direct emissions) or planning and decision-making purposes (direct and indirect emissions). In addition, some models do not address emissions from certain waste management processes such as open burning or dumping and flaring of landfill gas (LFG). This is of particular importance in developing economies where a high fraction of waste is still burned or disposed of in open dumps or landfilled with an inefficient LFG collection system or flaring at best. On the other hand, while introducing a food waste disposer (FWD) policy to divert the organic fraction of food waste from the waste stream into the wastewater (WW) management system has proved to be an effective alternative for waste reduction (Marashlian and El-Fadel, 2005; Yi and Yoo, 2014; Bernstad et al., 2013; Bernstad Saraiva et al., 2016), none of the existing models was designed to assess its impact on emissions' inventory.

In this study, we attempt to address the limitations outlined above by developing a model that can assess the impact on emissions from municipal solid waste (MSW) management systems when coupled with WW and sludge (S) management through the introduction of a FWD. The model allows the disaggregation of emissions by source (from collection to final disposal), or type (direct and indirect), or main gases (CH_4, CO_2, N_2O) and offers the flexibility of allowing the user to select processes or modify input parameters. The model will be tested in the context of developed and developing economies to assess the impact of a FWD policy, waste composition, management processes, and input parameters for improved planning and decision making about process viability for investing in carbon credit.

1.2 Research Objectives

This research aims to evaluate the integrated waste management systems following a life cycle inventory approach to identify economically viable waste management (MSW, WW, and S)

alternatives with minimal environmental externalities including best strategies for emission reduction in the context of regions in developed and developing economies. More specifically, this research targets the following objectives:

Objective no.1: Compare existing methods commonly used in estimating emissions from waste management while assessing the implications of the differences in emissions estimation. The ultimate objective is to identify gaps in quantifying emissions particularly beyond the region for which existing models were developed.

Objective no.2: Develop a comprehensive model to assess emissions from MSW management as well as from WW and S management when a FWD is introduced to divert food waste from the MSW stream into the WW stream.

Objective no.3: Evaluate best strategies for emission reduction and assess how different waste management (MSW, WW, and S) systems can be combined and optimized for this purpose. This is coupled by defining economically attractive policies while taking into consideration the carbon credit of related policies in the context of regions with developed and developing economies.

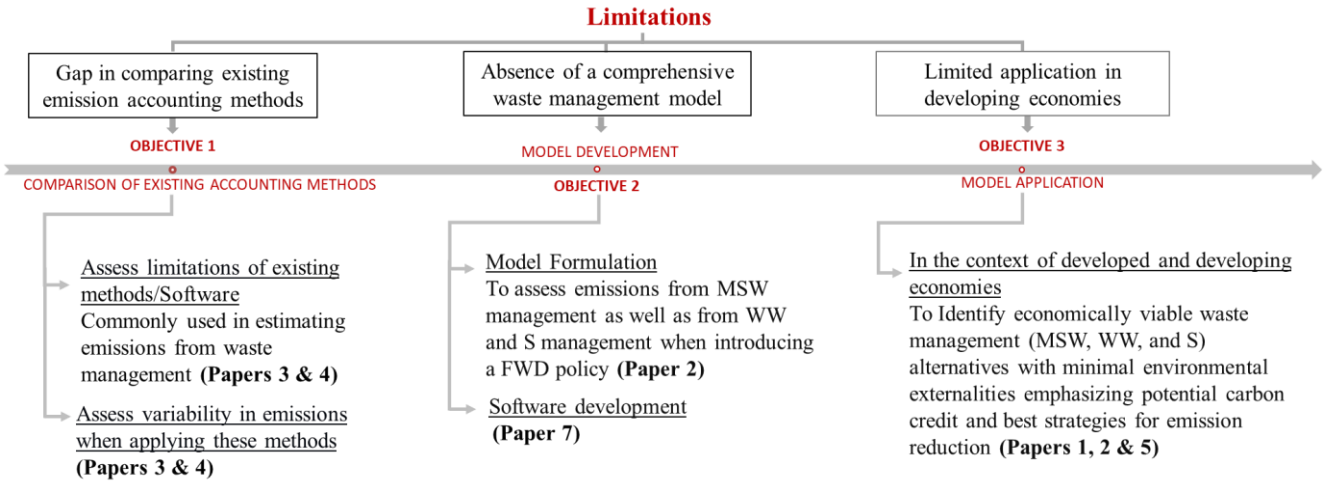


Figure 1.1. Objectives and outcomes

1.3 Research Innovation

This research is innovative in being the first to assess the impact on carbon emissions when combining the MSW and WW with S management systems upon introducing a FWD policy for grinding food waste. The model examines the MSW management system within a wide context

involving all components of MSW management from collection, recycling, composting, anaerobic digestion, incineration, to landfilling, and open dumping or open burning. It accounts for indirect upstream, direct operating, and indirect downstream emissions along with energy produced and consumed across all stages. The developed model includes several advantages with respect to existing models, by offering the flexibility of allowing the user to select processes or modify input parameters and disaggregates emissions by source (waste processes), or type (direct and indirect), or gas (CH_4, CO_2, N_2O). The developed model also offers an optimization tool to provide decision-makers with optimum integrated waste management systems for any region based on economic or environmental costs including carbon credits. Life Cycle Assessment (LCA) approach proved to be effective in identifying integrated waste management systems with minimal impacts, comparing different waste management alternatives and technologies. Decision-makers using only waste-LCA based models can only obtain the optimum integrated waste management policy by selecting the option with the least impact among predetermined policies. However, the optimum integrated waste management policy might be a combination that was not predetermined and thus was not assessed by the LCA model. Accordingly, the model presented in this study is innovative because it offers an optimization tool that considers a wide range of possible combinations whether predetermined or not in order to obtain an optimum integrated waste and wastewater management system while quantifying emissions based on LCA approach.

1.4 Dissertation structure

The dissertation consists of seven chapters that can be grouped into three parts besides the introduction (Chapter 1) and the conclusion (Chapter 7). The first part, consisting of Chapters 2 and 3, presents an assessment of existing methods commonly used in estimating emissions from waste management and examines the variability in aggregated and disaggregated emissions from waste management. The second part consists of Chapter 4 that presents how the new comprehensive model was developed to assess emissions from MSW management as well as from WW and S management when introducing a FWD policy and how it addresses limitations in existing emissions

accounting methods, which were defined in Chapters 2 and 3. This model was also developed using a MATLAB-based Graphical User Interface (GUI) that allows users to control the operations on the entire system, which is further elaborated in Appendix A. The third part is presented in Chapters 5 and 6, and it focuses on model application to identify economically viable waste management (MSW, WW, and S) alternatives with minimal environmental externalities emphasizing potential carbon credit and best strategies for emission reduction in the context of developed and developing economies. This part also presents the application of a life cycle assessment (LCA) approach adopted to identify integrated systems with minimal impacts and reduced emissions in a developing context coupled with an economic valuation and sensitivity analysis to assess the effect of varying influencing parameters individually. The overall outcome of the thesis and its findings are concluded in Chapter 7, where recommendations and some suggestions for further work are also found.

CHAPTER 2

TOWARDS IMPROVING EMISSIONS ACCOUNTTING METHODS IN WASTE MANAGEMENT: A PROPOSED FRAMEWORK

2.1 Introduction

Concerns about anthropogenic contributions to global warming from solid waste management have stimulated efforts aiming at quantifying and reducing emissions from the waste sector. This practice also referred to as emissions inventorying or accounting or carbon footprint, is dependent on waste treatment and management processes, the type of waste and corresponding physical composition, in addition to the accounting method (Chen and Lin, 2008). In this context, several methods that differ in data requirements and scope have been reported (Gentil et al., 2009) in examining emissions based on specific waste treatment and management processes: 1) the country level accounting with reference to the IPCC; 2) the organizational annual reporting on environmental issues and social responsibility used by corporates, facilities, or municipalities; 3) the LCA modelling as an environmental basis for evaluating waste management systems and technologies; and 4) the carbon trading methodology under the clean development mechanism (CDM). Friedrich and Trois (2011) expressed the need to assess the relationship between these methods and arising emissions from various processes. As such, comparing commonly used methods for estimating emissions from municipal solid waste (MSW) management attracted considerable attention as detailed below in the literature background Section 2.2 (Table 2.1). In short, these methods were applied theoretically or for specific case studies to relate their outcomes using default parameters that are invariably dependent on the location where a particular method was developed. In this context, uncertainties are reportedly inevitable when applying any particular method beyond its geographical boundaries (Maalouf and El-Fadel, 2018; Gentil et al., 2010; Friedrich and Trois, 2013; Laurent et al., 2014). This chapter examines the variability in predicting

emissions from MSW management associated with differences in underlying fundamentals and in default parameters including emission factors (EFs). The objective is to define how and what emissions accounting method to use for policy planning and to develop a conceptual framework model to address potential limitations in existing methods. This chapter compares common emission accounting methods (country level accounting with reference to the IPCC, LCA modelling, and organizational reporting) with a breakdown of emissions into direct operational, indirect upstream, and indirect downstream contributions related to waste management processes from collection to final disposal. We quantify the differences in accounting methods by source (i.e. waste management processes), type of emissions (i.e. direct or indirect), and gas (i.e. CO_2, CH_4, N_2O) while also considering the waste composition. This chapter provides insights about the variability in emissions associated with various methods and highlights related limitations when applied geographically beyond the context for which they were developed.

2.2 literature background

Accounting methods to estimate emissions from waste management have been classified under four main types namely: life cycle assessment (LCA), country accounting, corporate reporting, and carbon credit trading mechanisms (Gentil et al. 2009). The LCA approach is accepted internationally as a standardized method (ISO 2006a, 2006b) to identify, assess, and compare the environmental burdens associated with waste management (Nabavi-Pelesaraei et al., 2017) with many applications in the context of greenhouse gas (GHG) emissions (Table 2.1) in various countries. The accuracy of LCA tools is strongly dependent on the ability of modeling local conditions and the use of site-specific input data (Ripa et al. 2017). As such, in many countries, the Intergovernmental Panel on Climate Change (IPCC) guidelines (IPCC, 1996; 2006) are still used for national communications under the United Nations Framework Convention on Climate Change (UNFCCC) due primarily to the lack of data required under an LCA approach (Gentil et al., 2009). These guidelines account for direct emissions from the waste sector without consideration to potential inter-linkages with other sectors. Similarly, several protocols and accounting methods

were developed based on voluntary industry-led approaches at the organization, facility, corporation, or a municipality level. Whether a mandatory or a voluntary initiative, it is seen as an important contributor to society by reducing GHG emissions from waste management activities. In this context, the Entreprises pour l'Environnemnent (EpE) protocol is widely accepted and was adapted to the waste management industry to account for direct and indirect emissions. As such selecting the proper waste management alternative and estimation method is directly associated with the assessment and mitigation of emissions. The latter is of particular significance in the context of GHG trading schemes that have evolved and reached an advanced stage of implementation. Trading schemes, whether voluntary or regulatory based, have indeed recognized the potential of the waste sector for appreciable GHG mitigation. However, these schemes have exhibited wide discrepancies among them, which necessitate consistent accounting procedures to ensure accurate quantification of emissions (Gentil et al., 2009; ISWA, 2009). This can be of importance for country commitment to report regularly on emissions and implementation efforts through nationally determined contributions (NDCs) under the Paris Agreement (UNFCCC, 2015).

In summary, several studies compared these methods and identified differentiating factors such as system boundaries, waste composition, time horizon, energy modelling, and most importantly EFs. However, no study quantified the independent contribution of each factor to the variability in disaggregated emissions by type or source (Table 2.1). Hence, more efforts are needed in this context towards the development of a framework to address this gap, which is the ultimate objective of this chapter. The corresponding policy implications of differences in accounting methods can affect mitigation measures and reporting targets under the UNFCCC agreements or influence reduction targets using carbon credits to meet NDCs under the Paris Agreement.

Table 2.1. Past efforts at comparing accounting methods of emissions from waste management

Reference*	Description
Kulczycka et al. (2015)	Conducted a comparison of several impact categories using two life cycle assessment (LCA) models (generic and specific) applied on a single scenario.
Laurent et al. (2014)	Reviewed literature reported waste-related LCA models commonly used by practitioners.
Friedrich et al. (2013)	Provided a concise synthesis of existing tools, models, and publications deriving and using emission factors in the context of developed countries highlighting their implications when applied in the context of developing countries with the purpose of defining data and methods for a specific study area.
Itoiz et al. (2013)	Presented a technical and operational review of a proposed new tool and compared it with other European tools based on literature reported information.
Karmperis et al. (2013)	Reviewed decision support models that are commonly used in solid waste management while assessing their strengths and weaknesses.
Assamoi & Lawryshyn (2012)	Reviewed existing LCA models to extract data for a case study. Existing models were reported to provide no flexibility to incorporate changes in parameters.
Björklund et al. (2011)	Provided an overview of existing waste-LCA based models.
Eriksson et al. (2003; 2011)	Presented a theoretical comparison of two models to assess their effectiveness in decision-making.
Mohareb et al. (2011)	Compared four emissions estimation methods at a specific case study using default model parameters.
Pires et al. (2011)	Reviewed models illuminating overlapped boundaries in solid waste management (SWM) practices in EU.
Vergara et al. (2011)	Compared two waste-LCA models to assess their differences in emission estimation by considering default model parameters applied on a specific case study.
Cleary (2010)	Reviewed LCAs for SWM systems using 14 computer models emphasizing the need to identifying the scope and methodological assumptions of LCA towards reliable results.
Gentil et al. (2010)	Provided an overview of literature reported LCA models applied to SWM and compared them with respect to technical assumptions, methodologies, and input parameters.
Hanandeh & El-Zein (2010)	Compared simulations using default parameters at a specific case study to validate their developed model.
Del Borghi et al. (2009)	Reviewed existing SWM models and emphasized data constraints (e.g. time-related, geographical, and technological coverage).
Gentil et al. (2009)	Presented an overview analysis and comparison of four main types of emissions accounting methods in SWM. It highlighted the need to examine the relationship between them and SWM processes and technologies.
Rimaityté et al. (2007)	Compared incineration outputs of the LCA model with measured emissions data. Significant differences between simulated and measured data were reported.
Winkler & Bilitewski (2007)	Compared six waste-LCA models using the same waste management scenario and default models' parameters. Significant differences among models were highlighted reaching up to 1400% for some results.
Diaz and Warith (2006)	Model comparison was used in a case study to validate model results, which were then compared to simulations using existing models with their default parameters.
Morrissey & Browne (2004)	Provided a review of existing waste-models and highlighted corresponding shortcomings.
MacDonald (1996)	Provided a detailed review of existing solid waste management-models.

* In all studies, the contribution to differences in emissions were not reported and/or quantified independently for each influencing factor.

2.3 Methodology

2.3.1 Comparative assessment approach

Accounting methods for emissions from the waste sector that were tested and compared in this chapter encompassed the UN IPCC 2006 Guidelines, the US EPA WARM, the EU EpE protocols, the Canadian IWM, and the UK IWM-2 (Table 2.2). These methods were selected because they are publically accessible, widely reported in the literature, and adopted by cities or countries where they were originally developed (Itoiz et al., 2013; Mohareb et al., 2011; Gentil et al., 2010; Diaz and Warith, 2006). The IPCC guidelines in particular were supposedly put forth to standardize between methods at a global scale. Emissions arising from the waste management scheme involve indirect upstream emissions arising from inputs of energy (electricity & fuel) and materials, direct operational emissions from systems' operation including onsite operating equipment and waste processing, and indirect downstream emissions (or savings) related to energy generation, materials substitution, and carbon storage (Gentil et al., 2009). We emphasize that existing models used in the comparative assessment (Table 2.2) were selected based on their accessibility and common use worldwide. Other privately-owned models[1] may exist and offer additional features in the context of emissions accounting.

The comparative assessment was carried out under a two-phase approach (Figure 2.1). In the first phase, the difference in emissions were considered in the context of evaluation criteria (Table 2.3), which are reportedly of key relevance in emissions accounting from waste management (Gentil et al., 2009), particularly EFs. Additional testing was conducted to verify EFs. This phase entailed calculating the disaggregated and aggregated EFs to validate the variability in the observed

[1] Recent privately-owned models such as EaseTech, developed at the Technical University of Denmark (Clavreul et al., 2014) or the Solid Waste Optimization Life-cycle Framework (SWOLF) model (Levis et al., 2013) were not used in the comparative assessment because they have not been endorsed by governmental agencies for compliance purposes although they are useful models for waste management but not commonly reported for planning or decision making. In this study, the comparison targeted methods supported or endorsed by international or governmental organizations, particularly for compliance or GHG emissions reduction purposes.

emissions at various levels of waste management processes (collection to disposal). In this context, this phase involved checking whether the summation of individual EFs multiplied by MSW data characterizing the study area, provides approximately similar outcome as the aggregated EFs. Similarly, direct and indirect contributions were calculated in this additional testing to compare their equivalent disaggregated emissions using a unit category (1 tonne) of a single waste category (i.e. either food, or paper, or plastics, etc.) managed under a single process (collection to disposal). During the second phase, default parameters, particularly EFs, were standardized across methods to ensure a common basis for the comparison while running a single scenario. Following this phase, the methods were compared by source (management processes from collection to disposal) and type of emissions (direct or indirect) with concomitant consideration for waste composition.

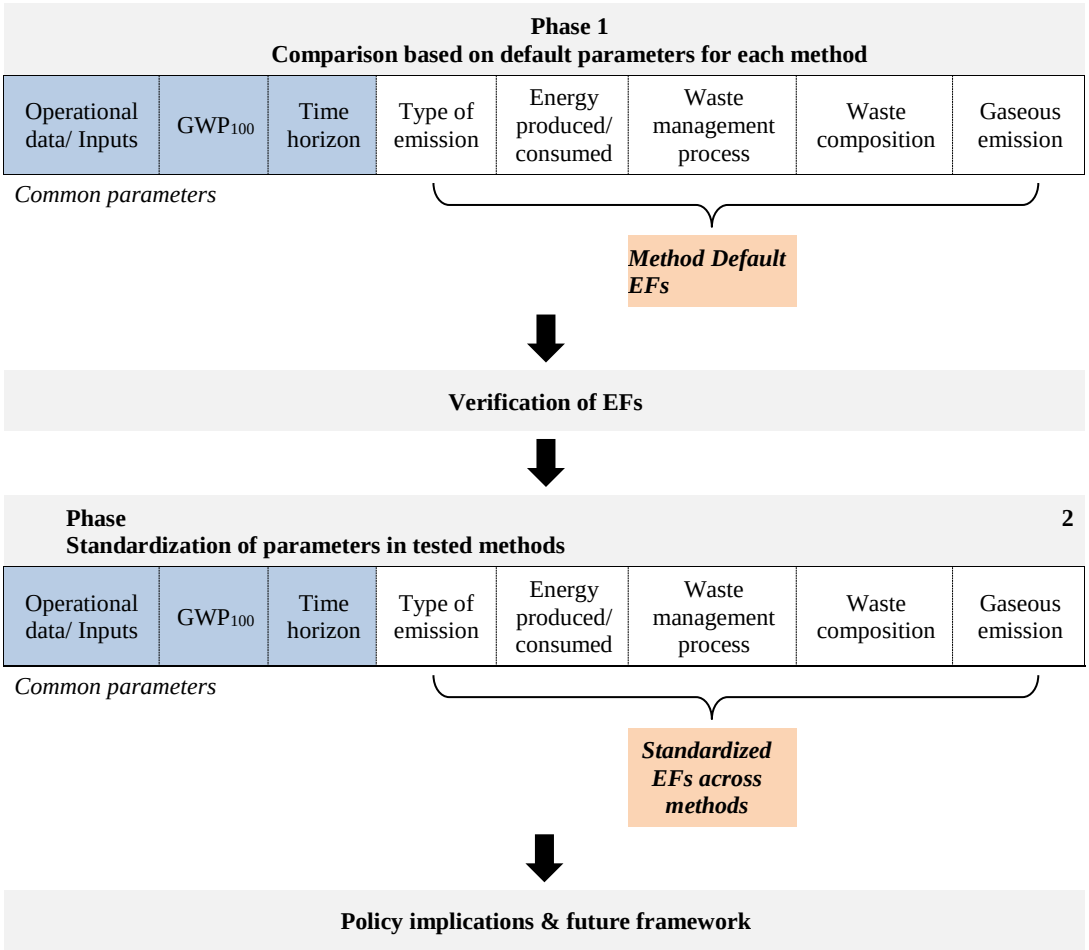


Figure 2.1. Comparative assessment approach
EFs: Emission factors

Table 2.2. Characteristics of tested emissions accounting methods

	IPCC 2006	EpE Protocol	WARM	IWM	IWM-2
Developed by	IPCC (2006)	EpE (2013)	US EPA/ICF (2012)	EPIC & CSR (2004)	McDougall et al. (2001)
Geographical scope	Worldwide	EU	US	Canada	UK
Intended use	National GHG reporting under the UNFCCC	Enterprise and local government accounting	Technical and environmental platform for decision making associated with municipal solid waste management alternatives		
Scope of accounting	Direct emissions	Life Cycle emissions	Direct & downstream emissions	Life Cycle emissions	Life Cycle emissions
Time consideration	10-50 years	1 Year	1 Year	1 Year	1 Year
GWP_{100} Reference	SAR (1995)	AR4 (2007)	AR4 (2007)	SAR (1995)	SAR (1995)
LF method	FOD	User selected	DM	DM	DM
Source/ sink	Yes/Yes	Yes/No	Yes/Yes	Yes/Yes	Yes/Yes
Management processes	Co, AD, I, Lf, OD	C, R, Co, AD, I, Lf	C, R, Co, I, Lf	C, R, Co, I, Lf	C, R, Co, AD, I, Lf
Waste categories	F, P,P_L, T, W, GA, N, O	Aggregated MSW	F,P,P_L,T, W, GA, G, M, O[f]	F, P, P_L, GA, G, M, O	F, P, P_L, T, G, M, O
Emissions	CO_2,CH_4,N_2O	CO_2,CH_4,N_2O	CO_2,CH_4,N_2O	Variable[a]	Variable[a]
Data requirement	High	High	Low	High	High
Modifiable/ dynamic	No	Yes	No	No	No
Data entry	Waste	Waste/fuel	Waste	Waste/fuel	Waste/fuel
Database/ EFs	Default/ User selected	User selected	Default	Default	Default

[a] Includes GHGs (greenhouse gases): CO_2, CH_4, N_2O emissions as well as other emissions such as CO, NOx, SOx, PM, HCl, HF, H_2S, Dioxins/Furans, NH_3, As, Cd, Cr, Cu, Lead, Mn, Hg, Ni, Zn.

IPCC: Intergovernmental Panel on Climate Change; **EpE:** Entreprises pour l'Environnemnent; **WARM:** Waste Reduction Model; **IWM:** Integrated Waste Management Model for Municipalities; **IWM-2:** Integrated Waste Management Model-2; **DM:** Default method (Theoretical yield gas); **FOD:** First order decay method; **LF Method:** method for accounting of methane gas emitted during landfilling; **LC (Life cycle) emissions**: include direct and indirect (upstream and downstream emissions); **EFs:** Emission factors; C: Collection; **R:** Recycling; **Co:** Composting; **AD:** Anaerobic Digestion; **I:** Incineration; **Lf:** Landfilling; **MSW**: Municipal solid waste; **F**: Food; **P**: Paper; **P_L**: Plastics; **T**: Textiles; **GA**: Garden; **W**: Wood; **N**: Nappies; **G**: Glass; **M**: Metals; **O**: others

Table 2.3. Elements of the comparative assessment of tested emissions accounting methods

Type of Criteria	Description	Example and Standardization
Scope of accounting	Accounting methods may vary between national GHG inventorying that consider direct emissions (IPCC), and LCA that accounts for both direct and indirect emissions.	Methods were compared by type of emissions: – Direct emissions from waste degradation or from systems' onsite operating equipment. – Upstream emissions from inputs of electricity, fuel, and material. – Indirect downstream emission savings related to energy-electricity generation, material substitution, or carbon storage.
Choice of system's boundary	Accounting methods may consider different waste management processes.	– Example of WARM that incorporates emissions from collection by default to EFs related to simulated processes (e.g. landfilling, composting, etc.), other methods include them under a separate category (e.g. collection). To ensure uniformity, such emissions were credited in all methods as an outcome from waste collection.
Time consideration	Accounting methods consider different reporting timeframe and GWP's time horizon.	– LCA-based methods consider methane emissions over a 100-year time horizon, while the IPCC-2006 adopts a first order decay (FOD). Accordingly, the IPCC-2006 was modified to incorporate a 100-year forecast of emissions. – All methods were set for a single time horizon of 100 years for consistency (GWP_{100}).
Interaction with energy systems	Energy system (consumed or produced) plays a role in the estimation of indirect emissions.	– The default electricity grid and its EF were adjusted for all methods to reflect the study area, which is $688x10^{-6}$ $MTCO_2E/kWh$ (IEA, 2014).
Default data / Other parameters	The methods incorporate default input parameters depending on the location where developed.	– Example about the fraction of landfill gas (LFG) collected: WARM considers a fraction of 0.6 of LFG collected (EPA /ICF, 2016), whereas the actual fraction is dependent on the study area and hence adjusted accordingly in all methods to 0.18 (MoE/UNDP/GEF, 2015).
Biogenic CO_2	The methods consider Biogenic CO_2 emissions with GWP of 0 differently.	– Some methods report them separately while others include them in the accounting of emissions such as IWM that considers biogenic CO_2 emissions during composting. In this study, biogenic CO_2 was excluded from the total emissions for all methods.
Global warming potential (GWP)	The GWP for 100 years' time horizon has evolved with time and the methods adopt by default different GWPs.	– Example of WARM uses IPCC (2007) resulting in 19% increase in GWP_{100} of CH_4, in comparison to IWM-2 (IPCC, 1995) thus the GWP was adjusted in all methods to follow the IPCC reference definition.
Choice of emissions	The methods can consider different gaseous emissions.	– EFs adopted by each accounting method were disaggregated by gaseous emissions (CO_2, CH_4, N_2O, etc. with corresponding GWP).
Waste type and composition	The methods can consider different waste type and composition.	– While some methods consider 7 types of waste categories, others like WARM can consider 45. Moreover, waste components can be managed differently by each method. In this study, the same waste composition was introduced in all methods.
Emission Factors (EFs)	The methods adopt different default EFs.	– EFs were disaggregated by type and source of emissions for each waste category including direct and indirect contributions. During the second phase of the comparative assessment, the same EFs were introduced in all methods.

2.3.2 Scenario definition and testing

The methods were tested at a pilot area (Beirut, Lebanon) for a comparative assessment of differences and suitability beyond the context in which they were developed. It is worth noting that globally, the contribution of landfilling to CH4 emissions is ~45% of total emissions from the waste sector (IPCC, 2014). In the pilot area, this contribution reached ~80% (MoE/UNDP/GEF, 2015) highlighting the relative importance of potential carbon credits from the sector at locations with similar characteristics.

The baseline conditions (S0) in the study area consists of commingled MSW collection,

sorting and recycling (7%), composting (10%), and landfilling (83%). Waste is collected daily by a fleet of 332 collection vehicles that consume an average volume of diesel equivalent to 6.2 L/Ton of waste generated (Laceco-Ramboll, 2012), which is within reported ranges (Larsen et al., 2009). The waste is then transferred into two material recovery facilities (MRFs) where it is sorted into bulky items, inerts, biodegradable organics, and recyclables. The biodegradable fraction is sent for windrow composting with relatively low-quality compost often rejected by consumers and hence mostly transferred along with other rejects to be used as intermediate cover at the landfill. The collection of landfill gas (LFG) for flaring was initiated partially 4 years after the site opening (at a measured 3 Gg/Year). The number of flares was increased over the lifespan of the landfill to reach 8 continuously operating flaring systems with varied capacities at a measured equivalent of 14 Gg of CH_4 recovered/year in 2013 for potential energy recovery (MoE/UNDP/GEF, 2015). Figure 2.2 displays the mass and energy sources for all baseline and alternative scenarios while Table 2.4 summarizes models' input parameters. The two additional scenarios that were considered:

Alternative Scenario 1 (S1): Collection / recycling / anaerobic digestion / landfilling. This scenario is similar to the baseline scenario S0, except for replacing the composting process with anaerobic digestion (10%) with energy recovery.

Alternative Scenario 2 (S2): Collection / recycling / composting / Incineration. This scenario considers incineration (83%) with energy recovery instead of landfilling in the baseline scenario S0. Note that emissions associated with the management of residues is not considered in all methods except WARM.

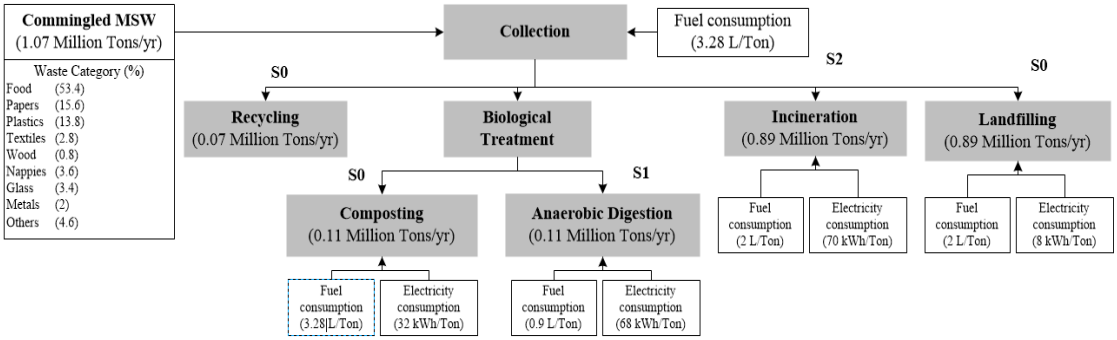


Figure 2.2. Baseline conditions and scenarios tested at study area
(Data extracted from Laceco-Ramboll, 2012; MoE/UNDP/GEF, 2015)

Table 2.4. General input parameters

Parameter	Adopted average value	Reference
Fuel consumption for on-site daily operation	~2 Liters/Ton of waste landfilled ~3.28 Liters/Ton of waste composted	1 to 3 Liters of diesel/ Ton of waste landfilled (Manfredi et al., 2009); 0.4 to 6 Liters of diesel/ Ton of waste composted (Boldrin et al., 2009; EPA, 2006; Smith et al., 2001), in most cases an average of 3 Liters/ Ton of waste composted is reported
Provision of electricity	8 kWh/Ton of waste landfilled and 32 kWh/Ton of waste composted	2 to 12 kWh/ Ton of waste landfilled (Manfredi et al., 2009); 8 or 32 kWh/ Ton of waste landfilled or composted (McDougall, 2001)
Fraction of LFG collected	0.18	0.18 at a measured equivalent 14 Gg of CH_4/year in 2013 (MoE/UNDP/GEF, 2015)

2.4 Results and discussion

2.4.1 Emissions variability

The results using all methods showed that landfilling was the largest contributor to total emissions followed by collection and composting, with recycling contributing to savings in total emissions (Figure 2.3). Considering each method at a time to be the base for the comparative assessment, the absolute variability in estimated emissions ranged from 3 to 65 % (Figure 2.4), reflecting the potential change in emissions' reporting using the different methods with their default parameters.

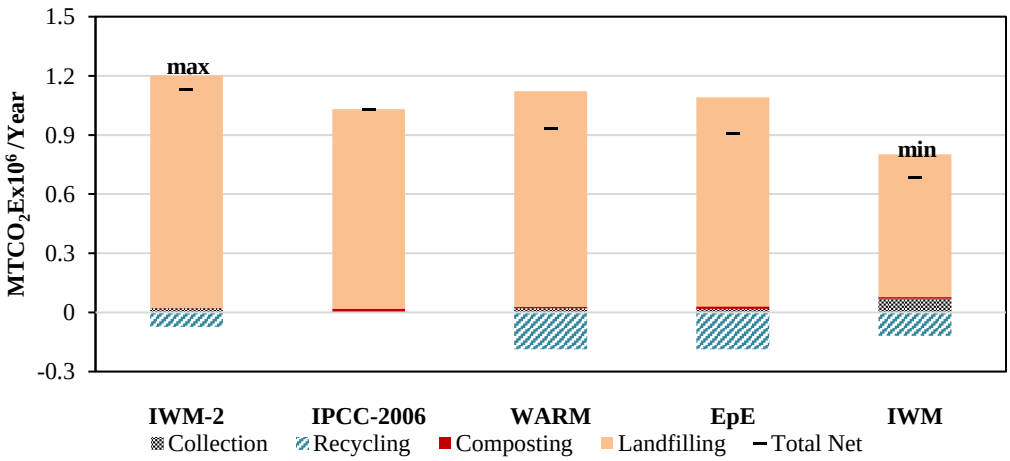
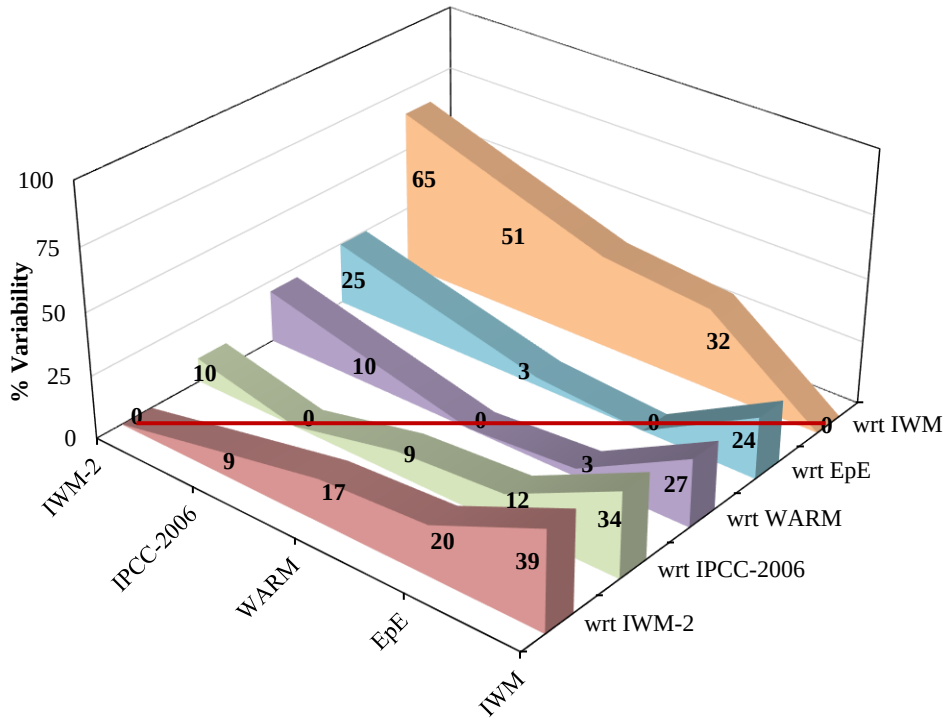


Figure 2.3. Emissions under baseline conditions (Scenario S0)



	IWM-2 (%)	IPCC-2006 (%)	WARM (%)	EpE (%)	IWM (%)
with respect to IWM-2	0	9	17	20	39
with respect to IPCC-2006	10	0	9	12	34
with respect to WARM	21	10	0	3	27
with respect to EpE	25	14	3	0	24
with respect to IWM	65	56	37	32	0

Figure 2.4. Absolute variability in emissions with non- standardized parameters when considering each method at a time to be the base for the comparative assessment
Variability % = | (Value of tested method$_{(i)}$ - Value of tested method$_{(j)}$) / Value of tested method$_{(i)}$| x100

The variability in emissions between methods is detailed in Table 2.5 by source (waste management process from collection to disposal) and type (direct or indirect) of emissions with values and absolute percent range of differences in comparison to each method. While all methods indicated that the direct emissions from waste degradation and fuel consumption by onsite operating equipment constitute the largest contributor (77 to 93%) to total emissions, a significant variability (3 to 87%) resulted from the usage of various methods (Table 2.5). Absolute indirect emissions from electricity provision (for composting and landfilling), fuel consumption (for collection or transport), as well as avoided emissions from material recovery (for recycling) accounted for 7 to 23% of total emissions with equally high variability between methods that ranged between 0.3 and 125% (Table 2.5).

Table 2.5. Emissions ($MTCO_2E \times 10^6$/Year) variability in comparison to each method disaggregated by source and type[a]

Emissions	Waste (Tons x 10^6)	IPCC-2006	EpE Protocol	IWM-2	WARM	IWM
Per Source Type						
Collection	1.069		0.018	0.021	0.020	0.070
Difference range %			16-297	14-241	14-241	71-75
Recycling	0.071		-0.187	-0.073	-0.187	-0.118
Difference range %			37-61	62-157	37-61	38-59
Composting	0.111	0.020	0.014	0.001	0.006	0.007
Difference range %		31-93	45-90	414-1283	18-218	15-169
Anaerobic digestion	0.111	0.023	0.005	0.04		
Difference range %		78-80	360-728	45-88		
Incineration	0.887	0.399	0.308	0.88	-0.01	-0.42
Difference range %[b]		4-131	16-199	55-67	14-158	4-121
Landfilling	0.887	1.011	1.060	1.179	1.094	0.724
Difference range %		5-28	3-32	7-39	3-34	40-63
Per Type of accounting						
Direct emissions		1.03	1.066	1.18	1.10	0.712
Difference range %		3-31	3-33	7-40	3-35	45-87
Landfilling		1.011	1.055	1.179	1.094	0.711
Difference range %		4-30	4-33	7-40	4-35	42-66
Composting		0.020	0.011	0.001	0.006	0.001
Difference range %		44-95	44-91	11-1643	80-218	12-1852
Indirect emissions[c]			-0.162	-0.052	-0.167	-0.029
Difference range %			2-56	120-125	2-55	0.3-55
Landfilling			0.005	0.001		0.013
Difference range %			89-164	838-2374		62-96
Composting			0.002	0.0003		0.006
Difference range %			88-158	731-2041		61-95
Total emissions						
S0[d]		**1.030**	**0.904**	**1.128**	**0.933**	**0.683**
Difference range %		**9-34**	**3-25**	**9-39**	**3-27**	**32-65**
S1[e]		1.034	0.892	1.120		
Difference range %		8-14	16-26	8-20		
S2[f]		0.443	0.148	0.830	-0.135	-0.569
Difference range %		67-228	191-484	40-169	210-715	76-246

[a] The absolute variability in emissions is calculated with respect to each method as follows:
Difference % = |(Value of tested $method_{(i)}$ - Value of tested $method_{(j)}$) / Value of tested $method_{(i)}$| x100

[b] Difference (%) in emissions is calculated based on total emissions excluding avoided emissions from energy recovery.

[c] Total indirect emissions include emissions (savings) from recycling; collection; as well as indirect upstream emissions from landfilling and composting (e.g. electricity and fuel provision)

[d] Scenario (S0): Baseline conditions

[e] Scenario (S1): composting of organic waste in S0 substituted by anaerobic digestion (AD) with energy recovery

[f] Scenario (S2): substituted waste landfilling in baseline scenario by incineration with energy recovery

More significant differences are discerned at the process level particularly composting, anaerobic digestion and incineration due to variations related mainly to default EFs. Process emissions were disaggregated by type of accounting[2] to shed light on differences in the way they

[2] Emissions are categorized by type of accounting as (1) direct (waste degradation and fuel combustion by onsite operating equipment); (2) indirect upstream (e.g. electricity provision); and (3) indirect downstream (or avoided) (energy and material recovery and carbon storage), depending on each waste management method (e.g. collection, recycling, composting, and landfilling).

are handled in each method (Figure 2.5). At the disaggregated level, a wider variability was discerned reaching several folds depending on the source, gas or type of emissions.

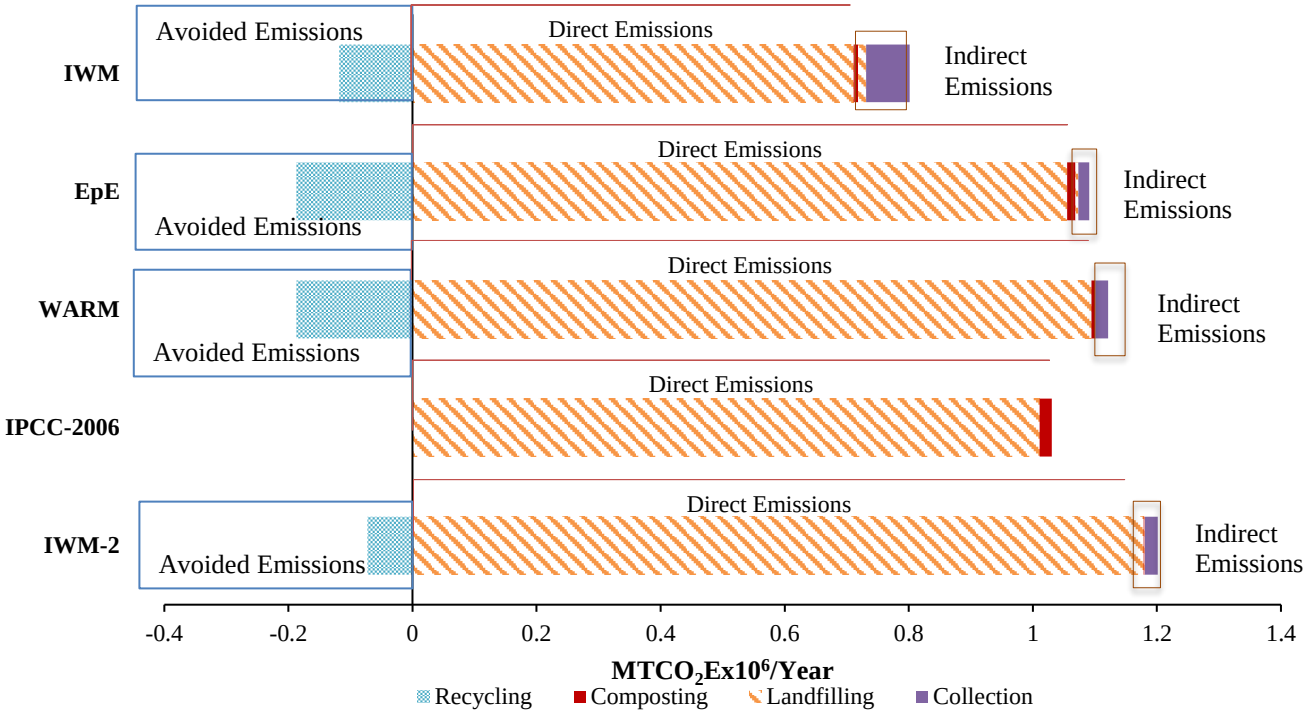


Figure 2.5. Emissions disaggregated by source and type of emissions for the accounting methods

Direct emissions: emissions from waste degradation/ processing and fuel combustion by onsite operating equipment;
Indirect emissions: upstream emissions from electricity provision;
Avoided emissions: indirect downstream emissions from material recovery

At the *collection and transport* level, IWM-2, WARM, IWM and the EpE protocol account only for direct emissions (during fuel combustion of operating equipment), which varied between methods from 14 to 241% (Table 2.5), with no consideration for upstream emissions (i.e. fuel provision for the extraction, processing, storage, and transport of fuel). While EpE, WARM, and IWM-2 resulted with comparable total aggregated emissions, IWM resulted in the highest emissions from collection because other methods adopt EFs with 70 to 74% lower values. IWM considers that the EF of N_2O (~0.007 $MTCO_2E$/Liter of Diesel) are higher than CO_2 (~0.003 $MTCO_2E$/Liter of Diesel) (

Table *3.3*), which is inconsistent with reported literature that recognizes CO_2 as the major contributor to emissions from fuel combustion during transportation, while N_2O accounts for 2 to 2.8% (Kahn et al., 2007). This explains the increase in emissions from collection and subsequently the high value of total indirect emissions (including savings from recycling) exhibited by IWM in comparison to other methods (0.3 to 55% higher) (Table 2.5).

Emissions savings from recycling consist of the difference in emissions associated with extracting and manufacturing of raw material versus remanufacturing of recyclables. The corresponding emissions exhibited differences between methods from 37 to 157% (Table 2.5). The EpE and WARM methods adopt similar EFs (EPA/ICF, 2012) and hence have identical savings Figure 2.5). In contrast, IWM exhibited lower savings in comparison to other methods. This can be due to its lower adopted absolute EF value of -0.83 $MTCO_2E$ per Ton of paper (Table 3.4), which still falls within the range reported in the literature (-4.4 to 1.5 $MTCO_2E$ per Ton of paper, Merrild et al., 2009), yet, it has a lower absolute saving value than WARM (-3.52 $MTCO_2E$ per tonne of paper). The deviations reflect also the significance of variations in the amount of material diverted to a specific process, which differ depending on the waste distribution adopted in each method. For instance, IWM-2 exhibited the lowest downstream savings from recycling (Figure 2.5) because by default, it diverts paper waste to composting. Moreover, losses of material during processing, which depend on the efficiency of the sorting process, differ considerably among methods, for instance IWM considers an efficiency of 95% vs. 88% in IWM-2, thus, reflecting differences in emissions.

Biologically, a wide variability in emissions is evident among methods ranging from15 to 1283% (Table 2.5). For all methods, direct emissions from waste degradation and fuel consumption by onsite operating equipment at the composting facility are higher than indirect emissions from electricity consumption (Figure 2.5). As a by-product, the compost would offset some CO_2 emissions from fertilizer and peat production or carbon storage from land application (Maraseni and Maroulis, 2008), which are accounted for in WARM only, although relatively insignificant.

However, WARM does not consider indirect upstream emissions from electricity and fuel provision (Figure 2.5). IWM and IWM-2 exhibited the lowest emissions (Figure 2.5) because they consider CH_4 and N_2O emissions from composting as negligible (Table 3.5) that contradicts the reported literature recognizing them as fugitive emissions produced during the decomposition process (Boldrin et al., 2009; EPA/ICF, 2016; IPCC, 2006).

Substituting composting by anaerobic digestion (AD) with energy recovery in scenario S1 decreased emissions in comparison to the baseline scenario (Table 2.5). IWM-2 exhibited higher emissions than the IPCC-2006 and EpE methods while other methods do not consider emissions from AD. IWM-2 considers that the produced biogas (containing CH_4 and CO_2) also forms CO_2 when CH_4 is burned (McDougall et al., 2001). This produces an equivalent EF of 0.440 $MTCO_2E$ per Ton of wet organic material in comparison to 0.009 $MTCO_2E$ per Ton of wet organic material in EpE, which is inconsistent with the reported literature (Boldrin et al. 2011; Møller, et al. 2009; EPA/ICF, 2016). The other two methods consider fugitive CH_4 emissions due to unintentional leakages (0-10%) during the AD process and CO_2 emissions as biogenic. Also, IWM-2 includes savings from energy recovery whereas the IPCC-2006 guidelines do not account for such savings under the waste sector, thus emphasizing the interdependence of emissions and the interaction with energy systems that is invariably neglected.

The variability across methods in emissions from *landfilling* ranged from 3 to 63% (Table 2.5). Direct emissions consist of 1) emissions from fuel combustion of onsite operating equipment that are similar in all methods, and 2) emissions from waste degradation processes that differed across methods. While similar operational data are introduced in all methods, the choice of waste composition with corresponding EFs, is different between methods. For instance, the IPCC-2006 considers emissions from certain types of landfilled degradable MSW (e.g. organic, paper, wood, textiles, and nappies) and resulted with 40% higher emissions than IWM (Table 2.5). IWM considers EFs from landfilled paper and food waste only (Table 3.6). Accordingly, IWM resulted

with the lowest emissions amongst the tested methods, with a variability of 40 to 63% (Table 2.5) with respect to other methods.

Avoided emissions from landfilling include savings from energy recovery that are generally considered by all methods (except IPCC), and savings from carbon storage (considered only in WARM). For the case of the pilot test area, additional savings from energy recovery might not be significant (up to -4%) due to the low efficiency of collected LFG. However, savings from carbon storage, is critical to consider in emissions accounting (Manfredi et al., 2009; Christensen et al., 2009) because it can reportedly cause a significant difference in emissions reaching up to 49% at times (Friedrich and Trois, 2013).

The comparison has also identified a limitation among all methods (except for IWM), which do not account for N_2O emissions from flaring of LFG. In addition, most methods adopt an average of 0.6 for LFG collected (WARM, EPA /ICF, 2016), whereas the actual fraction can be site-dependent as is the case in the study area with a 0.18 value (MoE/UNDP/GEF, 2015). It is noteworthy that none of the tested methods, including LCA-based methods (IWM, IWM-2, and WARM), consider a complete cycle from construction to final closure of a landfill. They tend to rely on databases for large direct emissions from waste, particularly landfill methane emissions without field-validation or consideration to other drivers such as soil cover material, surface oxidation, or gaseous transport (Spokas et al., 2015). These drivers have serious implications for developing a more realistic and science-based landfill inventory.

Substituting landfilling by *incineration* coupled with energy recovery in scenario S2 resulted in a significant variation in emissions (Table 2.5). This can be attributed to different assumptions adopted in various methods such as the choice of EFs for energy produced or consumed; type of energy sources substituted; energy efficiency; and energy content of waste categories (Table 3.7). This emphasize the interdependence of emissions from waste management systems with energy systems. However, none of the methods accounts for indirect emissions

associated with the management of solid residues from waste incineration (e.g. savings from slag recovery and load from bottom ash landfilling) except WARM that considers avoided CO_2 emissions due to recycling of metals recovered from bottom ash.

Earlier assessment of several waste-LCA models highlighted the significant differences among models that reached up to several folds for some scenarios (Winkler and Bilitewski 2007). Building up on previous literature findings (Table 2.1), the above analysis quantified the independent contribution of each factor to the variability in disaggregated emissions by type, gas, and source of emissions. Moreover, it also confirmed that the choice of certain parameters particularly EFs can cause significant differences in emissions accounting emphasizing the need to ensure clarity and flexibility regarding these parameters.

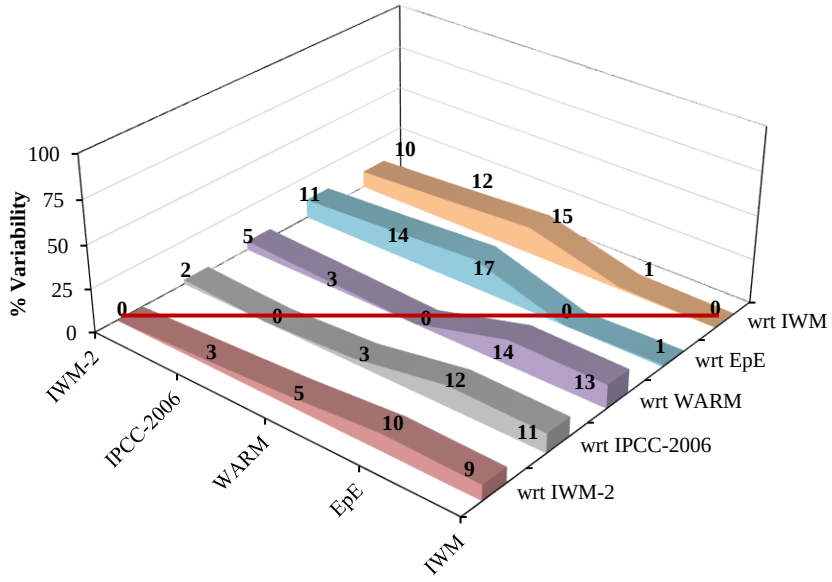
2.4.2 Verification of emission factors

A cross checking step was implemented to verify EFs and testing results. This phase entailed calculating the disaggregated and aggregated EFs to validate the variability in the observed emissions at various levels of waste management processes (collection to disposal). Aggregated EFs ($MTCO_2E$/ ton of waste) are the cumulative indirect-upstream, direct-operational, and indirect-downstream emissions from treating one ton of waste by individual waste management processes. Disaggregated EFs are expressed in metric tons of CO_2 equivalents ($MTCO_2E$) per characteristic unit (e.g. ton of waste treated; kWh of electricity; liter of diesel fuel). These EFs are separated by waste category, gas, waste process, and type of emissions (direct or indirect). A further illustration of the EFs (disaggregated and aggregated) adopted in each method is displayed (see Appendix B Table B.1) with corresponding flow diagrams of management systems (see Figures Figure *3.1* Figure 3.5) that display the energy sources and resulting emissions for each method. The cross checking ascertained the proper application of the tested methods and provided a verification of EFs used within all methods. For example, the disaggregated EFs for composting of food waste using WARM consist of EF related to fuel consumption for the operation of equipment (0.003

$MTCO_2E$/liter of diesel fuel); EF of CH_4 emitted during waste degradation (0.005 $MTCO_2E$/ton of food waste); EF of N_2O emitted during waste degradation (0.041 $MTCO_2E$/ton of food waste); EF related to carbon storage from the application of compost on land (-0.24 $MTCO_2E$/ton of food waste). The summation of individual EFs multiplied by MSW data characterizing the study area (Figure 2.4), provides similar outcome as the aggregated EFs of -0.184 $MTCO_2E$/ton of food waste composted. Moreover, the EFs proved to be the cause of the variability in the overall emissions exhibited by the methods for the same study area and management processes (collection, recycling, composting, anaerobic digestion, incineration, or landfilling); waste category; corresponding mass input; GWP; and similar type of emissions (direct or indirect). In this context, it is imperative while using these methods, to provide a greater clarity in reported emissions, by providing details on related calculations and aggregated EFs particularly in the context of carbon trading.

2.4.3 Standardization of parameters

The above analysis quantified the range of variability in emissions between the various methods while the second phase of the comparative assessment considered the standardization of all methods. Accordingly, similar operational data and default parameters, particularly EFs were introduced in all methods. EFs were adopted from WARM for all methods and tested for the baseline scenario. EFs from the WARM model were selected because it follows a life cycle inventory approach that includes all direct and indirect processes and accounts for various waste composition. In addition, WARM is the most updated (in terms of energy and emission factors used) among the various methods with the last version 15 released in 2016 (EPA/ICF, 2016) including results from laboratory and field testing. The resulting absolute variability between methods in estimated emissions dropped to 2-17% (Figure 2.6).



	IWM-2 (%)	IPCC-2006 (%)	WARM (%)	EpE (%)	IWM (%)
with respect to IWM-2	0	3	5	10	9
with respect to IPCC-2006	2	0	3	12	11
with respect to WARM	5	3	0	14	13
with respect to EpE	11	14	17	0	1
with respect to IWM	10	12	15	1	0

Figure 2.6. Absolute variability in emissions with standardized parameters when considering each method at a time to be the base for the comparative assessment

Variability % = $|$(Value of tested method$_{(i)}$ - Value of tested method$_{(j)}$) / Value of tested method$_{(i)}$$|$ x100

A disaggregation of the absolute variability in emissions by source (collection to landfilling) is displayed in Figure 7 to further delineate the difference with respect to each method. While all methods resulted in similar emissions at the waste collection level (Figure 2.7), after standardization the difference in emissions remained evident at various waste management processes. This can be attributed to default assumptions; the choice of gases (CO_2, CH_4, and N_2O); the choice of waste composition; as well as embedded system boundary conditions whereby, certain methods neglect upstream contributions. For instance, the variability in emissions from composting was the highest with respect to IWM (15-107%) and IWM-2 (118-144%) because both methods do not account for CH_4 and N_2O emissions from waste degradation during composting thus resulting in lower emissions. For recycling, all methods (except IWM-2) resulted in nearly similar emissions, which are higher (107-172%) than IWM-2 (Figure 2.7) because the latter diverts paper waste into composting by default. IWM and IWM-2 account for emissions from paper and food waste during landfilling by default, which resulted with comparable emissions to IPCC-2006 and EpE that

consider emissions from food, paper and wood wastes during landfilling whereas WARM accounts for various waste components (paper, food, wood, and mixed waste, etc.).

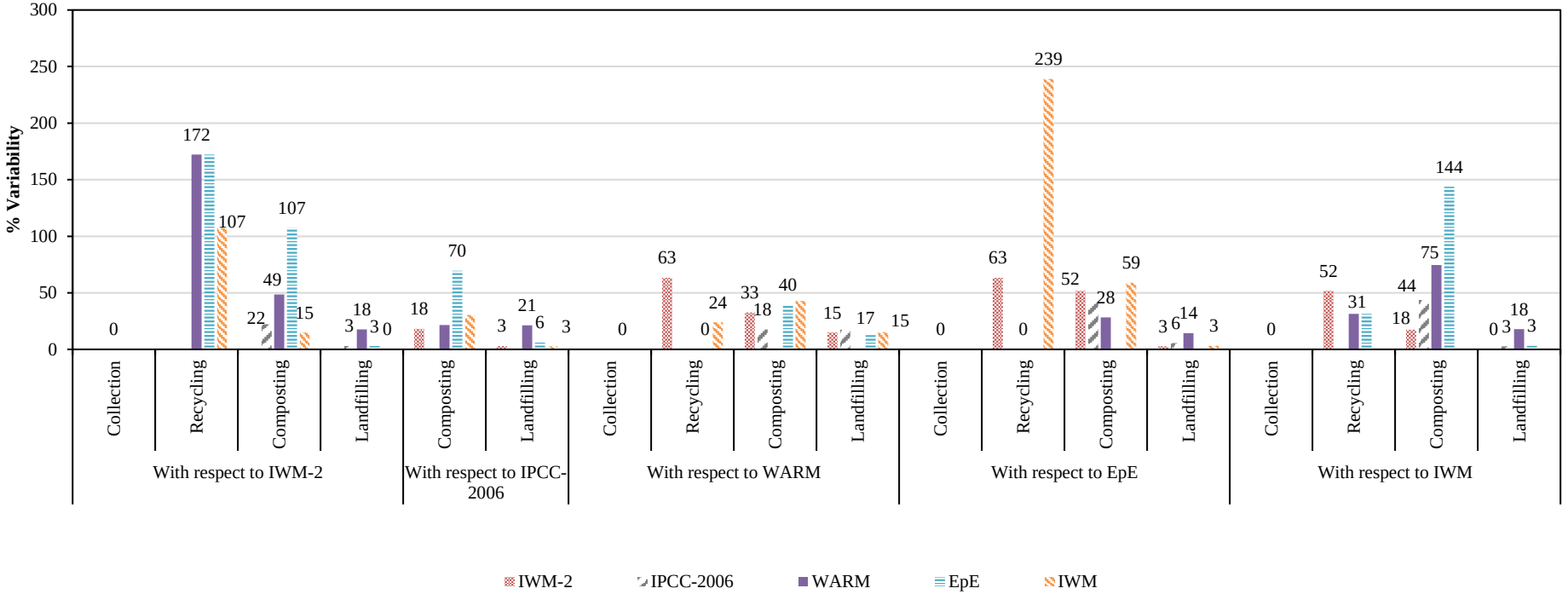


Figure 2.7. Absolute variability in emissions disaggregated by source
when considering each method at a time to be the base for the comparative assessment

Variability % = $|(\text{Value of tested method}_{(i)} - \text{Value of tested method}_{(j)}) / \text{Value of tested method}_{(i)}| \times 100$

2.4.4 Summary comparison

The comparative assessment defined several limitations in existing methods mainly at the level of neglecting upstream (e.g. fuel/energy and material provision) or downstream (e.g. avoided emissions from carbon storage and material recovery) processes. In addition, some methods do not address emissions from certain waste management processes such as flaring of LFG collected from landfilling or open dumping and burning. While the latter are improper, they remain common practices in developing economies where a high fraction of the waste is still disposed of in open dumps, or openly burned, or landfilled with an inefficient LFG collection and flaring system (Devkota et al., 2012). Furthermore, most methods were applied in developed economies with default data for respective countries and a lack of flexibility with regards to modifying input parameters as they are not readily accessible or adjustable. The latter is coupled with a difficulty to disaggregate emissions based on the scope of reporting whether for national inventorying (direct emissions) or for LCA (direct and indirect) decision-making and planning purposes. While existing accounting methods consider many direct and indirect contributions, most (except for EpE) do not consider emissions by type (direct vs. indirect). Similarly, most methods neglect downstream contributions with the exception of the WARM model that accounts for offset of CO_2 emissions from fertilizer and peat production or carbon storage from land application of compost, and savings from carbon storage during landfilling. In addition, existing methods (except for WARM) do not account for indirect emissions associated with the management of residues from waste incineration (savings from material recovery and load from bottom ash landfilling). Similarly, no method account for auxiliary fuel needed to satisfy the low heating value (LHV) during incineration and indirect emissions related to landfill construction.

All methods targeted developed economies with default input data introduced for specific locations and often with uncertainty about emission factors that as stated above, are not readily accessible or adjustable (Assamoi and Lawryshyn, 2012; Laurent et al., 2014). Arguably, the IPCC guidelines were developed to address these shortcomings, but these guidelines do not consider emissions savings from waste recycling and do not account for emissions from the collection of waste within the waste sector. The latter are embedded within the transport sector under energy and must be redirected under the waste sector for comparative purposes of emissions reduction targets and potential carbon credit from this sector. More importantly and due to lack of area-specific input data particularly EFs, the application of the IPCC guidelines has relied on borrowing such data from other locations, mainly developed economies, thus undermining the very purpose for which they were developed in the first place.

While it might be evident that methods with different scope of accounting will likely generate different emissions, the variations were equally significant for methods having similar accounting scope such as LCA-based methods. The variability can be attributed to how several influencing factors are controlled including system's boundary assumptions of waste management processes, the choice of gases and EFs3, as well as input data and parameters used to describe the MSW management system or using different waste and gas categories for composition4 and type of emissions5. Some of these

[3] The choice of gases and corresponding emission factors affect the results significantly, for example, IWM resulted in the highest emissions from collection and indirect emissions among methods because it considers that N_2O emissions are higher than CO_2 emissions during fuel combustion.

[4] For instance, while WARM (following EPA guidelines) considers a wide variety of waste categories and accounts for corresponding EFs, the IWM and IWM-2 methods consider EFs for < 5 categories.

[5] For example, IWM and IWM-2 neglect CH_4 and N_2O emissions from composting

factors are also related to geographical conditions (electricity generation and fuel consumption with corresponding EFs) while others are related to the equipment performance (efficiency factors). Concurrently, the results underline the interdependency of emissions and the amount of material applied to a specific process, which may differ with the default waste stream6 adopted by various methods.

2.4.5 Policy implications and future conceptual framework

At the policy planning level, the relationships between the quantification approach (or emissions accounting method) and carbon credit from waste management, can be schematically represented by Figure 2.8 where parameters adopted in quantifying emissions from waste management can affect carbon credits when assessing emissions mitigation, reduction targets, or NDCs under the Paris Agreement.

Despite voluntary and carbon market driven initiatives in developed economies, developing countries did not have mandatory obligations for reducing emissions under the Kyoto Protocol. The situation has changed following the Paris agreement (UNFCCC, 2015) whereby it became mandatory for all parties to report regularly on their emissions and implementation efforts through NDCs that incorporate attempts by each country to decrease national emissions and adapt to climate change impacts.

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[6] Although the same input of waste material was introduced in all methods, the amount of material diverted to a specific process differs depending on the waste distribution adopted in each method. For instance, in IWM-2, paper waste is diverted to composting by default.

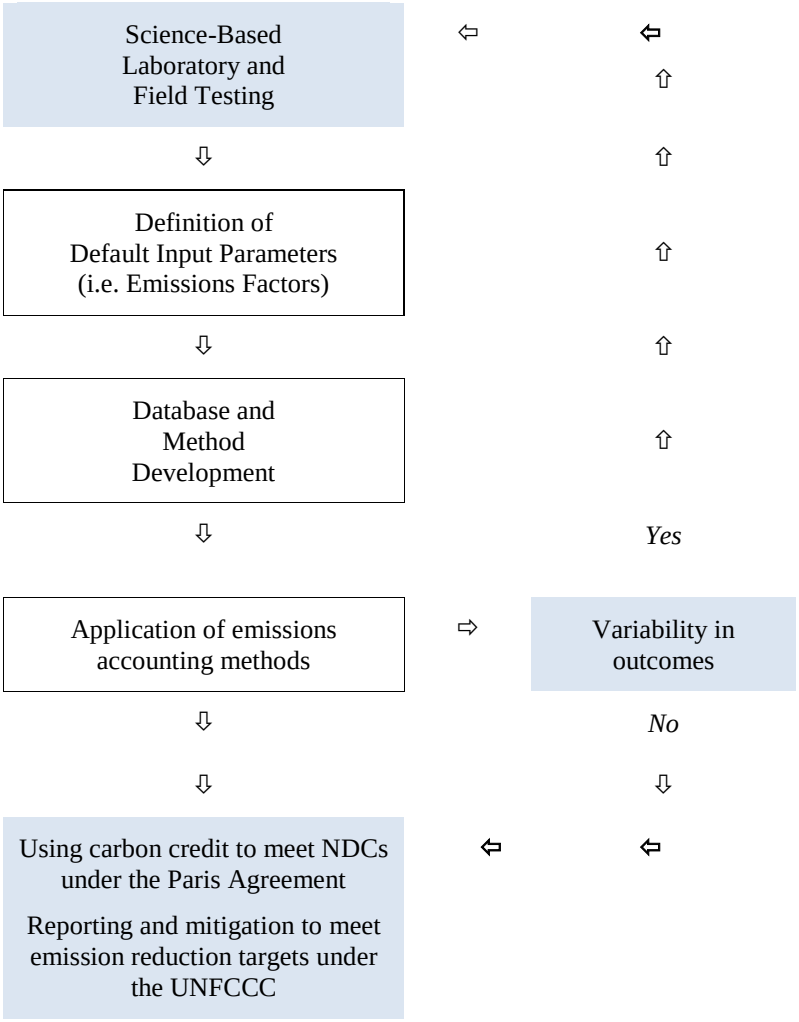

Figure 2.8. Impact of emission quantification in the context of using carbon credit to meet NDCs under the Paris Agreement or emissions reporting and mitigation under the UNFCCC

NDCs: Nationally Determined Contributions;
UNFCCC: United Nations Framework Convention on Climate Change

In this context, it is imperative to develop a well conceptualized and designed tool to harmonize and validate non-geographic assumptions towards strengthening modelling efforts with applicability to both developed and developing economies. Equally important, emissions accounting and reporting methods should include similar data that can then be used differently depending on the scope of reporting whether for national inventorying, LCA modelling purposes for planning and decision-making purposes, corporate reporting, or emission reduction targets using carbon credit. It is also

necessary to consolidate the reporting of emissions under existing methods, by providing a single framework such as the Upstream-Operating-Downstream approach (Gentil et al. 2009) to improve accuracy and robustness in reporting background data. Such a framework would build on existing emissions accounting methods with the aim of adding uniformity amongst methods by confirming clarity and traceability for the waste management data. Consequently, this will increase the credibility of mitigation initiatives in the waste management industry and demonstrate its commitment to climate change actions.

Accordingly, a conceptual framework model (Figure 2.9) was developed to address limitations discerned in this study (Table 2.6). The proposed framework can accommodate general and specific locations equally with input data from both developed and developing economies defined more explicitly all while offering users the flexibility of modifying input parameters in contrast to a closed source code. Last but not least, the proposed framework encompasses the ability to simulate emissions from a wider range of waste management processes. We re-emphasize that tested methods in the comparative assessment were selected based on their accessibility, common use worldwide, and sponsorship / approval by cities or countries where they were originally developed.

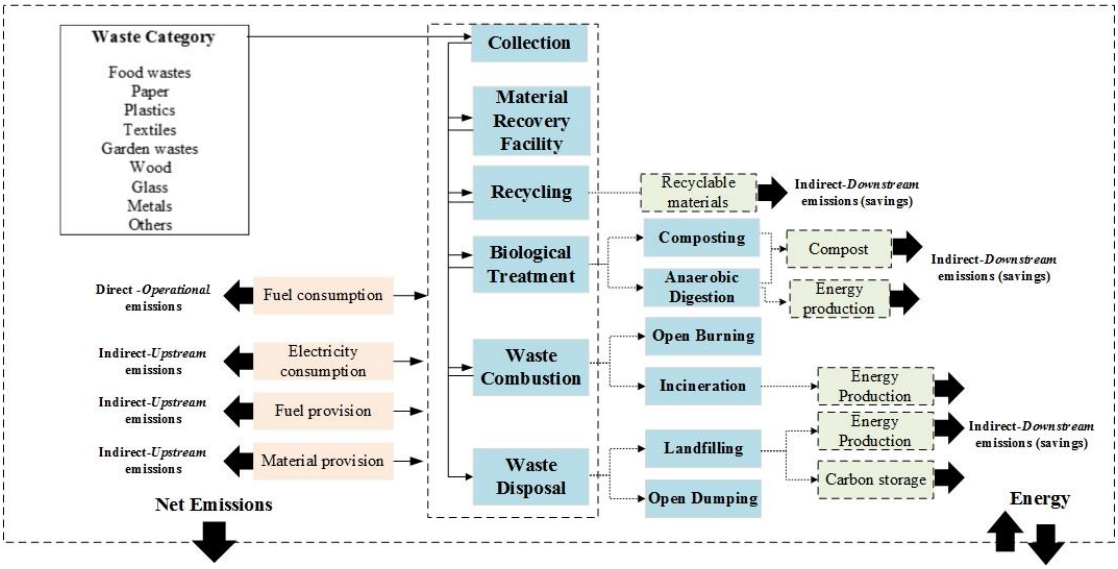

Figure 2.9. Proposed conceptual framework

Table 2.6 Comparison of proposed conceptual framework model with existing methods

		IPCC 2006	WARM	EpE	IWM	IWM-2	Framework
Database		Default	Default	User selected[a]	Default	Default	User selected[b]
Modifiable/ dynamic		N	N	Y	N	N	Y
Select emissions by type [c]		N	N	Y	N	N	Y
Select EF/input parameter		N	N	Y	N	N	Y
Select by gas type		Y	N	Y	Y	Y	Y
GWP_{100} Reference		N	N	N	N	N	Y
Collection/transport	*Fuel combustion*	N	Y	Y	Y	Y	Y
	Fuel provision	N	N	N	N	N	Y
Biological treatment	*Waste degradation*	Y	Y	Y	Y	Y	Y
	Fuel combustion	N	Y	Y	Y	Y	Y
	Electricity consumption	N	N	Y	Y	Y	Y
	Fuel provision	N	N	N	N	N	Y
	Carbon storage	N	Y	N	N	N	Y
	Peat substitution	N	Y	N	N	N	Y
	Energy recovery	N	Y	N	N	Y	Y
Incineration process	*Waste combustion*	Y	Y	Y	Y	Y	Y
	Electricity consumption	N	N	Y	N	N	Y
	Energy recovery	N	Y	Y	Y	Y	Y
	Material recovery	N	N	N	N	N	Y
	Fuel combustion	N	N	Y	N	N	Y
	Fuel provision	N	N	N	N	N	Y
Landfill processes	*Waste degradation*	Y	Y	Y	Y	Y	Y
	Fuel combustion	N	Y	Y	Y	Y	Y
	Electricity consumption	N	N	Y	Y	Y	Y
	Fuel provision	N	N	N	N	N	Y
	Material provision	N	N	N	N	N	Y
	Carbon storage	N	Y	N	N	N	Y
	Energy recovery	N	Y	Y	Y	Y	Y
	N_2O from flaring	N	N	N	Y	N	Y
Assessments	*Carbon Credit*	N	N	N	N	N	Y
	Economic	N	N	N	N	Y	Y
	Social	N	N	N	N	N	Y

IPCC 2006: Intergovernmental Panel on Climate Change 2006 Guidelines; **WARM:** Waste Reduction Model; **EpE:** Entreprises pour l'Environnemnent; **IWM**: Integrated Waste Management Model for municipalities; **IWM-2:** Integrated Waste Management Model-2.

[a] In order to calculate direct emissions from waste degradation in landfills, the user selects a common methodology and refers to the regulatory methodologies recommended by the authorities of the country where the site is located.

[b] Ability to disaggregate emissions based on scope of reporting whether for national / GHG inventorying or for LCA / planning and decision-making purposes.

[c] Type of emissions: indirect-upstream, direct-operational, and indirect-downstream contributions (direct and indirect).

2.5 Conclusion

This chapter examined the variability in aggregated and disaggregated emissions from waste management when using commonly adopted international methods (the UN IPCC 2006 Guidelines, the US EPA WARM, the EU EpE protocols, the Canadian IWM, and the UK IWM-2). The results reflect a persistent variability across methods in estimating emissions whether in total (aggregated), or by disaggregated sources (waste management process from collection to disposal), by gas or type (direct and indirect). All methods rely on default parameters that are invariably not representative of characteristics

encountered beyond the geographic location where the method was originally developed. The IPCC guidelines were intended specifically to address this limitation nevertheless key parameters remain largely not available for most countries with a common trend to still use those reported at locations with different characteristics. In addition, the IPCC guidelines that are advocated as a common international ground under the UNFCCC, still do not consider direct and indirect contributions from upstream or downstream processes within the waste management sector. This highlights the need for 1) developing key parameters when lacking with less reliance on those reported beyond the location under consideration; and 2) increased flexibility in accessing and changing default parameters to represent a wider context while accounting for direct and indirect contributions. A conceptual framework was developed to address the latter limitation and provide an improved future tool for assessing emissions reporting targets under the UNFCCC commitments or guiding decision making and reduction targets using carbon credit to meet NDCs under the Paris Agreement.

CHAPTER 3

AGGREGATED AND DISAGGREGATED DATA ABOUT DEFAULT EMISSION FACTORS IN EMISSIONS ACCOUNTING METHODS FROM THE WASTE SECTOR

3.1 Introduction

The dataset presented in this chapter is related to the research article entitled "Towards improving emissions accounting methods in waste management: A proposed framework" (Maalouf and El-Fadel, 2018) that examines the variability in aggregated and disaggregated emissions from waste management when using commonly adopted international methods (the UN IPCC 2006 Guidelines, the US EPA WARM, the EU EpE protocols, the Canadian IWM, and the UK IWM-2). The dataset presents the aggregated and disaggregated emission factors (EFs) used in existing accounting methods to estimate emissions from the waste sector. The EFs were retrieved from accounting methods to clarify their contribution to variability in estimating emissions across methods. The data contains three parts: aggregated EFs per tonne of waste category for individual waste management processes; disaggregated EFs per management process for a tonne of waste type; and emission flow diagrams of waste management systems for tested methods.

Table 3.1. Specifications Table

Subject area	*Environmental engineering*
More specific subject area	*Emission accounting from waste management*
Type of data	*Tables, figures, and text*
How data was acquired	*Secondary data sources (e.g. reports, literature, and existing models/software)*
Data format	*Raw and analyzed data*
Data source location	*Department of Civil & Environmental Engineering, American University of Beirut, Lebanon*
Data accessibility	*Data is included in this article*
Related research article	*Maalouf, A., El-Fadel, M. Towards improving emissions accounting methods in waste management: A proposed framework. J. Clean. Prod. 206 (2019)197-210. doi: http://dx.doi.org/10.1016/j.jclepro.2018.09. 014.*

Value of the Data

- The data consist of aggregated and disaggregated emission factors that are adopted in existing accounting methods to estimate emissions from the waste sector.
- A significant difference is evident in emission factors across tested methods.
- Data analysis accentuates the need for uniformity in emissions accounting methods and corresponding default parameters particularly emission factors.
- The data can guide the estimation process of emissions from the waste sector.
- The data can influence decision making when assessing emissions mitigation measures and reporting targets under the United Nations Framework Convention on Climate Change (UNFCCC) agreements or influence reduction targets using carbon credits to meet nationally determined contributions (NDCs) under the Paris Agreement.

3.2 Data

The data presented in this article provides details about emission factors (EFs) used in estimating emissions from the waste sector. The data clarifies the contribution to the variability in emissions when using commonly adopted international methods (the UN IPCC 2006 Guidelines, the US EPA WARM, the EU EpE protocols, the Canadian IWM, and the UK IWM-2). These methods were selected because they are publically accessible, widely reported in the literature, and adopted by cities or countries where they were originally developed. The Intergovernmental Panel on Climate Change (IPCC) guidelines in particular were supposedly put forth to standardize between methods at a global scale. The data consist of disaggregated EFs expressed in metric tonnes of CO_2 equivalents ($MTCO_2E$) per characteristic unit and refer to EFs separated by waste category, gas, waste processes, and type of emissions (direct or indirect). It also includes details on aggregated

EFs ($MTCO_2E$/ tonne of waste), which are the combined outcome of indirect-upstream, direct-operational, and indirect-downstream emissions from treating one tonne of waste by individual waste management processes. Note that waste always refers to wet waste. Moreover, given that the 100-year global warming potential (GWP100) for greenhouse gases (GHGs) has evolved with time as outlined in (Table 3.2), the GWP100 was adjusted in all methods to follow the IPCC (1995) reference definition. The latter was selected as a reference in all methods because most of them rely on the IPCC (1995) by default. Note that changing the GWP100 affect emissions estimation. For instance, WARM uses IPCC (2007) resulting in 19% increase in GWP100 of CH4, in comparison to IWM-2 (IPCC, 1995).

Tables

Table 3.3 to Table *3.7* show the aggregated default EFs per tonne of waste category for individual waste management processes. A further illustration of the EFs (disaggregated and aggregated) adopted in each method is presented in Appendix B Table B.1. Flow diagrams of waste management systems with energy sources and resulting emissions for each method are displayed in Figures Figure *3.1* to Figure *3.5*.

3.3 Experimental Design, Materials, and Methods

Data on EFs for various waste management processes was collected through secondary sources of accessible reports, literature, guidelines, and models/software. The data was categorized into:

(1) Disaggregated EFs, which are by definition factors determined from a number of processes representing characteristics calculated per unit of activity; thus, they are expressed in MTCO2E per characteristic unit (tonne of municipal solid waste treated; KWh of electricity; Liter of Diesel fuel) using a GWP100, (IPCC, 1995). EFs are fixed default values within every method except for the EpE method where the user can select EFs of recycling (adapted from USEPA/ICF, 2012) and landfilling (adapted from IPCC-2006 Guidelines).

(2) Aggregated EFs is the combined outcome of disaggregated EFs expressed in $MTCO_2E$ per tonne of waste category. Note that LFG (landfill gas collected) = 0.6; Electricity consumed = 32 kWh/tonne of waste composted, 70-80 kWh/tonne of waste incinerated, 68-50 kWh/tonne of waste anaerobically digested, and 8kWh/tonne of waste landfilled; Fuel consumed = ~2 Liters/tonne of waste landfilled, ~3.28 Liters/tonnne of waste composted, and 0.89 Liters/tonne of waste anaerobically digested.

Table 3.2. GWP for 100-year time horizon

GHGs	Symbol	First assessment report (FAR) IPCC (1990)	Second assessment report (SAR) IPCC (1995)	Third assessment report (TAR) IPCC (2001)	Fourth assessment report (AR4) IPCC (2007)	Fifth assessment report (AR5) IPCC (2013)[1]
Carbon dioxide	CO_2	1	1	1	1	1
Methane	CH_4	21	21	23	25	34
Nitrous oxide	N_2O	290	310	296	298	298

[1] Including climate-carbon feedbacks.

Table 3.3. Emission factors related to waste collection

Method	Type of EF	Values	Variability in EFs(%)[(d)]
IPCC-2006 [(a)]	Not considered		
EpE	Aggregated [(b)]	0.018	11-289
	Disaggregated [(c)]	$EF_{fuel\ CO_2}$= 0.0026	
IWM	Aggregated	0.07	70-74
	Disaggregated	$EF_{fuel\ CO_2} = 2.6x10^{-3}$	
		$EF_{fuel\ CH_4} = 2.8x10^{-6}$	
		$EF_{fuel\ N_2O} = 0.007$	
IWM-2	Aggregated	0.021	14-233
	Disaggregated	$EF_{fuel\ CO_2} = 0.003$	
		$EF_{fuel\ CH_4} = 7.7x10^{-5}$	
		$EF_{fuel\ N_2O} = 2.2x10^{-6}$	
WARM	Aggregated	0.02	10-250
	Disaggregated	$EF_{fuel\ CO_2} = 0.003$	

[(a)] The IPCC does not account for emissions from collection of waste within the waste sector. Such emissions are embedded within the Transport sector under Energy.
[(b)] Aggregated Emission Factor (EF): ($MTCO_2E$ per tonne of waste category) (GWP_{100}; IPCC, 1995).
[(c)] Disaggregated $EF_{fuel\ g}$ = Emission factor of gas *g* from fuel combustion ($MTCO_2E$/Liters of fuel) with 6.2 L of fuel consumed/tonne of waste collected in the study area (GWP_{100}; IPCC, 1995).
[(d)] The absolute variability in EFs is calculated with respect to each method.
Variability % = |(Value of tested $method_{(i)}$ - Value of tested $method_{(j)}$) / Value of tested $method_{(i)}$| x100

Table 3.4. Aggregated emission factors per tonne of waste category recycled ($MTCO_2E$ per tonne of waste category)

Waste Category	IWM	IWM-2	WARM
Paper	-0.83		-3.52
Plastics	-4.53	-1.20	-0.98
Textiles		-5.87	-2.37
Wood			-2.46
Glass	-0.92	-0.09	-0.28
Metals	-1.99	-4.55	-3.97

Table 3.5. Aggregated emission factors per tonne of waste category composted ($MTCO_2E$ per tonne of waste category)

Waste Category	IPCC-2006	EpE	IWM	IWM-2	WARM
Food			0.066	0.012	-0.184
Garden					-0.155
Other	0.177[(a)]	0.175[(b)]			

[(a)] Considers total mass of municipal solid waste (MSW) treated.
[(b)] Considers CH_4 emissions from the Organic fraction of MSW and N_2O emissions from MSW

Table 3.6. Aggregated emission factors per tonne of waste category landfilled ($MTCO_2E$ per tonne of waste category)

	IPCC-2006[a]	EpE[b]	IWM	IWM-2	WARM
Food	0.436		0.496	0.832	0.578
Paper	1.590		0.684	0.832	0.036
Plastics	0				0.006
Textiles	0.954			0.832	0.006
Garden	0.663				0.988
Wood	2.016				-0.614
Glass					0.006
Metals					0.006
Other		0.009			1.242

[a] Emissions from landfilling are calculated based on regulatory methodologies recommended by local authorities. It also considers direct emissions (from permanent thermal facilities and on-site mobile equipment) and indirect emissions (from electricity or heat consumption), and avoided emissions (from electricity and heat recovery).

[b] LCA-based methods consider methane emissions from landfilling of waste disposed in a selected inventory year (using the gas yield method), over a 100-year time horizon, while other methods such as the IPCC-2006 adopt the first order decay (FOD) that considers the cumulative emissions of waste deposited in previous years. Instead of accounting for emissions over a time-period and considering the accumulation of emissions for every year from previous years, year 0 was selected as the inventory year to account for the waste behavior of this year over a 100-year prediction.

Table 3.7. Aggregated emission factors per tonne of waste category incinerated ($MTCO_2E$ per tonne of waste category)

Waste Category	IPCC-2006	EpE	IWM	IWM-2	WARM
Food			-0.04	0.57	-0.12
Paper	0.03		-1.1	1.24	-0.42
Plastics	2.22		-1.71	2.65	1.56
Textiles	0.25			1.24	1.23
Garden					-0.19
Wood					-0.43
Glass			0.38	0.09	-0.02
Metals			0.5		-0.02
Other	0.022	0.382	-0.58	1.24	-0.01

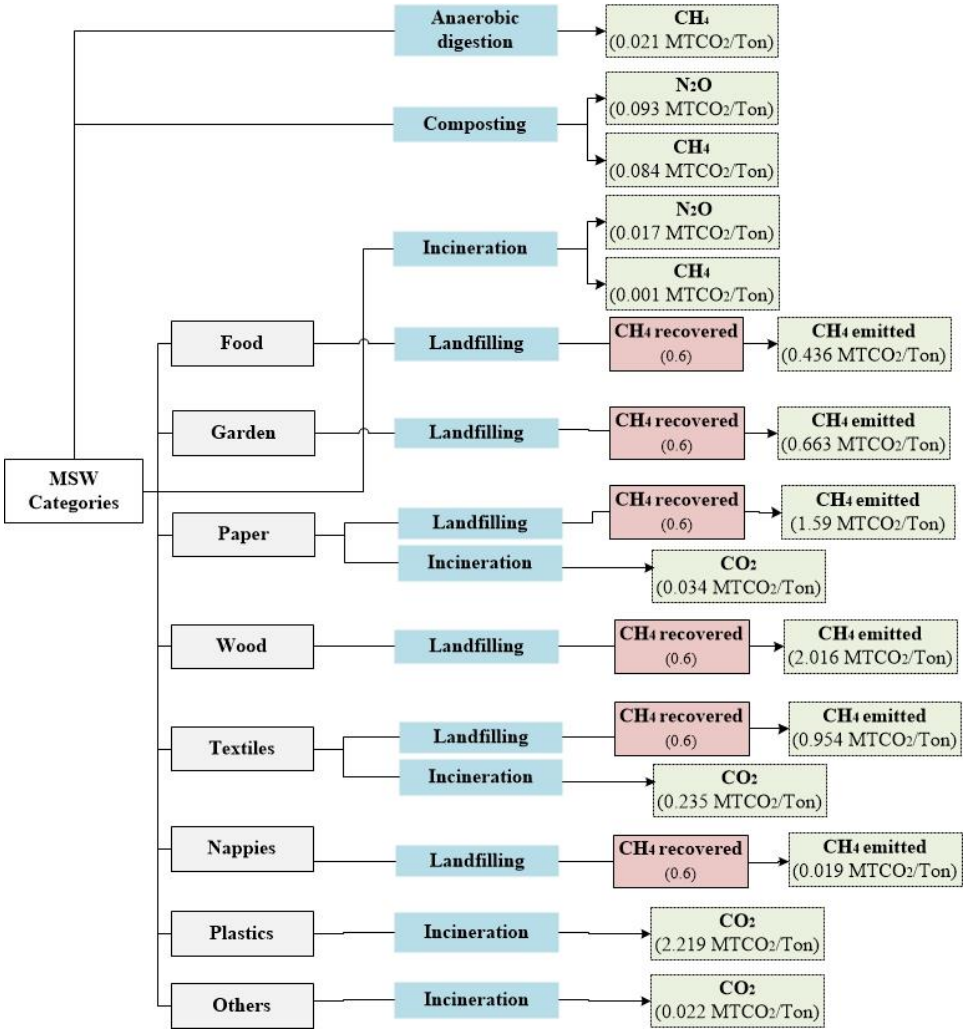

Figure 3.1 IPCC-2006

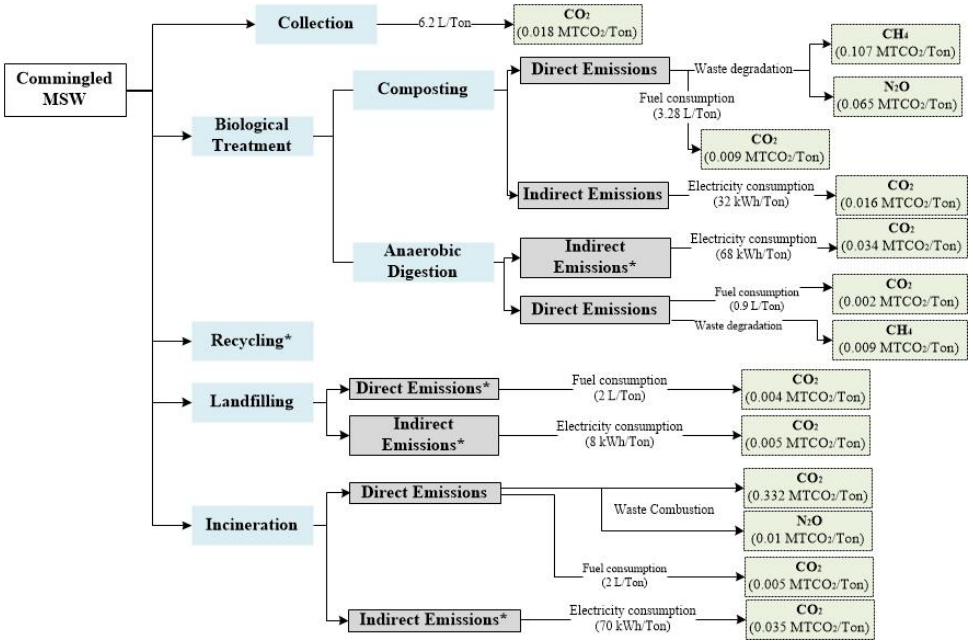

Figure 3.2. EpE protocol

*Note that EpE does not provide methodologies to estimate avoided emissions from recycling, energy recovery from anaerobic digestion, landfill, and incineration as well as direct emissions from waste degradation during landfilling.

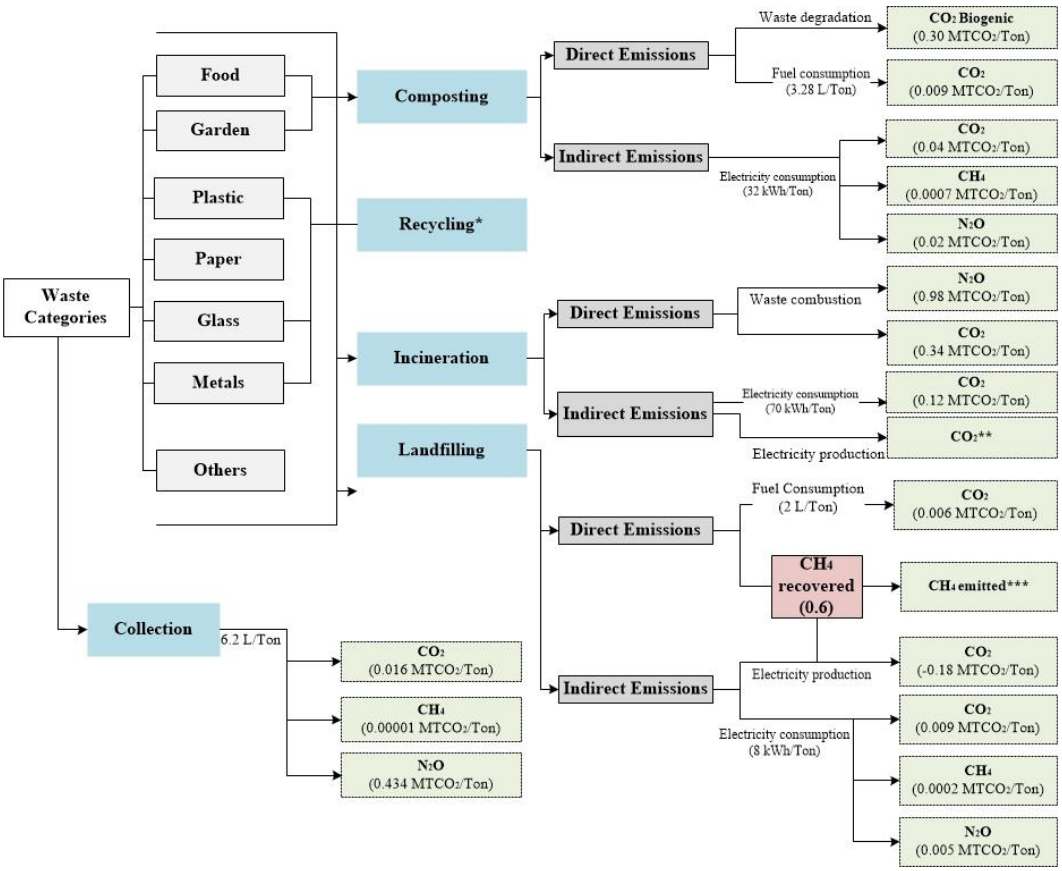

Figure 3.3. IWM

*During recycling IWM considers avoided emissions from plastics, glass, and metals
**During incineration IWM only considers CO_2 emissions from paper, glass, metals, plastics, food, and others
***During landfilling IWM only considers CH_4 emissions from paper, and food

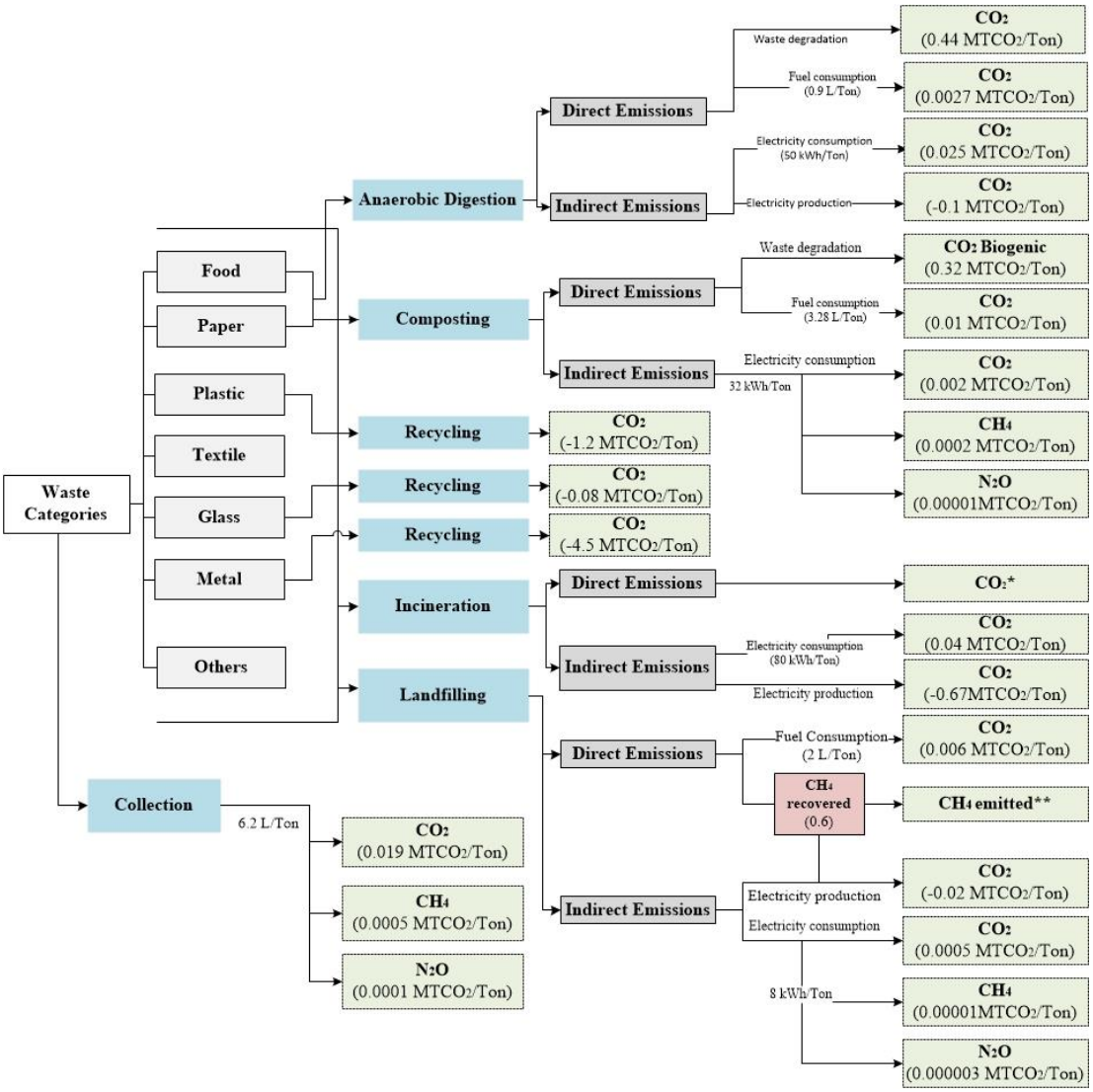

Figure 3.4. IWM-2

*During incineration IWM-2 only considers CO_2 emissions from paper, glass, plastics, textiles, food, and others
**During landfilling IWM-2 only considers CH_4 emissions from paper, textiles, and organics

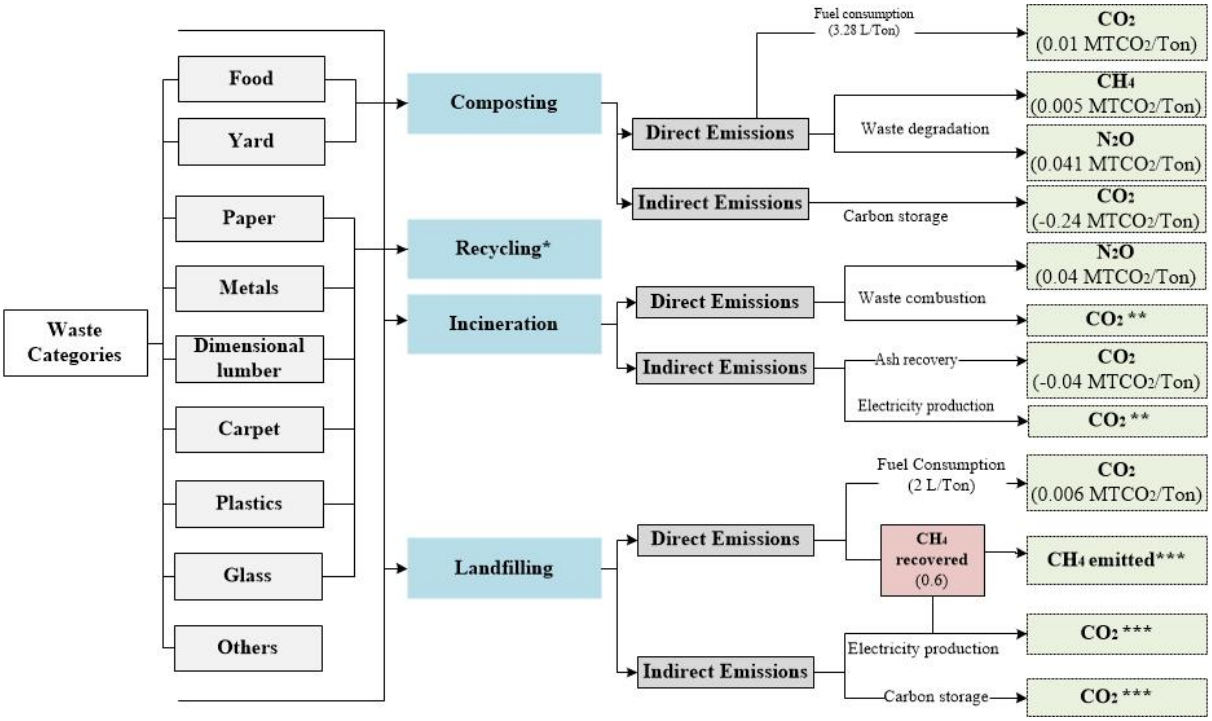

Figure 3.5. WARM

*During recycling WARM considers avoided emissions from paper, plastics, glass, carpet, dimensional lumber, and metals
**During incineration WARM only considers CO_2 emissions from paper, plastics, textiles, wood, food, and others
***During landfilling

CHAPTER 4

CARBON FOOTPRINT OF INTEGRATED WASTE MANAGEMENT SYSTEMS WITH IMPLICATIONS OF FOOD WASTE DIVERSION INTO THE WASTEWATER STREAM

4.1 Introduction

The waste sector contributes to greenhouse gas (GHG) emissions primarily in the form of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), and a few other gases with less significant quantities. These emissions are released through various processes and components of the waste management cycle (from collection to material recovery, biological and thermal processes, and landfilling) and accounted for ~3% ($1446x10^6$ $MTCO_2E$) of worldwide GHG emissions in 2010 (Blanco et al., 2014). While relatively a smaller contributor to total GHG emissions, the waste sector is considered to present an appreciable potential towards emissions' reduction through selected technologies (Bogner et al., 2007; IFEU and Ökoinstitut, 2010) particularly in developing economies where emissions from waste can account for a larger percentage reaching 15% of total country emissions due to the greater content of highly biodegradable organics (Friedrich and Trois, 2011; IFEU and Ökoinstitut, 2010).

Over the years, several studies and models have been reported to estimate emissions from the waste sector and assess environmental burdens associated with waste management processes (Dalemo et al., 1997; McDougall et al., 2001; El Hanandeh and El-Zein, 2010; Wang et al., 2012; Itoiz et al., 2013; Levis et al., 2013; Clavreul et al., 2014; EPA/ICF, 2016; Marchi et al., 2017; Thomsen et al., 2017). A review of studies (Table 4.1) assessing global warming factors (GWFs) for emission contribution

associated with waste management show that many models targeted individual processes and provided a solid theoretical understanding about the quantification of life cycle emissions from these processes. In this context, emissions from waste management encompasses indirect upstream emissions arising from inputs of materials and energy (electricity & fuel), direct operational emissions from system operation such as onsite operating equipment and waste degradation, and indirect downstream emissions (or savings) related to energy generation, materials substitution, and carbon storage (Gentil et al., 2009).

Table 4.1. Global Warming Factors per waste management process

Reference	**$MTCO_2E$ / 1 Ton of waste managed**					
	Collection	**Recycling**	**Composting**	**Anaerobic Digestion**	**Incineration**	**Landfilling**
Astrup *et al.* 2009a	-	Pl: -0.06– -1.6	-		-	-
Astrup *et al.* 2009b	-	-	-		0.35-0.53	-
Boldrin *et al.* 2009	-	-	-0.6		-	-
Moller *et al.* 2009				-0.01– -0.004		
Cadena *et al.* 2009	-	-	0.06		-	-
Chen & Lin 2008	0.016	-2.49	0.03		-0.22	0.02
Damgaard *et al.* 2009	-	Al: -5– -19.3 St: -0.6– -2.4	-		-	-
Eisted *et al.* 2009	0.005–0.03	-	-		-	-
Friedrich & Trois 2013a, b	0.015	-0.29– -19.11	0.186		-	0.44– 2.53
Hermann *et al.* 2011	-	-	1.1–1.7		-	-
ISWA, 2009	-	-0.19– -0.50	-		-	-
Kim & Kim 2010	-	-	0.12		-	1.10
Larsen *et al.* 2009a	0.004–0.03	-	-		-	-
Larsen *et al.* 2009b	-	G: -0.5– -1.5	-		-	-
Manfredi *et al.* 2009	-	-	-		-	0.30
Merrild & Christensen 2009b	-	P: -0.4– -4.4	-		-	-
Merrild & Christensen 2009a	-	W: -0.07– -1.4	-		-	-
Nguyen & Wilson 2010	0.008– 0.04	-	-		-	-
Smith *et al.* 2001	0.007	-	-0.037		-	-
Range	*0.004–0.04*	*-19.3– -0,06*	*-0.6–1.7*	*-0.01 – -0.004*	*-0.22–0.53*	*0.02–0.53*

Pl: Plastics, Al: Aluminum, St: Steel, G: Glass, W: Wood, P: paper

Existing models have continuously evolved providing a valuable holistic approach towards understanding the functionality of waste management systems while accounting for different interactions between processes and flows. Accordingly, recent efforts included integrated systems and complex technologies (e.g. combined treatment

of various waste streams and new thermal systems) (Clavreul et al., 2014; Hilty et al., 2014). However, commonly used GHG accounting models do not address certain upstream (fuel/energy and material provision) or downstream (avoided emissions from carbon storage and material recovery) processes. Additionally, commonly used models do not address emissions from certain waste management processes such as open burning or dumping and flaring of landfill gas (LFG). While such processes are seldom practiced in developed economies, they can be significant in the context of developing economies where a high fraction of the waste is still burned or disposed of in open dumps or landfilled with an inefficient LFG collection system or flared at best.

On the other hand, introducing a food waste disposer (FWD) policy to divert the organic fraction of food waste from the waste stream into the wastewater management system has proved to be an effective and economically viable alternative for waste reduction under certain conditions (Table 4.2). To the best of our knowledge, none of the existing models was designed to assess its impact on the emissions' inventory from the combined system of waste and wastewater including sludge management. It is worth noting that the Intergovernmental Panel on Climate Change (IPCC) guidelines for GHG emissions reporting from the waste sector includes emissions from both MSW and WW management systems and are reported under the same chapter.

While existing models have been highly recognized in assisting decision makers in defining cost effective and environmentally sound waste management alternatives, uncertainties in emission estimation seem inevitable when applied beyond their geographical boundaries where originally developed (Gentil et al., 2010; Friedrich and Trois, 2013a; Laurent et al., 2014). Equally important is the difficulty to disaggregate emissions using existing models based on scope of reporting whether for national

inventorying (direct emissions) or planning and decision-making purposes (direct and indirect emissions). Hence, Gentil et al. (2010) and Friedrich and Trois (2011) recognized the need for flexible tools designed to harmonize and validate non-geographic assumptions to strengthen modelling efforts as well as to be applicable in both developed and developing economies.

Table 4.2. Studies assessing the impacts of a Food Waste Disposer

Reference	**Impact coverage**	**Reported impact**
Maalouf and El-Fadel (2017)	Carbon footprint and economic	Positive
Bernstad Saraiva *et al.* (2016)	Carbon footprint and energy	Positive
Yi and Yoo (2014)	Environmental and economic	Positive
Bernstad *et al.* (2013)	Operational	Positive
Evans (2012)	Environmental and economic	Positive
Kim *et al.* (2011)	Economic	Positive
Evans *et al.* (2010)	Operational and economic	Positive
Battistoni *et al.* (2007)	Operational and economic	Positive
Constantinou (2007)	Operational and economic	Negative
Evans (2007)	Environmental and economic	Positive
Lundie and Peters (2005)	Environmental	Positive
Marashlian and El-Fadel (2005)	Operational and economic	Positive
Bolzonella et al. (2003)	Operational	Negative/Positive
CECED (2003)	Operational	Negative
Diggelman and Ham (2003)	Environmental and economic	Positive
Galil and Yaacov (2001)	Operational and economic	Negative/Positive
Wainberg *et al.* (2000)	Operational and economic	Positive
De Koning and Van der Graaf (1996)	Operational and economic	Positive
Raunkjaer *et al.* (1995)	Operational	Positive
Jones (1990)	Operational	Positive
Nilsson *et al.* (1990)	Operational	Negative
Iacovidou *et al.* (2012 a)	Operational and environmental	Positive
Iacovidou *et al.* (2012 b)	Operational and environmental	Positive

Building on past experience and limitations, a new model is developed with the objective to assess the impact on emissions from waste management systems when coupled with wastewater / sludge management through the introduction of a food waste disposer (FWD). The model allows the disaggregation of emissions by source (from collection to final disposal), or type (direct and indirect), or main gases (CH_4, CO_2, N_2O) and offers the flexibility of allowing the user to select processes or modify input parameters. The model was tested in the context of developed and developing economies to assess the impact of a FWD policy, waste composition and management processes, as

well as input parameters towards improved planning and decision making about process viability for investing in carbon credit.

4.2 Methodology

4.2.1 Model development

4.2.1.1 Theoretical Framework

The modeling domain (Figure 4.1) integrates the municipal solid waste (MSW) and wastewater (WW) management systems under a single framework upon the introduction of a food waste disposer (FWD) at the household level for grinding food waste and discharging it with the WW stream. The MSW management system consists of waste categories (c) (e.g. food wastes, papers, textiles) and various management processes including collection (C), recycling (R), composting (Co), anaerobic digestion (AD), incineration (I), landfilling (Lf), open dumping (OD), and open burning (OB) with corresponding emissions (E), materials recovered (recyclables r), by-products such as compost (Comp), and electricity produced (Elecpro). On the other hand, the WW management system may consist of aerobic (e.g. centralized aerobic treatment plant) or anaerobic processes (e.g. anaerobic lagoon, septic system) with several sludge management (SM) options including anaerobic digestion, composting, landfilling, incineration, or land application. Emissions from upstream, operational, and downstream processes (direct and indirect) were estimated in Metric Tons of CO_2 equivalent ($MTCO_2E$). Indirect upstream emissions are derived from inputs of materials and energy (electricity & fuel), direct operational emissions are emitted from systems' operation such as onsite operating equipment and waste degradation, and indirect downstream emissions (or savings) are related to energy generation, materials substitution, and carbon storage. GWFs are estimated from the sum of products of emission factors (EFs) for each gas

(CH_4, CO_2, N_2O) and the corresponding 100-year global warming potential (GWP100), then divided by the total tons of wet waste managed. When aggregated GWFs are added, they represent the potential contribution to global warming from upstream, operational, and downstream processes expressed in ($MTCO_2E$) per ton of wet waste (ww) managed: collected, recycled, composted, anaerobically digested, openly burned or incinerated, and openly dumped or landfilled. A GWF is positive when there is a contribution to global warming and negative when constituting offsets or savings. The estimation of emissions from individual waste management processes, k, follows with corresponding EFs. Data sources related to each process are detailed in the supplementary material.

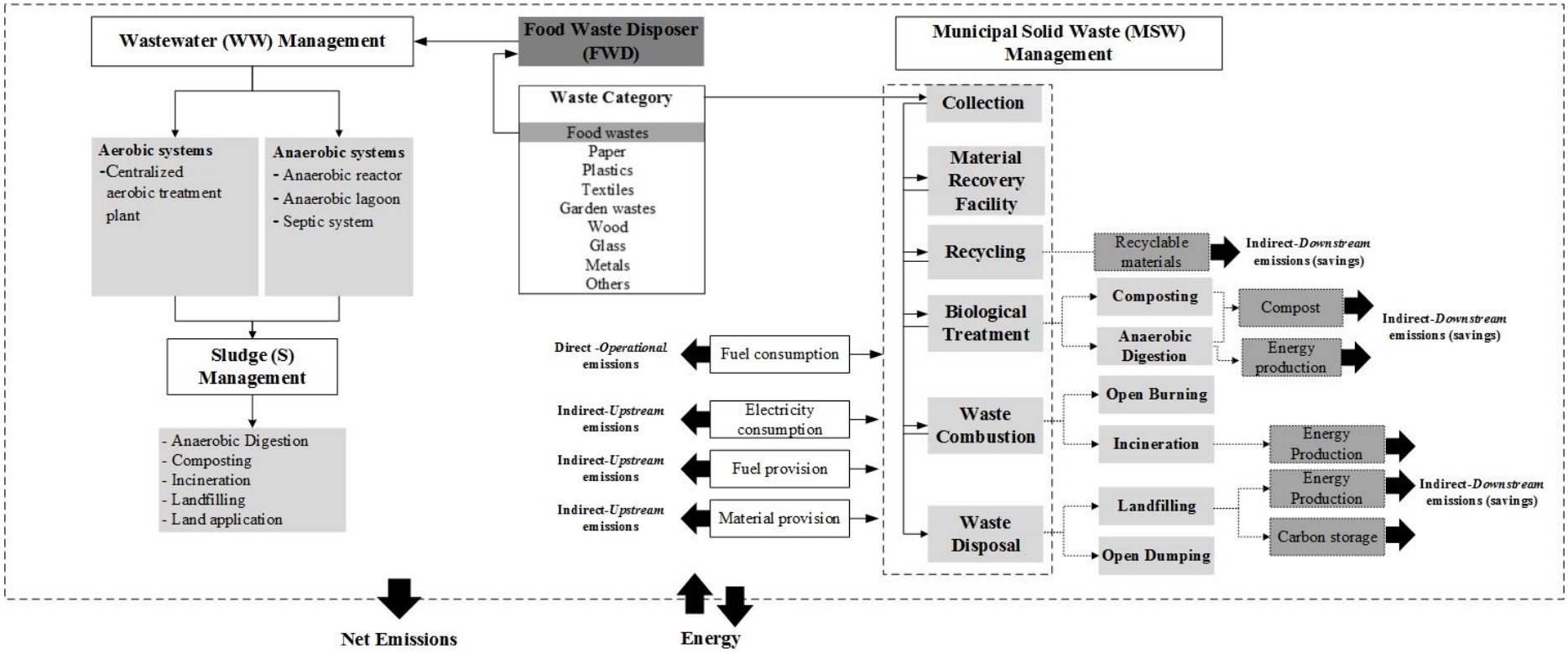

Figure 4.1. Model framework

4.2.1.2 Model input data

The collection of data related to waste generation and composition constitute the starting point for calculating emissions. Some data is related to geographical conditions (e.g. electricity generation and fuel consumption with corresponding emission factors) while others are location independent such as the performance of equipment at waste treatment plants (i.e. efficiency factors). It is imperative for the user to provide location-specific data to ensure representative results. When data is lacking, the model allows the user to select average input data by default. Typical waste data for running the model with corresponding units are summarized in Table 4.3.

Table 4.3. General description of input data

Input data	Unit	Default value	Source
Waste data			
Total mass of waste generated	Tons/ yr		User input
Population	Persons/ yr		User input
Generation rate	Tons/ cap/ yr		User input
Waste composition	% for each waste category		User input
Fraction of waste managed as MSW	% for each waste category		User input
Energy			
Electricity consumed during waste treatment	kWh/yr		User input/average default data from the literature
Emission factor for electricity mix	$MTCO_2E$/ kWh		User input/average default data corresponding to the study area (country or region) from IEA (2014)
Fuel consumption (collection & waste treatment facilities)	L/yr		User input/average default data from the literature
Emission factor for CO_2, CH_4, and N_2O ($EF_{fuel_{CO_2}}$, $EF_{fuel_{CH_4}}$, $EF_{fuel_{N_2O}}$) from fuel combustion	$MTCO_2E$ /L of diesel fuel	$EF_{fuel_{CO_2}}$ = 0.003 $EF_{fuel_{CH_4}}$ = $1.2x10^{-4}$ $EF_{fuel_{N_2O}}$ = $2.2x10^{-6}$	Fruergaard *et al.* (2009) McDougall *et al.* (2001) McDougall *et al.* (2001)
Emission factor for fuel provision $EF_{fuelpro_{CO_2}}$(extraction, processing, storage, and transportation of the fuel)	$MTCO_2E$ /L of diesel fuel	$EF_{fuel_{CO_2}}$ = $4.5x10^{-4}$	Fruergaard *et al.* (2009)
Waste treatment facilities			
Efficiency of combustion process	%		User input/average default data from the literature
Amount of material used during landfilling	Tons/yr		User input/average default data from the literature
Fraction of landfill gas collected	%		User input/average default data from the literature
Efficiency factor of electricity generation from waste facilities	%		User input/average default data from the literature
GWP_{100}	$MTCO_2E$ of gas *g* / MT of gas *g*	$CO_{2\ biogenic}$=0 CO_2=1 CH_4=34; N_2O=298 CH_4=21; N_2O=290 CH_4=21; N_2O=310 CH_4=23; N_2O=296 CH_4=25; N_2O=298	 IPCC (2013) IPCC (1990) IPCC (1995) IPCC (2001) Forster *et al.* (2007)

All input data can be modified by the user. Default averages are provided when data is not available. Note that data sources related to each process are detailed in the supplementary material (Tables SM.1 to SM.8 in Supplementary Material).

The total amount of waste (MT) is extrapolated from the population (Pop) based on per capita generation rates (GR) for a general study area and inventory year (t) (Equation 1), whereby each fraction (fc) for waste category (c), and the corresponding mass (Mc) can be expressed by Equation 2. Similarly, the fraction (fk) of waste collected, recycled, composted, anaerobically digested, incinerated or open burned, landfilled or open dumped is multiplied by the total waste generated (MT) to estimate the amount of waste (Mk) sent to a waste management process (k) (Equation 3).

$$M_T = Pop * G_R \quad (1)$$

$$M_c = f_c * M_T \quad \sum_{c=F}^{W} f_c = 1 \;;\; c \in \{F; G; G_A; M; N; O; P; P_L; T; W\} \quad (2)$$

$$M_k = f_k * M_T \quad \sum_{K=R}^{Lf} f_k = 1 \;;\; k \in \{R; C_o; AD; I; L_f; O_D; O_B\} \quad (3)$$

Where

M_T	Total mass of waste generated in inventory year *t* (Tons/yr)
Pop	Population in inventory year *t*
G_R	Generation rate (Tons/cap/yr)
c	Category of waste: *F*= Food waste; *G*= Glass; G_A= Garden waste; *M*= Metals; *N*= Nappies; *O*= others; *P*= Paper; P_L= Plastics; *T*= Textiles; *W*= Wood
M_c	Mass of waste category c generated in year t (Tons/yr)
f_c	Fraction of waste category *c*
f_k	Fraction of waste under management method *k*
M_K	Mass of waste under management process k [collection (C), recycling (R), composting (Co), anaerobic digestion (AD), incineration (I), landfilling (Lf), and open dumping (OD)] in inventory year t (Tons/yr)

4.2.1.3 Waste management elements

4.2.1.3.1 Collection and transport

Emissions from waste collection are estimated more accurately when using the annual fuel consumption (Vfuel) instead of distance traveled because it is more representative of collection trucks using the engine to power the compactor during waste collection particularly on roads with different topographical conditions that affect fuel consumption (Chen and Lin, 2008; Friedrich and Trois, 2013a). When data is lacking at a particular study area, the model allows the user to select the amount of fuel consumed

during waste collection depending on the characteristics of the study area (see Table C.1 in Appendix C). Emissions from waste collection and transport consist of direct emissions from fuel consumption during operation and while driving waste collection trucks to treatment facilities (Manfredi et al., 2009), as well as indirect upstream emissions from fuel provision during the extraction, processing, storage, and transport of fuel. Therefore, emissions from waste collection (EC) is calculated by multiplying the amount of waste collected (MC) and the liters of fuel consumed by the collection trucks (Vfuel C) per ton of waste collected with the corresponding emission factor to fuel combustion (EFfuelg) and provision (EFfuelproCO$_2$) for GHG g emitted (Equation 4). Note that the emissions associated with the manufacturing of collection trucks are not reported and hence not considered.

$$E_C = M_C * V_{fuel_C} * \left[\sum_{g=CH_4}^{N_2O} \left(EF_{fuel_g} * GWP_g \right) + EF_{fuelpro_{CO_2}} \right] \qquad (4)$$

k=C; and g = {CH$_4$; CO$_2$; N$_2$O}

Where

E_C	Direct *D* and indirect *ID* emissions for collection in inventory year *t* (MTCO$_2$E/yr)
M_C	Mass of waste generated and collected in year *t* (Tons/yr)
$Vfuel_C$	Volume of fuel consumed during waste collection by onsite mobile equipment in inventory year *t* (Liters/Ton treated/ yr)
$EFfuel_g$	Emission factor for fuel combustion for gas *g* (Metric Tons of g/Liter of fuel)
GWP_g	Global warming potential of gas *g* for a 100-year time horizon (MTCO$_2$E of gas g /MT of gas *g*)
$EFfuelpro_{CO_2}$	Emission factor for provision of fuel (MTCO$_2$/Liter of fuel)

4.2.1.3.1 Recycling

Direct emissions (ER D) from remanufacturing of recyclables (Equation 5) and indirect (avoided) emissions (ER ID) from virgin material manufacturing (Equation 6) are combined to estimate recycling net total emissions (ER) or savings as expressed in Equation 7 (see Table C.2 in Appendix C for corresponding EFs). The model takes into account GHG emitted from individual waste category c recycled and considers fractions

of residues from recovered recyclable materials (rrm) or from losses during the manufacturing of recyclables and virgin materials (rrrc). Note that the model assumes a closed-loop recycling process whereby end-of-life product is recycled into the same product.

$$E_{R\,D} = M_R * \sum_{c=G}^{W} \sum_{g=CH_4}^{N_2O} f_{R_c} * EF_{rm_{cg}} * (1 - rrm)(1 - rrr_c) GWP_g \quad (5)$$

$$E_{R\,ID} = M_R * \sum_{c=G}^{W} \sum_{g=CH_4}^{N_2O} f_{R_c} * EF_{vm_{cg}} * (1 - rrm)(1 - rrr_c) GWP_g \quad (6)$$

$$E_R = M_R * \sum_{c=G}^{W} \sum_{g=CH_4}^{N_2O} \left[f_{R_c} * \left(EF_{rm_{cg}} - EF_{vm_{cg}} \right) * (1 - rrm)(1 - rrr_c) \right] * GWP_g \quad (7)$$

k=R; g = {CH_4; CO_2; N_2O}; and c ={G; M; P; PL; T;W}

Where

$E_{R\,D}$	Direct *D* GHG emissions for recycling in inventory year *t* ($MTCO_2E$/yr)
M_R	Mass of waste recycled in year *t* (Tons/yr)
c	Waste category: G=Glass; M= Metal; P=Paper; PL=Plastic; T=Textile; W=Wood
f_{R_c}	Fraction of waste category *c* recycled
$EFrm_{c\,g}$	Emission factor of gas g from waste category *c* from re-manufacturing of recyclables *rm* (Tons of *g*/Ton of *rm*)
rrm	Fraction of residues from recovered recyclable materials
rrr_c	Fraction of residues from remanufacturing of recyclables and virgin material manufacturing of waste category *c*
GWP_g	Global warming potential of gas *g* for a 100-year time horizon ($MTCO_2E$ of gas g /MT of gas *g*)
$EFvm_{cg}$	Emission factor of avoided gas *g* from waste category c from virgin manufacturing *vm* (Tons of *g*/Ton of *vm*)
$E_{R\,ID}$	Indirect *ID* GHG emissions for recycling in inventory year *t* ($MTCO_2E$/yr)
E_R	Total net emissions from direct *D* and indirect *ID* GHGs emitted during recycling in inventory year *t* ($MTCO_2E$/yr)

4.2.1.3.2 Biological treatment

The model considers several composting technologies including open windrow, enclosed vessels or tunnels, and home composting, which differ in electricity and fuel consumption (see Table C.3 in Appendix C for default input data). Direct GHG emissions from waste decomposition during composting (ECo D) consist of biogenic CO_2 (considered neutral), N_2O, and minor amounts of CH_4, as well as emissions from fuel (VfuelCo) combustion by on-site mobile equipment (Equation 8). Indirect emissions from composting (ECoID) include upstream emissions from electricity consumption (ElecCo)

and the provision of fuel with corresponding emission factor ($EFfuelproCO_2$) related to extraction, processing, storage, and transportation of the fuel (Equation 9). In addition to avoided emissions from carbon storage associated with the application of compost to soils or substitution of peat production (Equation 9).

$$E_{C_o\,D} = M_{C_o} * \sum_{c=G}^{W} \sum_{g=CH_4}^{N_20} \left(f_{Co_c} * EF_{C_o\,c\,g} * GWP_g\right) + M_{C_o} * V_{fuel_{C_o}} \sum_{g=CH_4}^{N_20} \left(EF_{fuel_g} * GWP_g\right) \quad (8)$$

$$E_{C_o\,ID} = \sum_{g=CH_4}^{N_20} (M_{C_o} * Elec_{C_o} * EF_{elec_g} * GWP_g) + \left(M_{C_o} * V_{fuel_{C_o}} * EF_{fuelpro_{CO_2}}\right) - \left(M_{C_{omp}} * EF_{C_o\,cs\,c\,CO_2}\right) - \left(M_{C_{omp}} * EF_{C_o\,peat\,c\,g} * GWP_g\right) \quad (9)$$

g = {CH_4; CO_2; N_2O}; (i= D or ID; c=F, GA; k=C_O)

Where

$E_{Co\,D}$	Direct *D* GHG emissions during composting in inventory year *t* ($MTCO_2E$/yr)
M_{Co}	Mass of waste composted in inventory year *t* (Tons/yr)
c	Waste category: F=Food; GA= Garden waste
$f_{Co\,c}$	Fraction of waste category *c* composted
$EF_{Co\,c\,g}$	Emission factor for gas *g* from each ton of waste category *c* composted (Metric Tons of *g*/ Ton composted)
GWP_g	Global warming potential of gas *g* for a 100-year time horizon ($MTCO_2E$ of gas *g* /MT of gas *g*)
$Vfuel_{Co}$	Volume of fuel consumed during waste composting by onsite mobile equipment and combustion facilities in inventory year *t* (Liters/Ton treated/ yr)
$EFfuel_g$	Emission factor for fuel combustion for gas *g* (Metric Tons of g/Liter of fuel)
$E_{Co\,ID}$	Indirect *ID* GHG emissions for composting in inventory year *t* ($MTCO_2E$/yr)
$Elec_{Co}$	Electricity consumed during composing in inventory year *t* (kWh/Ton of ww treated)
$EF_{elec\,g}$	Emission factor of electricity consumed for gas *g* emitted based on national electricity grid (Tons of *g* /kWh)
$EFfuelproCO_2$	Emission factor for provision of fuel ($MTCO_2$/Liter of fuel)
M_{comp}	Mass of compost produced in inventory year *t* (Tons/yr), which is assumed 50% of M_{C_o}
$EF_{Co\,CS\,c\,CO2}$	Emission factor from carbon storage, which is avoided carbon from each ton of compost applied on land for agriculture ($MTCO_2E$/Ton of compost)
$EF_{Co\,peat\,c\,g}$	Avoided emission factor for gas *g* from compost used in growth of media preparation instead of peat produced ($MTCO_2E$ /Ton of compost)

Net emissions from anaerobic digestion (EAD) include direct emissions (EAD D) from fugitive CH_4 during waste degradation and emissions from fuel consumption (VfuelAD) of onsite operating equipment (Equation 10). Indirect emissions (EAD ID) include upstream emissions from electricity (ElecAD) and fuel provision. Indirect

downstream or avoided emissions from carbon storage are associated with the application of compost to soils or substitution of peat production as well as electricity production from biogas collected (Equation 11). See Table C.4 in Appendix C for corresponding input data.

$$E_{AD\,D} = M_{AD} * \sum_{c=G}^{W} \sum_{g=CH_4}^{N_20} \left(f_{AD_c} * EF_{AD\,c\,g} * GWP_g\right) + M_{AD} * V_{fuel}{}_{AD} \sum_{g=CH_4}^{N_20} \left(EF_{fuel_g} * GWP_g\right) \quad (10)$$

$$E_{AD\,ID} = \sum_{g=CH_4}^{N_20} (M_{AD} * Elec_{AD} * EF_{elec_g} * GWP_g) + \left(M_{AD} * V_{fuel}{}_{AD} * EF_{fuelpro}{}_{CO_2}\right) - \left(M_{C_{comp}} * EF_{AD\,CS\,c\,CO_2}\right) - \left(M_{C_{comp}} * EF_{AD\,peat\,c\,g} * GWP_g\right) - (Elecprod_{AD} * M_{AD} * EF_{elec\,g} * GWP_g) \quad (11)$$

g={CH_4}; (c=F, GA; k=AD)

Where

$E_{AD\,D}$	Direct *D* emissions during anaerobic waste decomposition in inventory year *t* ($MTCO_2E$/yr)
M_{AD}	Mass of waste under anaerobic digestion in inventory year *t* (Tons/yr)
c	Waste category: F=Food; GA= Garden waste
$f_{AD\,c}$	Fraction of waste category *c* under anaerobic digestion
$EF_{AD\,c\,g}$	Emission factor for gas *g* from each ton of waste category *c* under anaerobic digestion (Metric Tons of *g*/ Ton of waste anaerobically digested)
GWP_g	Global warming potential of gas *g* for a 100-year time horizon ($MTCO_2E$ of gas *g* /MT of gas *g*)
$Vfuel_{AD}$	Volume of fuel consumed during AD by onsite mobile equipment and combustion facilities in inventory year *t* (Liters/Ton treated/ yr)
$EFfuel_g$	Emission factor for fuel combustion for gas *g* (Metric Tons of g/Liter of fuel)
$E_{AD\,ID}$	Indirect *ID* emissions for anaerobic waste decomposition in inventory year *t* ($MTCO_2E$/yr)
$Elec_{AD}$	Electricity consumed during anaerobic digestion in inventory year *t* (kWh/Ton of ww treated)
$EF_{elec\,g}$	Emission factor of electricity consumed for gas *g* emitted based on national electricity grid (Tons of *g* /kWh)
$EFfuelproCO_2$	Emission factor for provision of fuel ($MTCO_2$/Liter of fuel)
M_{comp}	Mass of compost produced in inventory year *t* (Tons/yr), which is assumed 50% of M_{C_O}
$EF_{AD\,CS\,c\,CO2}$	Emission factor from carbon storage, which is avoided carbon from each ton of compost applied on land for agriculture ($MTCO_2E$/Ton of compost)
$EF_{AD\,peat\,c\,g}$	Emission factor for gas *g* from avoided fertilizer offsets, which is from each ton of digestate applied to agricultural land, thus avoiding synthetic fertilizer use ($MTCO_2E$ avoided /Ton of compost)
$Elecprod_{AD}$	Power potential during anaerobic digestion in inventory year *t* (kWh/ton of ww anaerobically digested)

4.2.1.3.3 Combustion

Direct emissions from incineration (EI D) are estimated using Equation 12 that considers emissions from waste combustion corresponding to individual waste category c as well as emissions from fuel combustion (VfuelI) of onsite operating equipment. The

latter (VfuelI) also includes the auxiliary amount of fuel needed when the low heating value (LHV) of waste is less than 5000–6000 kJ/kg (Chen and Christensen, 2010; Zhao et al., 2012), which is required to sustain the burning process. This is particularly important in developing economies where waste is characterized by a high biodegradable organic fraction and a high moisture content leading to a lower calorific value (Liu et al., 2017 a, b). The model accounts for a wide range of emission factors associated with various types of auxiliary fuel (e.g. diesel/gas oil, fuel oil, and hard coal for power plants) needed to satisfy the LHV depending on the study area (country or region).

Indirect emissions (EI ID) involve emissions from the management of solid residues generated from waste incineration (savings from slag recovery and load from bottom ash landfilling) as well as avoided emissions from electricity production (Equation 13) which depends on the: (1) energy content of mixed waste or of waste category c (ElecprodI c) in KWh/Ton of waste (see Table C.5), (2) combustion system efficiency, a, in converting the energy content of waste materials to recovered electricity, and (3) the emission factor of electricity avoided (EFelec g). See Table C.5 in supplementary material for corresponding input data. The model also accounts for direct emissions during waste combustion from open burning (Equation 14), which is a common practice in developing economies.

$$E_{I\,D} = M_I * \sum_{c=F}^{W} \sum_{g=CO_2}^{N_20} (f_{I_c} * EF_{I\,c\,g} * GWP_g) + M_I * V_{fuel_I} \sum_{g=CH_4}^{N_20} \left(EF_{fuel_g} * GWP_g\right) \quad (12)$$

$$E_{I\,ID} = \sum_{g=CH_4}^{N_20} (M_I * Elec_I * EF_{elec_g} * GWP_g) + \left(M_I * V_{fuel_I} * EF_{fuelpro_{CO_2}}\right) - M_I * \sum_{c=F}^{W} \sum_{g=CO_2}^{N_20} (Elecprod_{I\,c} * f_{I_c} * a * EF_{elec\,g} * GWP_g) + (M_{I\,residues} * EF_{I\,residues}) \quad (13)$$

g = {CH_4; CO_2; N_2O}; (k=; i= D or ID); c={F; G; GA; M; N; O; P; PL; T; W}

$$E_{OB} = f_{OB} * M_{OB} * \sum_{g=CO_2}^{N_2 0} (EF_{OB\ g} * GWP_g) \quad (14)$$

g = {CH_4; CO_2}; c={F; G; GA; M; N; O; P; PL; T; W}; (K=OB, i=D)

Where

$E_{I\,D}$	Direct emissions during incineration in inventory year *t* ($MTCO_2E$/yr)
M_I	Total mass of waste incinerated in inventory year *t* (Tons/yr)
c	Waste category: *F*=Food; *G*=Glass; *M*=Metals; *O*=others; *P*=Paper; P_L=Plastics; *T*=Textiles; *W*=Wood
$f_{I\,c}$	Fraction of waste category *c* incinerated
$EF_{I\,c\,g}$	Emission factor for gas g from each ton of waste category c incinerated (Metric Tons of g/ Ton treated)
GWP_g	Global warming potential of gas *g* for a 100-year time horizon ($MTCO_2E$ of gas *g* /MT of gas *g*)
$Vfuel_I$	Volume of fuel consumed during waste management method *k* by onsite mobile equipment and combustion facilities as well as auxiliary fuel needed to satisfy the LHV depending on the study area, in inventory year *t* (Liters/Ton treated/ yr)
$EFfuel_g$	Emission factor for fuel combustion for gas *g* (Metric Tons of g/Liter of fuel)
$E_{I\,ID}$	Indirect emissions for incineration in inventory year *t* ($MTCO_2E$/yr)
$Elec_{\,I}$	Electricity consumed during incineration process in inventory year *t* (kWh/Ton of ww treated)
$EF_{elec\,g}$	Emission factor of electricity consumed for gas *g* emitted based on national electricity grid (Tons of *g* / kWh)
$EFfuelproCO_2$	Emission factor for provision of fuel ($MTCO_2$/Liter of fuel)
$Elecprod_{I\,c}$	Power potential during incineration in inventory year *t* [Electricity produced/Mass of category c incinerated (kWh/Ton of ww incinerated)]
a	Mass burn combustion system efficiency (fraction) (a=0.178)
$M_{I\,residues}$	Mass of residues generated from incineration in year *t* (Tons/yr) 0.23 Tons/Ton of waste; (Astrup *et al.*, 2009b)
$EF_I\,residues$	Emission factor of management of residues produced from landfilling, which consist of savings from slag recovery and emissions from landfilling of bottom ash ($MTCO_2E$/Tons of residues)
E_{OB}	Direct emissions for open burning in inventory year *t* ($MTCO_2E$/yr)
M_{OB}	Total mass of waste open burned in inventory year *t* (Tons/yr)
$EF_{OB\,g}$	Emission factor for gas *g* from each ton of waste open burned (Metric Tons of *g*/ Ton treated)

4.2.1.3.4 Landfilling

Direct emissions from landfilling (ELf D) consist of emissions from waste degradation (CH_4 and N_2O) and fuel used for onsite activities (mobile equipment, electric generators, dozers, compactors and other landfill vehicles) (Equation 15). The model estimates emissions from landfills 1) without LFG recovery systems; (2) with flaring of recovered CH_4 and N_2O emissions; (3) combustion of CH_4 for energy recovery, depending on the LFG recovery scenario. Note that CO_2 emissions from waste degradation and flaring were considered as biogenic sources. The model allows the estimation of methane emissions from individual waste category c (see Table C.6) in

accordance to the theoretical yield assuming that CH_4 emissions are released in the same year of waste deposition. Fuel combusted by waste collection trucks while unloading at and driving to the site are ascribed to collection and transport of waste.

$$E_{L_f\,D} = \left(M_{L_f} * \sum_{c=F}^{W} f_{Lf_c} * EF_{L_f\,c\,CH_4} * GWP_{CH_4}\right) - b + \left(b * EF_{L_f\,N_2O,flaring} * GWP_{N_2O}\right) + \sum_{g=CH_4}^{N_20} \left(M_{Lf} * V_{fuel_{L_f}} * EF_{fuel_g} * GWP_g\right) \quad (15)$$

For k=Lf; i=D; c={F;G;GA;M;N;P;PL;T;W;O}; b=fraction of CH_4 collected *amount of CH_4 generated

Where

$Ef_{Lf\,D}$	Direct emissions during landfilling in inventory year *t* ($MTCO_2E$/yr)
M_{Lf}	Mass of waste landfilled in year *t* (Tons/yr)
c	Waste category: *F*= Food waste; *G*= Glass; GA=Garden; *M*= Metals; *O*= others; *P*= Paper; P_L= Plastics; *T*= Textiles; *W*= Wood
f_{Lf_c}	Fraction of waste category *c* landfilled
$EF_{Lf\,c\,CH4}$	Emission factor for CH_4 from each ton of waste category *c* landfilled (Metric Tons of *g*/ Ton landfilled)
GWP_{CH4}	Global warming potential of CH_4 for a 100-year time horizon ($MTCO_2E$ of gas g /MT of CH_4) (depending on the selected IPCC guideline)
b	Amount of recovered methane (MT of CH_4/yr) (fraction of CH_4 recovered*$MTCH_4$ generated)
$EF_{Lf\,N2O\,flaring}$	Emission factor of N_2O from methane combustion during flaring ($MTCO_2E$ of N_2O/ $MTCO_2E$ of R)
GWP_{N2O}	Global warming potential of N_2O for a 100-year time horizon ($MTCO_2E$ of gas g /MT of N_2O) (depending on the selected IPCC guideline)
$Vfuel_{Lf}$	Volume of fuel consumed during landfilling by onsite mobile equipment and combustion facilities in inventory year *t* (Liters/Ton treated/ yr)
$EFfuel_g$	Emission factor for fuel combustion for gas *g* (Metric Tons of g/Liter of fuel)
GWP_g	Global warming potential of gas *g* for a 100-year time horizon ($MTCO_2E$ of gas *g* /MT of gas *g*)

Indirect emissions from landfilling (ELf ID) of waste include upstream emissions associated with electricity (ElecLf) and fuel provision as well as emissions during landfill construction (provision of liner materials and construction of drainage system, etc.) as expressed in Equation 16. Indirect downstream emissions consist of avoided emissions from carbon storage and electricity production, which is dependent on the (1) energy content of recovered methane being combusted (ElecprodLf) (see Table SM.6) in kWh/ MT of CH4 recovered; (2) capacity factor for electricity generation, a; and (3) the emission factor of electricity avoided (EFelec g), depending on the energy mix in the study area (country or region) (Equation 16).

$$E_{L_f\,ID} = \left[\sum_{g=CH_4}^{N_20} (M_{L_f} * Elec_{L_f} * EF_{elec_g} * GWP_g) + \left(M_{L_f} * Vfuel_{L_f} * EF_{fuelpro_{CO_2}}\right) + \sum_{g=CH_4}^{N_20} \left(M_{Lf} * V_{fuel_{Lf\,const}} * EF_{fuel_g} * GWP_g\right) + \left(M_{L_f\,const} * EF_{L_f\,const\,g} * GWP_g\right)\right] - \sum_{g=CO_2}^{N_20} (Elecprod_{L_f} * b * a * EF_{elec\,g} * GWP_g) - \left(M_{L_f} * EF_{L_f\,CS}\right) \quad (16)$$

For k=Lf; i=ID; g = {CH_4; CO_2; N_2O}; b=fraction of CH4 collected *amount of CH4 generated

Where

$E_{Lf\,ID}$	Net indirect emissions for landfilling in inventory year *t* ($MTCO_2E$/yr)
M_{Lf}	Mass of waste landfilled in inventory year T (Tons/yr)
$Elec_{Lf}$	Electricity consumed during landfilling in inventory year *t* (kWh/Ton of ww treated)
$EF_{elec\,g}$	Emission factor of electricity consumed for gas *g* emitted based on national electricity grid (Tons of *g* /kWh)
GWP_g	Global warming potential of gas *g* for a 100-year time horizon ($MTCO_2E$ of gas *g* /MT of gas *g*)
$Vfuel_{Lf}$	Volume of fuel consumed during landfilling by onsite mobile equipment and combustion facilities in inventory year *t* (Liters/Ton treated/ yr)
$EFfuelproCO_2$	Emission factor for provision of fuel ($MTCO_2$/Liter of fuel)
$Vfuel_{Lf\,const}$	Fuel consumed for soil works during landfill construction in inventory year *t* (Liters/Ton treated/yr)
$EFfuel_g$	Emission factor for fuel combustion for gas *g* (Metric Tons of g/Liter of fuel)
$M_{Lf\,const}$	Mass of material used in landfill construction in inventory year *t* (Tons/yr)
$EF_{Lf\,const\,g}$	Emission factor for gas *g* from each ton of material used in landfill construction (Metric Tons of *g*/ Ton of material)
$Elecprod_{Lf}$	Electricity produced from CH_4 combusted (kWh/Metric Tons of CH_4 recovered) in inventory year *t* during landfilling
b	Amount of recovered methane (MT of CH_4/yr) (fraction of CH_4 recovered*$MTCH_4$ generated
a	Capacity factor for electricity generation (fraction) (a=0.85)
$EF_{Lf\,CS}$	Emission factor from waste landfilled avoided by carbon (biogenic) binding after 100 years ($MTCO_2$/ Ton of ww treated)

4.2.1.3.5 Open dumping

Open dumping comprises sites not meeting the criteria of managed solid waste disposal sites and which are classified as shallow (depth < 5 m) or deep (depth > 5 m and/or high-water table near ground level) (IPCC, 2006). Direct emissions from open dumping (EOD) of waste are calculated relative to controlled landfill sites whereby IPCC (2006) and UNFCCC (2008) recommended the use of a methane correction factor to account for a larger fraction of waste that is likely to decompose aerobically contributing to a lower amount of methane generation, as expressed in Equations 17.

$$E_{OD} = \left(M_{OD} * \sum_{c=F}^{W} f_{OD_c} * EF_{OD\,c\,CH_4} * GWP_{CH_4} * MCF\right) \quad (17)$$

For k=OD; c= {F; G; GA; M; N; P; PL; T; W; O}

Where

E_{OD}	Direct emissions during open dumping of waste of category c in inventory year *t* ($MTCO_2E/yr$)
M_{OD}	Mass of waste disposed in an open dumpsite in year *t* (Tons/yr)
c	Waste category: *F*= Food waste; *G*= Glass; GA=Garden; *M*= Metals; *O*= others; *P*= Paper; P_L= Plastics; *T*= Textiles; *W*= Wood
f_{OD_c}	Fraction of waste category *c* disposed in an open dumpsite
$EF_{OD\ c\ CH4}$	Emission factor for CH_4 from each ton of waste category *c* disposed in an open dumpsite (Metric Tons of *g*/ Ton open dumping) (equal to EFs of landfilling)
GWP_{CH4}	Global warming potential of CH_4 for a 100-year time horizon ($MTCO_2E$ of gas g /MT of CH_4) (depending on the selected IPCC guideline)
MCF	Methane correction factor such as MCF= 0.4 for an unmanaged-shallow (depth <5 m) solid waste disposal sites; MCF=0.8 for unmanaged solid waste disposal sites – deep (>5 m) and/or high water table; and MCF=1 for a managed landfill site, based on IPCC 2006 Guidelines.

4.2.1.4 Wastewater and sludge management

Under anaerobic conditions, wastewater and sludge produce CH_4 in quantities proportional to the degradable organic content and measured by proxy using BOD levels (IPCC, 2006). Accordingly, the total emissions from the use of a FWD (EFWD, Equation 18) consist of direct emissions from wastewater due to the additional BOD loading (Equation 19) and emissions associated with sludge management (such as landfilling, composting, incineration, and anaerobic digestion) as expressed in Equation 20 (IPCC, 2006). Emissions associated with landfilling of sludge were calculated using the first order decay (FOD) method adapted from IPCC (2006). The anticipated increase in BOD loading and the additional sludge generated due to the use of FWDs can be estimated based on laboratory testing (Marashlian and El-Fadel, 2005). Furthermore, avoided emissions associated with energy recovery from anaerobic treatment systems, can be estimated based on parameters of typical operating facilities (Diggelman and Ham, 2003). Energy consumption by the use of FWDs was assumed to be negliglible as reported by many studies (Iacovidou et al., 2012).

$$E_{FWD} = E_{FWD\ BOD} + E_{FWD\ S} \quad (18)$$

$$E_{FWD\ BOD} = (BOD - S) * EF_{FWD\ BOD\ F\ g} * MCF * GWP_g - (Elecprod_{FWD} * b * EF_{elec\ g} * GWP_g) \quad (19)$$

$$BOD = (M_F * m * z)\,(MC + w) * X_{BOD\ Food} * d$$

$S = (M_F * m * z)\,(MC + w) * ss * d$

g= {CH_4}; (c=F; k = FWD)

$$E_{FWD\,S} = S * EF_{FWD\,S\,F\,g} * GWP_g - (Elecprod_{FWD} * b * EF_{elec\,g} * GWP_g) \quad (20)$$

g={CH_4; N_2O}; (c=F; k=FWD)

Where

E_{FWD}	Total emissions from the diversion of waste to a FWD in inventory year *t* ($MTCO_2E$/yr)
$E_{FWD\,BOD}$	Emissions from the treatment of increased BOD loading due to the diversion of waste to a FWD in inventory year *t* ($MTCO_2E$/yr)
$E_{FWD\,S}$	Emissions from treatment of the increased sludge generated due to the diversion of waste to FWD in inventory year *t* ($MTCO_2E$/yr)
BOD	Added Biochemical oxygen demand (BOD) loading (Tons/yr)
S	Organic component removed as sludge due to the integration of a FWD policy in inventory year *t* (Tons/yr)
$EF_{FWD\,BOD\,F\,g}$	Emission factor for gas *g* from each ton of food waste grinded and disposed into the wastewater system (Metric Tons of *g*/ ton of waste grinded) (0.6 Ton CH_4/Ton BOD; IPCC, 2006)
MCF	Methane correction factor for wastewater treatment systems based on IPCC 2006 Guidelines (Table SM.7)
GWP_g	Global warming potential of gas *g* for a 100-year time horizon ($MTCO_2E$ of gas *g* /MT of gas *g*)
$Elecprod_{FWD}$	Power potential during anaerobic wastewater and MSW management methods in inventory year t, which is equal to Electricity produced from mass of organic waste/sludge anaerobically digested (190 kWh/ton of wet waste and 160 kWh/ton of sludge in anaerobic digestion; (Diggelman and Ham, 2003)
b	Mass of organic waste treated in anaerobic digestion (M_{AD}) (Tons/yr)
$EF_{elec\,g}$	Emission factor of electricity consumed or recovered for gas *g* emitted or avoided based on national electricity grid (Metric tons of *g* / kWh)
M_F	Mass of food waste generated in inventory year *t* (*F=Food*) (Tons/yr)
m	Market penetration of food waste disposers, %
z	Estimated food ground at the household level, %
MC	Average Moisture content of food waste (70%, Tchobanoglous *et al.*, 1993)
w	Volume of water needed to grind 1 Ton of organic food (11,700 L/Ton)
$X_{BOD\,Food}$	Average concentration of BOD of food waste based on experimental results (7042 mg/L, Marashlian and El-Fadel, 2005)
d	Water density (10^{-3} Ton/L)
ss	Average concentration of settable solids of the food waste based on experimental results (3327 mg/L, Wainberg *et al.*, 2000)
$EF_{FWD\,S\,F\,g}$	Emission factor for gas g from sludge treatment (Metric Tons of g / Ton of waste grinded) (Table SM.8)

4.2.2 Model application

The model was tested in the context of developed and developing economies where waste management determinants (e.g. waste generation and composition, energy consumption, management systems, and technological performance etc.) differ (Table 4.4).

Table 4.4. Average input parameters for developing vs developed economies

Input Parameter	Developed economy		Developing economy		Reference
Waste composition (%)	– Food	(30)	– Food	(60)	World Bank (2012); IPCC (2006)
	– Papers	(31)	– Papers	(5)	
	– Plastics	(11)	– Plastics	(8)	
	– Textiles	(3)	– Textiles	(2)	
	– Wood	(5)	– Wood	(6)	
	– Glass	(7)	– Glass	(3)	
	– Metals	(6)	– Metals	(3)	
	– Others	(7)	– Others	(13)	
MSW management method					
Recycling (%)	23		3		World Bank (2012)
Compositing (%)	12		2		World Bank (2012)
Incineration with energy recovery (%)	22		0		World Bank (2012)
Landfilling (%)	43		78		World Bank (2012)
Fraction of landfill gas collected (%)[(a)]	60		31		EPA/ICF (2016); Banar *et al.* (2009)
Open dumping (%)	0		17		World Bank (2012)
Energy					
Emission factor for electricity grid mi. (MTCO$_2$E/kWh)	$4x10^{-4}$[(c)]		$6.6x10^{-4}$[(b)]		IEA (2014)

[(a)]Landfill gas collected is flared in developing economies and recovered for energy production in developed economies.
[(b)]Considered low income countries (e.g. Africa region)
[(c)]Considered high income countries (e.g. OECD region)

In all tested scenarios, (Table 4.5) 4,000 Tons of commingled waste were considered. Scenario S1.1 is a baseline scenario describing the condition in developed economies locations whereby MSW is collected with 23% recovered for recycling and 12% biologically treated (composting). Incineration coupled with an energy recovery system (22%) is practiced more in developed economies particularly in regions with high land costs and low availability of land (World Bank, 2012). The remaining 43% of the waste is landfilled with landfill gas (LFG) collected (60%) for energy recovery. The impact of other biological treatment methods was tested in scenario S1.2 that substitutes composting in S1.1 by anaerobic digestion (12%) with energy recovery. Scenario S1.3 considers the impact of diverting the food waste (21%) through a FWD (at 75% market penetration and 95% of food waste ground)7 to the WW stream for aerobic treatment

[7] Note that this range represents upper values whereby the market penetration rate can range between 25 to 75% (Marashlian & El-Fadel, 2005; Galil & Yaacov, 2001) and the amount of food waste ground can range between 75 and 95 % (Wainberg et al., 2000; Marashlian & El-Fadel, 2005).

while the sludge is treated using anaerobic digestion. Scenario S1.4 and S1.5 consider landfilling or incineration of all the waste with energy recovery.

Scenario S2.1 represents the baseline scenario of general conditions in developing economies locations whereby MSW is collected with inefficient waste recycling (3%) or biological treatment such as composting (2%) even though the waste stream contains a high fraction of organic material. Developing economies still rely heavily on open dumping (17%) or poorly operated landfills (78%) with LFG occasionally flared (World Bank, 2012). Scenario S2.2 is an alternative for the baseline scenario that evaluates the potential of reducing the amount of waste landfilled or open dumped by optimizing recycling (13%) and composting (42%). The remaining 45% of the waste is landfilled with energy recovery. Scenario S2.3 substitutes composting in S2.2 by anaerobic digestion (42%) with energy recovery and scenario S2.4 considers the impact of diverting food waste through a FWD (at 75% market penetration and 95% of food waste ground) to the WW stream for aerobic treatment with sludge treatment using anaerobic digestion. Uncontrolled waste management practices, open dumping (S2.5) or burning (S2.6) of the waste (100%) were compared with landfilling (S2.7) or incineration (S2.8), respectively, both with energy recovery.

4.2.3 Impact of Input parameters

Model input parameters were first compared with values reported in the literature for consistency (Table 4.6). The parameter uncertainty was then examined through a sensitivity analysis whereby parameters were varied one at a time to assess their impact on emissions. These parameters included the fraction of the LFG collected, the methane correction factor (MCF) that differentiates between the waste disposal in landfills and dumpsites, the energy content of methane gas used in recovery systems, the efficiency factor of electricity generated from

landfilling, as well as emission factors related to direct N_2O from LFG flaring and upstream emissions from landfill construction (e.g. Material provision for liners and drainage systems). Several technical and operational criteria were selected to compare the developed model with commonly used models for emissions accounting including the 1996 and 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 1996; 2006); the Canadian Integrated Waste Management Model for Municipalities (IWM)[8] (EPIC & CSR 2004); the US EPA Waste Reduction Model (WARM)[9] (US EPA/ICF, 2016); the British Integrated Waste Management Model-2 (IWM-2)[10] (McDougall *et al.*, 2001); and the Entreprises pour l'Environnemnent (EpE)[11] (EpE, 2013). The above guidelines and models were selected for the comparative assessment based on their accessibility and their common use worldwide. Other privately-owned[12] models may exist and offer additional features in the context of GHG accounting of life cycle assessment (LCA), but to the best of our knowledge, none offers the capability of coupling solid waste and wastewater management systems.

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[8] Accepted by Environment Canada to evaluate the environmental performance of various elements of an ISWM system (EPIC and CSR, 2004; Mohareb *et al.*, 2008).

[9] Used to estimate emissions reductions in climate change impacts assessment but does not consider indirect upstream emissions (EPA/ICF, 2016).

[10] Based on a life cycle inventory of ISWM (McDougall *et al.* 2001).

[11] Used by European companies and local authorities for annual emissions inventories and accounts for gross and net direct emissions, as well as indirect (e.g. electricity consumption) and avoided emissions from energy and material recovery (EPE, 2013).

[12] Recent privately-owned models such as *EaseTech*, developed at the Technical University of Denmark (Clavreul *et al.*, 2014) or the Solid Waste Optimization Life-cycle Framework (SWOLF) model (Levis *et al.*, 2013) were not used in the comparative assessment because they are not endorsed by governmental agencies to be used for compliance purposes although they are useful models for waste management but not commonly reported for planning or decision making. In this study, the comparison targeted software supported or endorsed by international or governmental organizations, particularly for compliance or GHG emissions reduction purposes.

Table 4.5. Tested scenarios for assessing emissions from the waste sector under various management practices in a developed or developing economy

ID	**Description**	**Open dumping (%)**	**Landfilling (%)**	**Open Burning (%)**	**Incineration (%)**	**Recycling (%)**	**Composting (%)**	**Anaerobic Digestion (%)**	**FWD (%)**
Developed economy scenarios									
S1.1	Baseline condition		43		22	23	12		
S1.2	Substitute composting with AD		43		22	23		12	
S1.3	Substitute composting with FWD		34		22	23			21
S1.4	Landfilling with energy recovery		100						
S1.5	Incineration with energy recovery				100				
Developing economy scenarios									
S2.1	Baseline condition	17	78			3	2		
S2.2	Optimize recycling/composting		45			13	42		
S2.3	Substitute composting in S2.2 with AD		45			13		42	
S2.4	Substitute composting in S2.2 with FWD		45			13			42
S2.5	Open dumping	100							
S2.6	Open burning			100					
S2.7	Landfilling with energy recovery		100						
S2.8	Incineration with energy recovery				100				

Table 4.6. Model parameters in comparison to literature reported values

Parameter	Adopted averages	Reference
Fuel consumption for landfill construction	0.75 Liters/Ton of waste landfilled	0.5 to 1 Liters/Ton of waste landfilled (Manfredi *et al.*, 2009)
Fuel consumption for on-site daily operation	~2 Liters/Ton of waste landfilled ~3.28 Liters/Ton of waste composted	1 to 3 Liters of diesel/ Ton of waste landfilled (Manfredi *et al.*, 2009); 0.4 to 6 Liters of diesel/ Ton of waste composted (Boldrin *et al.*, 2009; EPA, 2006; Smith *et al.*, 2001), in most cases an average of 3 Liters/ Ton of waste composted is reported
Provision of diesel fuel	0.00045 $MTCO_2E$/ Liter of diesel	0.0004 to 0.0005 $MTCO_2E$/ L of fuel (Fruergaard *et al*,. 2009)
Fuel combustion	EF_{CO_2} (0.003 $MTCO_2E$/ L of diesel), EF_{CH_4} ($1.2x10^{-4}$ $MTCO_2E$/ L of fuel), EF_{N_2O} ($2.2x10^{-6}$ $MTCO_2E$/ L of fuel)	0.003 $MTCO_2E$/ L of diesel (Fruergaard *et al.*, 2009)
Provision of electricity	7 kWh/Ton of waste landfilled and 32 kWh/Ton of waste composted at 0.0005 $MTCO_2E$/ kWh	2 to 12 kWh/ Ton of waste landfilled (Manfredi *et al.*, 2009); 8 kWh/ Ton of waste landfilled and 32 kWh/ Ton of waste composted (McDougall, 2001) with 0.0001 to 0.0009 $MTCO_2E$/ kWh (Manfredi *et al.*, 2009) and 500 grams CO_2/kWh (IEA, 2014)
Energy content of Methane gas used for energy recovery	4,325 kWh / Ton of CH_4 generated	14,420 kWh/Ton (Friedrich and Trois, 2013a; Manfredi *et al.*, 2009); 4,325 kWh/ Tons of CH_4 generated (EPA/ICF 2016; EPA, 2013); 6,943 (EPIC & CSR, 2004; McDougal *et al*., 2001)
Efficiency factor of electricity generated from landfilling	0.85	0.85 (EPA/ICF 2016; EPA, 2013); 0.3 (EPIC & CSR, 2004; McDougal *et al*., 2001)
Provision of liner materials for construction of cells in the landfill	0.001 Ton of liner/ Ton of waste landfilled at 1.85 $MTCO_2E$/ Ton of material	1 kg liner / Ton of waste with EF for producing HDPE at 1.85 kg CO_2 / kg Liner (Manfredi *et al.*, 2009)
Provision of material for construction of the drainage system in the landfill	0.1 Ton of material/Ton of waste landfilled at $1.4x10^{-3}$ $MTCO_2E$/Ton of material	0.1 Ton of material/Ton of waste landfilled with 1.4 kg CO_2/Ton of material (Manfredi *et al*., 2009)
Carbon (biogenic) binding after 100 years	-0.16 $MTCO_2E$/ Ton of waste landfilled	-0.367 $MTCO_2E$/Ton of waste (EPA, 2006), -0.16 $MTCO_2E$/Ton of waste (Manfredi *et al*., 2009)
Carbon storage from the application of compost on land	-0.1 $MTCO_2E$/Ton of compost	-0.198 to -0.004 $MTCO_2E$/Ton of compost (Boldrin *et al*., 2009), -0.24 $MTCO_2E$/Ton of compost (EPA, 2006)
Avoided emissions from the use of compost as a substitute for peat production	-0.65 $MTCO_2E$/Ton of compost	-1.197 to -0.110 $MTCO_2E$/Ton of compost (Boldrin *et al*., 2009)
Fraction of LFG collected	0.75	Normalized LFG recovery rate of $126x10^{-6}$ Nm^3 CH_4 hr^{-1} Mg $waste^{-1}$ (Spokas *et al.*, 2015), 75% (EPA/ICF 2016), 17% (Itoiz *et al.*, 2013), 70% (Gentil *et al.*, 2010), and 20% (IPCC, 2006)

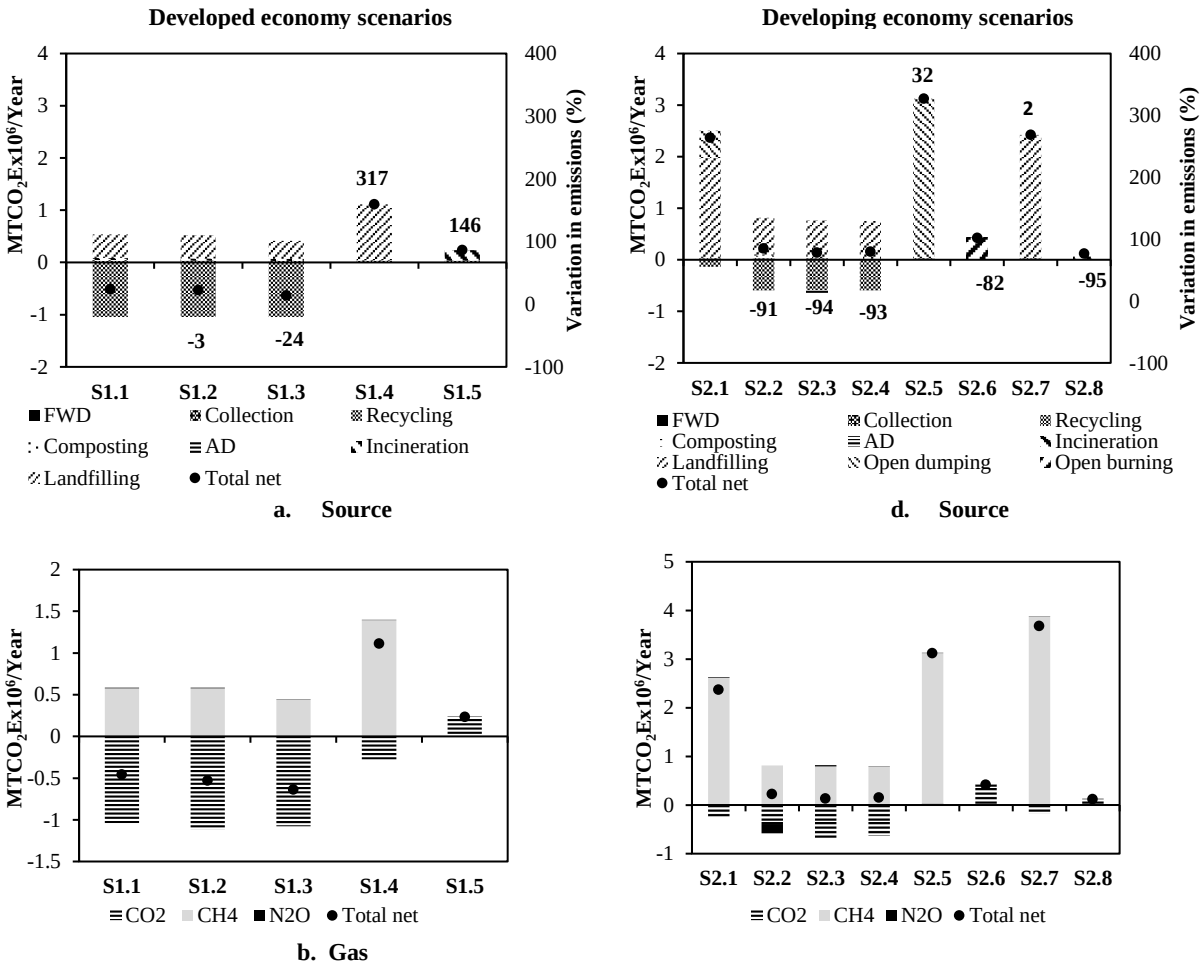
EF: Emission factor

4.3 Results and discussion

4.3.1 Model application

Figure 2 displays simulation results of various scenarios under developed (a, b, c) or developing economies (d, e, f) with corresponding contribution to total net emissions, categorized by source (e.g. FWD, collection, recycling, composting, AD, incineration, or landfilling), gas (e.g. CO_2, CH_4, N_2O), and type (e.g. direct-operating,

indirect-upstream, indirect-downstream), respectively. The variation in emissions for the tested scenarios (S1.2 to S1.5) with respect to the baseline condition in developed economies (S1.1) ranged from -24 to 317%, depending on the adopted waste management scheme (Figure 4.2.a). For instance, scenario S1.4 that considers landfilling of all MSW (100%) with energy recovery resulted with the highest emissions in comparison to the other tested scenarios mainly due to related methane emissions (Figure 4.2.b). Followed by scenario S1.5 that considers the incineration of all MSW with energy recovery (Figure 4.2.a). The adoption of a FWD policy (75% market penetration rate and 95% of food waste ground) in scenario S1.3 to divert food waste to an aerobic WW system with anaerobic digestion of sludge decreased total net emissions by ~24% with respect to baseline condition (Figure 4.2.a).



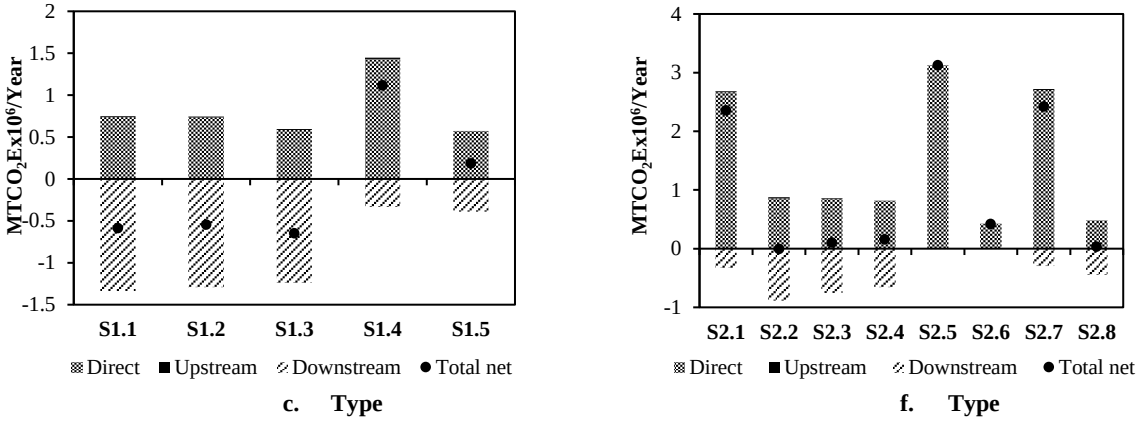


Figure 4.2. Comparison of management processes: Emissions by source, gas, and type

The variation in emissions is calculated with respect to the baseline scenario (S1.1 or S2.1) whereby
Variation in emissions is % = [(Old - New) / Old] x100
where Old = Net emissions from Baseline scenario (e.g. S1.1 or S2.1); and
New = Net emissions from alternative scenario (e.g. S1.2 or S2.2)

	Developed economy scenarios
S1.1	*Baseline condition*
S1.2	*Substitute composting with AD*
S1.3	*Substitute composting with FWD*
S1.4	*Landfilling with energy recovery*
S1.5	*Incineration with energy recovery*

	Developing economy scenarios
S2.1	*Baseline condition*
S2.2	*Optimize recycling/composting*
S2.3	*Substitute composting in S2.2 with AD*
S2.4	*Substitute composting in S2.2 with FWD*
S2.5	*Open dumping*
S2.6	*Open burning*
S2.7	*Landfilling with energy recovery*
S2.8	*Incineration with energy recovery*

Figure 4.3 depicts the disaggregation of emissions in developed and developing economies by individual management process (collection, recycling, composting, anaerobic digestion, incineration, and landfilling) with direct and indirect contributions. For instance, indirect upstream emissions consist of GHGs emitted during fuel provision, electricity provision, and material provision (e.g. during landfill construction); direct operational emissions consider emissions from waste degradation and fuel combustion; and indirect downstream emissions (or savings) are related to energy production, materials substitution (e.g. peat), management of residues, and carbon storage.

For the tested scenarios in developing economies (S2.2 to S2.8), variation in emissions with respect to the baseline condition ranged from -95 to 32%, depending on the adopted waste management scheme (Figure 4.2.d). Improper practices such as open burning of all the waste (S2.6) resulted in significantly higher total emissions (up to 295%) than incineration with energy recovery in S2.8 (Figure 4.2.d). Similarly, waste

disposal in the form of open dumping (S2.5) resulted with higher emissions (~ 30%) than controlled landfilling coupled with LFG collection with energy recovery in S2.7 (Figure 4.2.d). The perceived appearance that S2.6 (open burning) results in lower total emissions than baseline practices can be attributed mainly to emitting CO_2 with a lower GWP than CH_4 (Figure 2.e). However, the open burning of waste is a large source of pollutants other than GHGs, which are not included in emissions inventories for climate modeling (Wiedinmyer et al., 2014).

Alternative scenario S2.2 that optimizes the diversion of waste from landfilling through recycling (13%) and composting (42%) resulted in significant savings in emissions that reached up to 91% (Figure 4.2.d). Substituting waste composting with anaerobic digestion (42%) coupled with energy recovery in scenario S2.3 equally resulted with up to 94% savings in emissions. Similarly, diverting 42% of the waste to the WW system using FWDs in S2.4 resulted in 93% savings in emissions in comparison to baseline scenario S2.1 (Figure 4.2.d).

Consistent with literature findings (Blanco et al., 2014; Bogner et al., 2007; IPCC, 2014), the model simulations in both economies show that CH_4 emissions can be a major contributor (up to 99%) to total emissions from waste particularly from landfilling (Figure 4.2.b, e). It is also worth noting that the fraction of waste recovered for recycling (23%) in developed economies contributed to significant savings in emissions (Figure 3.b), thus, resulting in total net negative emissions (Figure 4.2.a). This is consistent with waste recovery from recycling in the US that reached about 26% (of the total MSW generated) in 2014 resulting in ~130% reduction in GHG emissions (EPA, 2016), thus contributing to total net negative emissions (EPA, 2017).

Naturally, the overall results are dependent on several variables such as waste composition, energy consumption, and efficiencies of waste technologies. For instance, the variation in waste composition in the context of developed and developing economies exhibited an appreciable impact on total emissions. Scenarios S1.5 (developed) resulted with up to 119% higher total net emissions than S2.8 (developing) whereby both scenarios consider the same amount of waste (4000 tons / day) collected for incineration (100%) with energy recovery (Figure 4.2.a, d). Direct emissions from waste incineration in developed economies were up to 21% higher than developing economies due to the presence of higher fractions of non-biomass combustible material (e.g. plastics, textiles, etc.) in the waste of developed economies (Figure 4.2.c). However, indirect downstream emissions from energy substitution were more significant for the case of developing economies (~14% higher) due to the variation in the electricity grid mix between developed and developing economies with the latter relying on coal for energy production (Figure 4.3.f).

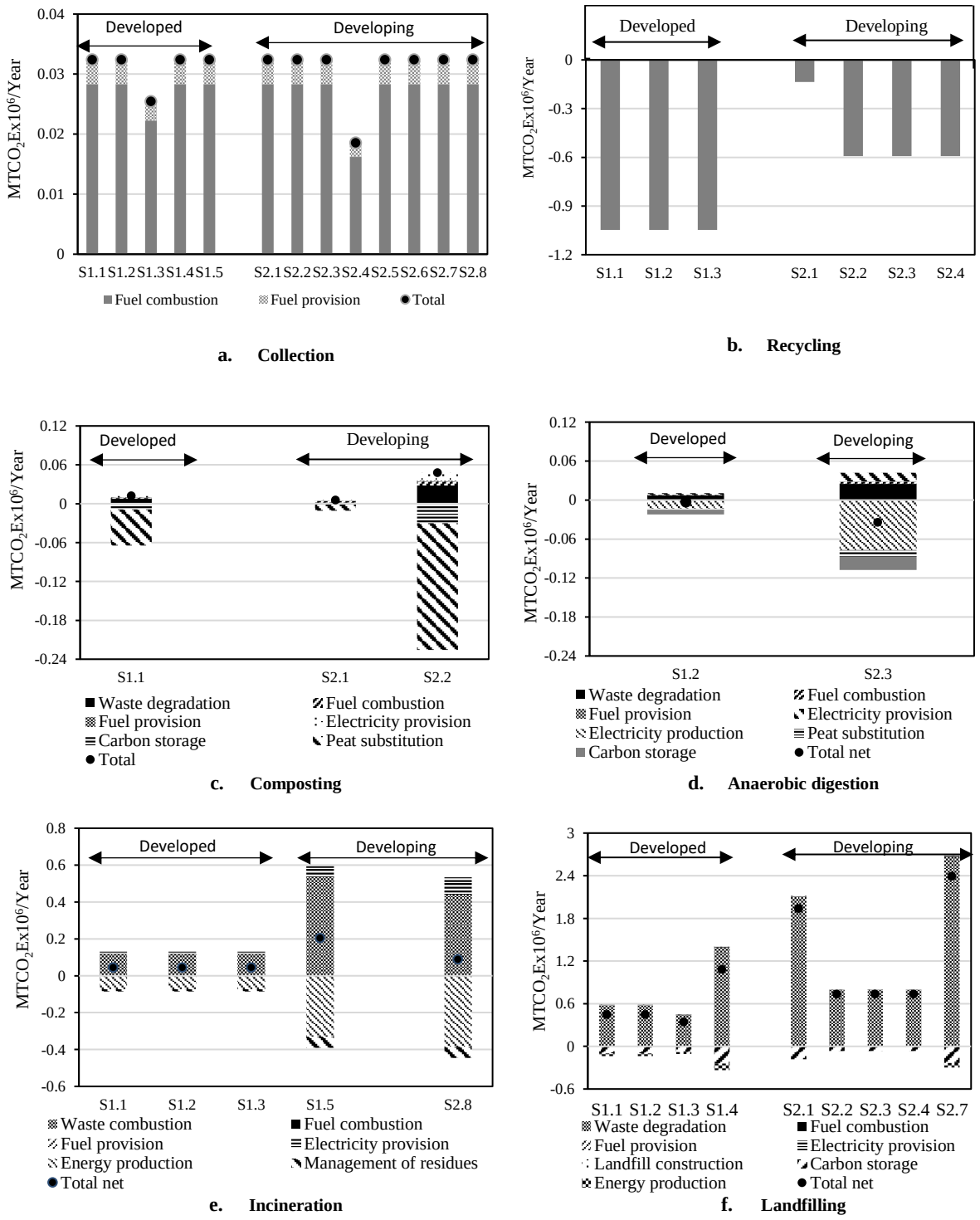


Figure 4.3. Emissions by source (waste management process)

	Developed economy		***Developing economy***
S1.1	*Baseline condition*	*S2.1*	*Baseline condition*
S1.2	*Substitute composting with AD*	*S2.2*	*Optimize recycling/composting*
S1.3	*Substitute composting with FWD*	*S2.3*	*Substitute composting in S2.2 with AD*
S1.4	*Landfilling with energy recovery*	*S2.4*	*Substitute composting in S2.2 with FWD*
S1.5	*Incineration with energy recovery*	*S2.5*	*Open dumping*
		S2.6	*Open burning*
		S2.7	*Landfilling with energy recovery*
		S2.8	*Incineration with energy recovery*

4.3.2 Impact of input parameters

Table 4.7 summarizes the impact of varying input parameters on emissions from the waste sector. The extent of input variation was defined based on default data reported in the literature. The results indicate that a LFG recovery system has a significant impact on total emissions, whereby an efficiency of 0.3 typical of developing economies (Banar *et al.*, 2009) compared to a 0.6 in developed economies (EPA/ICF, 2016) can cause a significant increase in the overall emissions reaching up to 84%. While accounting for N_2O emissions from flaring increased emissions by up to 11%, considering upstream emissions during site construction affected the overall emissions by ~1% only. On the other hand, a change in MCF[13] (methane correction factor) from 1 (corresponding to landfilling) to 0.4 (corresponding to open dumping) (IPCC, 2006), contributed to a ~54% decrease in emissions. Therefore, the efficiency of flaring and the MCF may contribute to a significant variation in the results, necessitating their inclusion in emission accounting models. Nonetheless, a decrease in the efficiency factor for electricity generated from 0.85 (EPA/ICF 2016; EPA, 2013) to 0.3 (EPIC & CSR, 2004; McDougal *et al.*, 2001) increased the overall emissions by only 5% because it is not very significant for the case of landfilling. A change in the energy content of LFG that is used for energy recovery (ER) from 4,325 (EPA/ICF 2016; EPA, 2013) to 14,420 kWh/ Tons of CH_4 (Friedrich and Trois, 2013a; Manfredi *et al.*, 2009) contributed to a reduction in emissions reaching up to 17% (Table 7).

[13] In unmanaged disposal sites, a larger fraction of waste decomposes aerobically in the top layer producing less CH_4 than anaerobically managed sites (IPCC, 2006).

Table 4.7. Sensitivity to key input parameters

Parameter	Value	Developed-S1.1	Comments
Fraction of LFG collected	0.6	-497,893	Collection efficiency of (60 %) for a typical operating landfill with wet waste (EPA/ICF, 2016) and 31% (Banar *et al.* 2009) usually in developing economies.
	0.31	-81,466	
	% Change	**+84%**	*% change from considering 0.6 to 0.3*
LFG capture system for flaring or ER	ER	-497,893	The model accounts for N_2O emissions from flaring, in contrast to other models such as WARM and IPCC-2006 that consider flared LFG as biogenic and not counted.
	Flaring	-449,614	
	% Change	**+11%**	*% change from considering ER to flaring*
EF for Landfill construction	With	-449,614	Emission factors for landfill construction are adapted from (Manfredi *et al.*, 2009).
	Without	-500,826	
	% Change	**-1%**	*% change from considering EF for landfill construction to not considering it*
MCF	1	-449,614	MCF=1 for landfilling and MCF= 0.4 assuming an unmanaged shallow dumpsite; adapted from IPCC 2006 Guidelines.
	0.4	-1,022,836	
	% Change	**-54%**	*% change from considering landfilling to open dumping*
Efficiency factor for electricity generated from landfilling	0.85	-449,614	0.85 (EPA/ICF 2016; EPA, 2013); 0.3 (EPIC & CSR, 2004; McDougal *et al.*, 2001).
	0.3	-473,781	
	% Change	**+5%**	*% change from considering 0.85 to 0.3*
Energy content of methane gas	4,325	-449,614	The energy content of methane gas used for ER expressed in kWh/ Tons of CH_4 generated is 4,325 (EPA/ICF 2016; EPA, 2013) and 14,420 from (Friedrich & Trois, 2013a; Manfredi *et al.*, 2009).
	14,420	-623,657	
	% Change	**-17%**	*% change from considering 4,325 to 14,420*

Developed-S1.1: Recycling (23%) / Composting (12%) / Landfilling (43%) / Incineration (22%)
The percent change is calculated with respect to the total emissions (expressed in $MTCO_2E$/Year)
ER: Energy recovery; **MCF:** Methane correction factor; **EF:** Emission factor

4.3.3 Comparative advantages

Table 4.8 compares the developed model with commonly reported models for emissions accounting using key parameters and assumptions affecting emissions from the waste sector. Except for the IPCC methods, all models targeted developed economies with default input data introduced for specific locations and often with uncertainty about emission factors that are not readily accessible or adjustable (Assamoi and Lawryshyn, 2012; Laurent et al., 2014). The developed model can accommodate general and specific locations equally with input data from both developed and developing economies defined more explicitly all while offering users the flexibility of modifying emission factors and input parameters in contrast to a closed source code. For instance, the developed model allows the user to adjust waste input parameters while examining their impact on uncertainty in model simulations. It also allows the selection of the energy mix with associated emissions of CO_2 by kWh, in addition to providing default averages.

Disaggregation of emissions based on direct vs. indirect or upstream vs. downstream contributions, is imperative to assess total net emissions from individual waste management processes and compare across them. In the developed model, emissions can be disaggregated, which allows users to select the scope of reporting whether for national inventorying (direct emissions) or for life cycle assessment (LCA) (direct and indirect emissions) decision-making and planning purposes. While existing GHG accounting models consider many direct and indirect contributions, most of them (except for EpE) do not consider emissions by type (direct vs. indirect) with limitations at the level of neglecting upstream[14] and downstream boundaries (Table 4.8). Similarly, while the WARM model accounts for offset of CO_2 emissions from fertilizer and peat production or carbon storage from land application of compost produced from biological treatment, and savings from carbon storage during landfilling, other models neglect downstream contributions. In addition, existing models do not account for indirect emissions associated with the management of residues from waste incineration (savings from material recovery and load from bottom ash landfilling), as well as indirect emissions related to the construction of a landfill, albeit it is relatively not as significant.

The developed model also encompasses the ability to simulate a wider range of emissions from waste management processes. Similar to many existing models, it includes life cycle stages from waste collection to landfilling (Table 4.8). Furthermore, it considers processes neglected by some models such as, open dumping or burning of waste, as well as N_2O emissions from flaring of LFG, which was demonstrated to affect total net emissions significantly (Table 4.8). We emphasize that existing models used in the comparative assessment (Table 4.8) were selected based on their accessibility and their common use worldwide. Other privately-owned

[14] Upstream emissions such as emissions from fuel provision (extraction, processing, storage, and transport of fuel), which is required for all waste management processes involving fuel consumption for the operation of equipment or collection/transport of waste

models[15] may exist and offer additional features in the context of GHG accounting of LCA, but to our knowledge, none offers the capability of coupling solid waste and wastewater management systems.

On the other hand, while introducing a FWD policy was reported in many cases to be an effective alternative for waste reduction (Maalouf and El-Fadel, 2017; Marashlian and El-Fadel, 2005; Yi and Yoo, 2014; Bernstad et al., 2013), existing emissions' accounting models were not designed to assess the impact of such a policy on emissions' inventory from the coupling of MSW and WW management systems. The developed model integrates both systems under a single framework to evaluate alternatives that can be particularly important in the context of developing economies because of the corresponding waste composition with more than 60% food waste.

Last but not least, a significant discrepancy amongst all models is apparent in using waste and gas categories for the composition and type of emissions. In the developed model, emissions were disaggregated based on both categories, and integrated to clarify the overall reported emissions, as well as to facilitate the assessment of the impact of waste composition on emissions. This allowed the comparison between developed and developing economies and between reported results from other models. Equally important is the uniformity in deriving and applying emission factors in quantifying emissions from the waste management, which can have serious implications on carbon credit trading, namely when assessing emissions mitigation or reduction targets under the United Nations Framework Convention on Climate Change (UNFCCC) agreements or when implementing Clean Development Mechanism (CDM) projects.

[15] Recent privately-owned models such as *EaseTech*, developed at the Technical University of Denmark (Clavreul *et al.*, 2014) or the Solid Waste Optimization Life-cycle Framework (SWOLF) model (Levis *et al.*, 2013) were not used in the comparative assessment because they are not endorsed by governmental agencies to be used for compliance purposes although they are useful models for waste management but not commonly reported for planning or decision making. In this study, the comparison targeted software supported or endorsed by international or governmental organizations, particularly for compliance or GHG emissions reduction purposes.

Table 4.8. Comparison with existing models

		IPCC 1996 & 2006	WARM	EpE	IWM	IWM-2	Developed Model
Geographical scope		Worldwide	US	EU	Canada	UK	Worldwide
Database		Default	Default	User selected[(a)]	Default	Default	User selected[(b)]
Modifiable/ dynamic		N	N	Y	N	N	Y
Select emissions by type[I]		N	N	Y	N	N	Y
Select EF/input parameter		N	N	Y	N	N	Y
Select by gas type		Y	N	Y	Y	Y	Y
GWP_{100} Reference		SAR (1995)	AR4 (2007)	AR4 (2007)	SAR (1995)	SAR (1995)	User selected
Emissions		CO_2,CH_4,N_2O	CO_2,CH_4,N_2O	CO_2,CH_4,N_2O	Variable[(d)]	Variable[(d)]	CO_2,CH_4,N_2O
Waste composition		F, P,P_L,T,W,GA,N,O	F,P,P_L,T,W,GA,G,M,O[(e)]	Aggregated	F,P,P_L,GA,G,M,O	F,P,P_L,T,G,M,O	F,P,PL,T,W,GA,G,M,N,O
MSW management processes		Co,AD,I,Lf,OD,OB	C,R,Co,AD,I,Lf	C,R,Co,AD,I,Lf	C,R,Co,AD,I,Lf	C,R,Co,AD,I,Lf	C,R,Co,AD,I,Lf,OD,OB
WW and S management systems		N	N	N	N	N	Y
Collection/transport	*Fuel combustion*	N	Y	Y	Y	Y	Y
	Fuel provision	N	N	N	N	N	Y
Biological treatment	*Waste degradation*	Y	Y	Y	Y	Y	Y
	Fuel combustion	N	Y	Y	Y	Y	Y
	Electricity consumption	N	N	Y	Y	Y	Y
	Fuel provision	N	N	N	N	N	Y
	Carbon storage	N	Y	N	N	N	Y
	Peat substitution	N	Y	N	N	N	Y
	Energy recovery	N	Y	N	N	Y	Y
Incineration process	*Waste combustion*	Y	Y	Y	Y	Y	Y
	Electricity consumption	N	N	Y	N	N	Y
	Energy recovery	N	Y	Y	Y	Y	Y
	Material recovery	N	N	N	N	N	Y
Landfill processes	*Waste degradation*	Y	Y	Y	Y	Y	Y
	Fuel combustion	N	Y	Y	Y	Y	Y
	Electricity consumption	N	N	Y	Y	Y	Y
	Fuel provision	N	N	N	N	N	Y
	Material provision	N	N	N	N	N	Y
	Carbon storage	N	Y	N	N	N	Y
	Energy recovery/	N/	Y/	Y/	Y/	Y/	Y/
	N_2O from flaring	N	N	N	Y	N	Y
Assessments	*Carbon Credit*	N	N	N	N	N	Y

IPCC 1996 & 2006: Intergovernmental Panel on Climate Change 1996 and 2006 Guidelines; **WARM:** Waste Reduction Model; **EpE:** Entreprises pour l'Environnemnent; **IWM**: Integrated Waste Management Model for municipalities; **IWM-2:** Integrated Waste Management Model-2.

MSW: Municipal solid waste; **WW:** Wastewater**; S:** Sludge**; LC:** Life cycle emissions (include direct and indirect emissions); **EFs:** Emission factors; **F**: Food; **P**: Paper; **P_L**: Plastics; **T**: Textiles; **GA**: Garden; **W**: Wood; **N**: Nappies; **G**: Glass; **M**: Metals; **O**: others; **C:** Collection, **R:** Recycling, **Co:** Composting, **AD:** Anaerobic Digestion, **I:** Incineration, **LF:** Landfilling, **OD:** Open Dumping, **OB:** Open Burning.

[(a)] In order to calculate direct emissions from waste degradation in landfills, the user selects a common methodology and refers to the regulatory methodologies recommended by the authorities of the country where the site is located.

[(b)] User has the ability to disaggregate emissions based on scope of reporting whether for national / GHG inventorying or for LCA / planning and decision-making purposes.

[(c)] Type of emissions: upstream, direct-operational, and downstream contributions (direct and indirect)

[(d)] Includes main greenhouse gases: CO_2, CH_4, N_2O emissions as well as other emissions such as CO, NOx, SOx, PM, HCl, HF, H_2S, Dioxins/Furans, NH_3, As, Cd, Cr, Cu, Lead, Mn, Hg, Ni, Zn.

[(e)] In its last release, WARM (v. 14) included 54 materials, products and mixed categories (EPA/ICF, 2016)

4.4 Conclusion

This chapter introduced a comprehensive model developed for decision support in quantifying and improving emissions estimation from integrated waste and wastewater/sludge management systems while considering upstream, operating, and downstream processes. The model provides guidelines towards a credible national inventory as well as a policy planning and decision making about process viability for investing in carbon credit. In addition to the current state of practice in developed economies, the model included emissions from waste management processes still practiced in developling economies (such as open dumping, open burning of waste, and poorly operated landfills with flaring systems as well as auxiliary fuel needed to satisfy the LHV during waste incineration) commonly not considered in most LCA models. It can disaggregate emissions by source (waste processes from collection to final disposal), or type (direct and indirect), or gas (CH_4, CO_2, N_2O) and offers users the flexibility to select processes or modify input parameters while examining their impact on uncertainty in model simulations. Equally important is a clarity in deriving and applying emission factors used to quantify emissions from waste management systems, which can have serious implications on carbon credit trading when assessing emissions mitigation or reduction targets under the UNFCCC agreements or when implementing CDM projects.

The model was tested in the context of both developed and developing economies to assess the impact of waste composition, management processes, energy consumption and other input parameters on variations in emissions. A scenario analysis demonstrated that best practices through recycling, biological treatment, food waste diversion, and / or energy recovery can contribute to significant savings in emissions that ranged between 24 and 95%, depending on the tested system. In contrast, improper waste

management such as open dumping or burning instead of controlled landfilling or incineration (with energy recovery) can increase the total equivalent emissions by ~30% and ~295%, respectively. In closure, we argue the benefits of the model application in providing guidelines for policy planning and decision making about process viability for investing in carbon credit.

CHAPTER 5

EFFECT OF A FOOD WASTE DISPOSER POLICY ON SOLID WASTE AND WASTEWATER MANAGEMENT WITH ECONOMIC IMPLICATIONS OF ENVIRONMENTAL EXTERNALITIES

5.1 Introduction

Population growth and development coupled with limited land resources in urban areas have brought about challenges for decision makers to manage continuously increasing quantities of municipal solid waste (MSW). In developing economies in particular, MSW is characterized with a high organic fraction in excess of 60% compared to less than 30% in developed economies (World Bank, 2012; IPCC, 2006). This fraction can be diverted from the waste collection system by introducing a food waste disposer (FWD) at the household level, which direct the food waste stream towards the wastewater collection and management system (Iacovidou et al., 2012a). While effective in reducing the amount of MSW to be managed, FWDs remain controversial because of associated impacts related in particular to the generation of greater and stronger volumes of wastewater and sludge to be managed in addition to increased energy and water consumption, thus requiring scrutiny when proposed to minimize waste sorting or landfilling (Marashlian and El-Fadel 2005). The increase in wastewater biochemical oxygen demand (BOD), suspended solids (SS), and other nutrients due to the use of FWDs contribute to an increase in emissions during wastewater management, coupled with an increase in energy consumption and sludge generation for ultimate treatment and disposal contributing also to an increase in emissions (Iacovidou et al., 2012a).

Past efforts targeted the operational and feasibility of introducing a FWD policy into the MSW management system, with some work reporting on the effect of such a policy on the net carbon footprint (Table 5.1). The latter consists of the net emissions generated from the diversion of food waste to the wastewater and sludge treatment systems.

On the other hand, while several emissions' accounting models have been developed [such as IWM (EPIC and CSR, 2004); WARM (EPA/ICF, 2016); SIWMS (Hanandeh and El Zein, 2010); EASTECH (Clavreul et al., 2014); EpE tool (EpE, 2013); IWM-2 (McDougall et al., 2001); CO2ZW tool (Itoiz et al., 2013), and the 2006 and 1996 IPCC models for National Greenhouse Gas Inventories], none was designed to assess the impact on emissions' inventory upon introducing a FWD policy into the MSW and wastewater (WW) management systems.

Table 5.1. Selected studies assessing the FWD system

Reference	**Impact coverage**	**Reported impact of FWD**
Battistoni et al. (2007)	Operational and economic	Positive
Bernstad (2013)	Operational	Positive
Bernstad Saraiva (2016)	Carbon footprint and energy	Positive
Bolzonella et al. (2003)	Operational	Negative/Positive
CECED (2003)	Operational	Negative
Constantinou (2007)	Operational and economic	Negative
De Koning and Van der Graaf (1996)	Operational and economic	Positive
Diggelman and Ham (2003)	Environmental and economic	Positive
Evans (2007)	Environmental and economic	Positive
Evans (2012)	Environmental and economic	Positive
Galil and Yaacov (2001)	Operational and economic	Negative/Positive
Iacovidou et al. (2012a)	Operational and environmental	Positive
Iacovidou et al. (2012b)	Operational and environmental	Positive
Jones (1990)	Operational	Positive
Kim et al. (2011)	Economic	Positive
Lundie and Peters (2005)	Environmental	Positive
Marashlian and El-Fadel (2005)	Operational and economic	Positive
Nilsson et al. (1990)	Operational	Negative
Raunkjaer et al. (1995)	Operational	Positive
Yi and Yoo (2014)	Environmental and economic	Positive
Wainberg et al. (2000)	Operational and economic	Positive

This chapter integrates these systems under a single framework model developed to evaluate the carbon footprint of introducing FWDs to reduce waste processing in the context of developing economies where the food waste fraction exceeds 60%. The results were then compared with a developed economy region with a lower food waste fraction of 30%. The analysis was conducted while considering the economics of environmental externalities with a focus on sludge management and net emissions for potential carbon trading.

5.2 Materials and methods

5.2.1 Theoretical framework

Figure 5.1 illustrates the model's framework linking the MSW and WW management systems upon the introduction of a FWD for grinding food waste and discharging it with the WW stream. The WW management system may consist of aerobic or anaerobic processes with several sludge management (SM) options including anaerobic digestion, composting, landfilling, incineration, or land application. On the other hand, the MSW management system include collection, transport, recycling, composting, anaerobic digestion, incineration / open burning, landfilling / open dumping1. The assessment targeted emissions, materials recovery (e.g. recyclables), by-products (e.g. compost), economic and environmental externalities, as well as energy produced or consumed across various stages. The model accounts for direct operational emissions arising from systems' operations such as onsite equipment and waste degradation, as well as indirect upstream emissions (inputs of energy and material) and indirect downstream emissions (such as savings related to energy and material substitution as well as carbon storage). Emissions are estimated in Metric Tons of CO_2 equivalent ($MTCO_2E$) with carbon dioxide (CO_2) having a 100-year global warming

potential (GWP_{100}) of 1 as a reference, CO_2 biogenic of 0, methane (CH_4) and nitrous oxide (N_2O) of 34 and 298, respectively (IPCC, 2013).

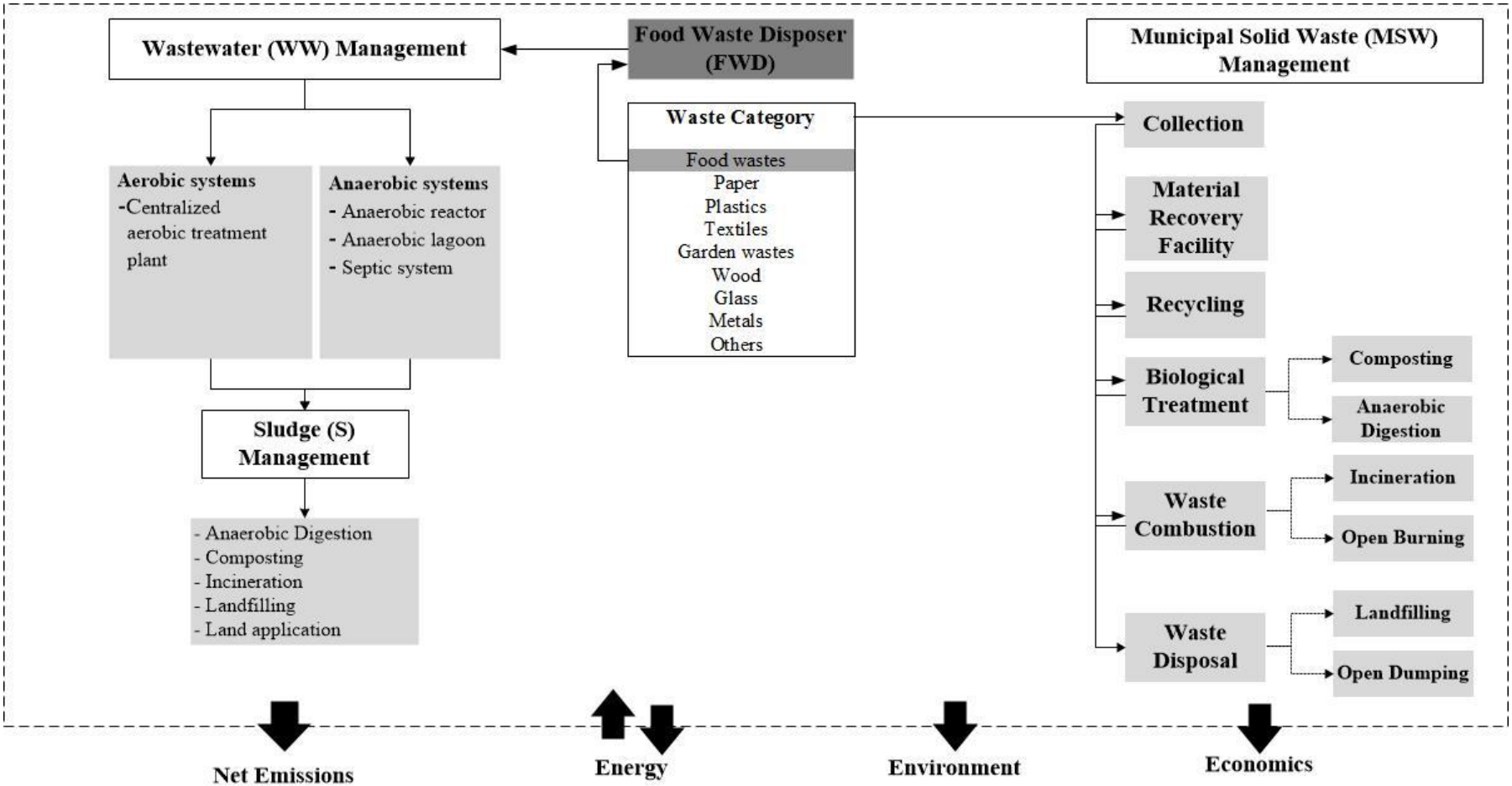


Figure 5.1. Model Framework

5.2.2 Scenario Definition: Policy and Economic Analysis

The carbon footprint of introducing a FWD policy was examined in the context of developing economies characterized with a high food waste content (Table 5.2) with the objective to discern viable waste management scenarios with considerations to the economics of the main environmental externalities (i.e. sludge management) while targeting minimal landfilling and emissions' reduction for potential carbon credit trading. The tested scenarios encompassed several variables including (1) FWD market penetration rate of ~75%; (2) amount of food waste grinded at the household level of ~95%; (3) wastewater treatment (aerobic and anaerobic processes) and sludge management alternatives (anaerobic digestion, composting, incineration, or landfilling); and (4) upstream, operating-direct, and downstream (direct and indirect) emissions. A quantity of 4,000 Tons / day of commingled MSW collected by a fleet of diesel-powered vehicles16 were considered with the waste composition presented in Table 5.2. The main scenarios with a waste composition associated with a typical developing economy characterized with a high food fraction in excess of 60% were then compared to a typical developed economy waste with less food waste of less than 30%. The analysis also considered variations in waste collected under several scenarios (Table 5.3) with the food waste diverted from the MSW management to the WW system. Note that many other combinations of scenarios can be tested and only a few were selected for illustrative purposes. The scenarios that were tested are:

[16] Using an average of ~6 Liters/Ton of waste as the overall diesel fuel consumption per Ton of municipal solid waste collected and transported (adapted from Chen and Lin, 2008; Friedrich and Trois, 2013).

Baseline Scenario 1 (SB.1): Collection / landfilling. This scenario considers that all MSW is collected and landfilled (100%) with energy recovery from landfill gas (LFG).

Baseline Scenario 2 (SB.2): Collection / Recycling / Composting / Landfilling. This scenario evaluates the potential to reduce the amount of MSW that is landfilled by recycling and composting instead of a FWD. Materials such as wood, paper, glass, metal, plastic, and textile are recovered for recycling (10%) and food waste fraction is composted aerobically (42%). The remaining waste stream (48%) is collected and landfilled.

Baseline scenario 3 (SB.3): Collection / recycling / anaerobic digestion / landfilling. This scenario also evaluates the potential to reduce the amount of MSW that is landfilled by recycling and anaerobic digestion instead of a FWD. Materials such as wood, paper, glass, metal, plastic, and textile are recovered for recycling (10%) and food waste fraction is digested anaerobically (42%) with energy recovery. The remaining waste stream (48%) is collected and landfilled with energy recovery.

Alternative Scenario 1 (SA.1): Collection / landfilling / aerobic wastewater treatment / anaerobic digestion of sludge. This scenario is an alternative for the baseline scenario SB.1 that considers the integration of a FWD for grinding food waste (42%) and discharging it with the WW stream for aerobic treatment while the sludge is treated using anaerobic digestion. The remaining waste stream (58%) is collected and landfilled.

Alternative Scenario 2 (SA.2): Collection / landfilling / anaerobic wastewater treatment / composting of sludge. This scenario is another alternative for baseline scenario SB.1 that considers the integration of a FWD for grinding food waste (42%) and discharging it with the WW stream for anaerobic treatment with energy recovery while

the sludge is composted aerobically. The remaining waste stream (58%) is collected and landfilled.

Alternative Scenario 3 (SA.3): Collection / recycling/ landfilling / aerobic wastewater treatment / anaerobic digestion of sludge. This scenario is an alternative for baseline scenarios SB.2 and SB.3 that considers the integration of a FWD for grinding food waste (42%) and discharging it with the WW stream for aerobic treatment while the sludge is treated using anaerobic digestion with energy recovery. The remaining waste stream (58%) is collected and landfilled with energy recovery.

Table 5.2. Average MSW composition (%)

(Adapted from IPCC, 2006; World Bank, 2012)

Waste category	**Developing economy**	**Developed economy**
Food	60	30
Metals	3	6
Plastics	8	11
Papers	5	31
Glass	3	7
Wood	6	5
Textiles	2	3
Others	13	7
Total	*100*	*100*

The main scenarios were first tested with a waste composition typical of a developing economy characterized with a high food fraction of 60%, and then compared to a developed economy with less food waste of < 30%.

Table 5.3. Tested Scenarios

	Baseline scenarios: without a FWD policy		*Alternative scenarios: with a FWD policy*
SB.1	***Generated*** 4000 T/d → ***Collection*** 4000 T/d → ***Landfilling*** 4000 T/d	**SA.1**	***Generated*** 4000 T/d → ***FWD*** 1680 T/d → ***Collection*** 2320 T/d → ***Landfilling*** 2320 T/d ***WWT (Aerobic process)*** → ***ST (AD)***
		SA.2	***Generated*** 4000 T/d → ***FWD*** 1680 T/d → ***Collection*** 2320 T/d → ***Landfilling*** 2320 T/d ***WWT (Anaerobic process)*** → ***ST (Composting)***
SB.2	***Generated*** 4000 T/d → ***Collection*** 4000 T/d → ***Recycling*** 400 T/d → ***Composting*** 1680 T/d → ***Landfilling*** 1920 T/d	**SA.3**	***Generated*** 4000 T/d → ***FWD*** 1680 T/d → ***Collection*** 2320 T/d → ***Recycling*** 400 T/d → ***Landfilling*** 1920 T/d ***WWT (Aerobic process)*** → ***ST (AD)***
SB.3	***Generated*** 4000 T/d → ***Collection*** 4000 T/d → ***Recycling*** 400 T/d → ***AD*** 1680 T/d → ***Landfilling*** 1920 T/d		

Notes

A fraction of 0.7 of food waste (equivalent to 42% of the total waste generated) is recovered for composting, anaerobic digestion, or diverted through the FWD (at ~75% market penetration and 95% of food waste grinded).

Recycling considered: wood, paper, glass, metal, plastic, and textile.

FWD: Food waste disposer; **AD:** Anaerobic digestion; **WWT:** Wastewater treatment (aerobic or anaerobic process); **ST:** Sludge treatment (composting, AD, incineration, or landfilling)

Economic associations targeted the assessment of conventional and environmental costs and savings for the tested scenarios. The cost of MSW management is estimated by multiplying the average costs (US$ per ton) of alternatives (Table 5.4) by the total amount of waste managed. The cost of introducing FWDs includes (1) capital/operating costs, (2) costs of managing additional wastewater and sludge loads, and (3) the cost of increased consumption of domestic water for grinding the food waste (Table 5.5) with electricity cost for operation of FWDs being minimal (Iacovidou et al., 2012a; Marashlian and El-Fadel, 2005). Environmental savings comprised costs forgone due to the decrease in requirements for managing food wastes diverted from the waste stream (e.g. leachate and gas management) (Table 5.5). Note that inflation was not considered, and values were taken at a constant year zero.

Table 5.4. Average cost of MSW management (US$/tonne)

	Collection	Sorting	Composting	Anaerobic digestion	Landfilling	Landfilling with energy recovery
Assamoi and Lawryshyn (2012)	…	…	…	…	18	…
Bianchini and Hewage (2012)	…	…	…	…	56	…
Damgaard et al. (2011)	…	…	…	…	70	67
Dijkgraaf and Vollebergh (2004)	…	…	…	…	45	…
EC (2002a)	…	…	…	80	62	58
Jamasb and Nepal (2010)	…	26	…	…	15	13
Kim et al. (2011)	61	…	…	…	10	…
NREL (2013)	…	…	…	34-90	…	…
Rabl et al. (2008)	…	…	…	…	45	40
Tsilemou & Panagiotakopoulos (2006)	…	…	17-73	22-67	12-50	…
World bank (2012)	20-250[a]	….	5-90	20-150	10-100	…
Wrap (2016)	…	28[b]	27	44	21	…
Range (US$/tonne)	20-250	26-28	5-90	20-150	10-100	13-67
Average (US$/tonne)	**135**	**27**	**47**[c]	**85**[d]	**72**[e]	**57**[e]

[a] Collection includes pick up, transfer, and transport to final disposal site for residential and non-residential waste.
[b] Cost of sorting of four waste categories or more that are delivered as comingled MSW to the material recovery facility (MRF).
[c] Composting excludes sale of finished compost (which ranges from 0 to 100 US$/tonne).
[d] Anaerobic digestion includes sale of energy from methane and excludes cost of residue sale and disposal.
[e] Includes an additional ~17 US$/tonne of waste for onsite leachate and gas collection, treatment and management (EC, 2002a; Damgaard et al., 2011).

Table 5.5. Unit average costs and savings

	Costs/savings
Capital and operating cost of FWD units[(a)]	222 US$/tonne of food waste treated/yr or 40 US$/Unit/yr (400 US$ /Unit/10 years)
Conventional cost of secondary wastewater treatment for added volume[(b)]	1.5 US$/tonne of food waste treated or 125 US$/tonne of BOD
Conventional cost of sludge treatment for added volume[(c)]	Land-spreading: 117-170 US$/Dry tonne (Wendland, 2016); Land-spreading of composted sludge/use in land reclamation: 223-265 US$/Dry tonne (Wendland, 2016); Landfilling/incineration: 276-371 US$/Dry tonne (Wendland, 2016); Agricultural use: 58 US$/Dry tonne (Lundin et al., 2004); Anaerobic digestion: 881 US$/Dry tonne (Murray et al., 2008); Range by Marashlian and El-Fadel (2005): 39-292 US$/Dry tonne; Range by Milieu Ltd, WRc and PRA (2010): 215-460 US$/Dry tonne; Range by Murray et al. (2008): 39-2,838 US$/Dry tonne *Adapted Cost (US$/Dry tonne): Min: 39; Max: 2,838; Average: 244*
Cost of increased domestic water consumption for grinding food waste[(d)]	27,000 L/tonne 0.32 US$/$m^3$ 8.6 US$/ tonne of food waste treated
Environmental cost[(e)]	Equivalent to 15% of conventional cost
Environmental savings [(f)]	~17 US$/tonnes/year of foregone leachate and gas management

[(a)] *Cost=tonnes of food waste grinded x 222 US$/tonne of food waste treated/yr;* assuming a capital cost of 208 US$/tonne of waste treated/ yr (with an average cost per unit of US$ 400 with and expected life span of 10 years) and operating cost of 13.8 US$/tonne/yr (Yi andYoo, 2014); Or *Cost=% market penetration x [Population/ (capita/household)] x 40 US$/Unit/yr;*

[(b)] *Cost=tonnes of waste treated per year x 1.5 US$/tonne of food waste treated* (average cost adopted from Yi & Yoo (2014) regardless of the size of the wastewater treatment plant, which is a limitation because it is commonly recognized that the cost is affected by the plant size); *Or Cost=tonnes of BOD per year x 125 US$/tonne of BOD* **(**Average cost of common technologies (Marashlian and El-Fadel, 2005));

[(c)] *Cost=tonnes of sludge per year x average cost of most selected sludge management method (US$/Dry tonne)*. The range adopted by Marashlian and El-Fadel (2005) considered technologies such as: centrifuge thickening and dewatering; belt filter press; composting; recessed-plate filter; aerobic digestion; anaerobic digestion; alkaline stabilization; thermal aerobic pre-treatment and anaerobic digestion; pre-pasteurization and anaerobic digestion; reactor composting; anaerobic digestion and thermal drying; and incineration. The selected technologies by Milieu Ltd, WRc and PRA (2010) include: incineration, landfilling, recycling, and land spreading of composted and digested sludge. Technologies considered by Murray et al. (2008) are: dewatering, lime stabilization, aerobic digestion, heat drying/compost, heat drying, anaerobic /heat drying, heat drying/aerobic, FBC incineration (natural gas or coal);

[(d)] *Cost=Volume of water needed to grind food waste (m^3/year) x 2.2% x 4 US$ /m^3* (note that the amount of water is negligible it only represents 2.2% of the total household water use and the average domestic water charging rate is 4 US$ /m^3 corresponding to annual consumption of 200 m^3 per year adapted from OECD, 2015) Or 8.6 US$/ tonne of food waste treated (Yi and Yoo, 2014);

[(e)] The environmental impacts include externalities associated with air emissions (e.g. energy recovery) but does not consider other potential impacts on health or soil/water pollution. They were set at 15% of conventional costs (EC, 2002b; Milieu Ltd, WRc and PRA, 2010; Marashlian and El-Fadel, 2005);

[(f)]Include cost forgone from leachate and gas management ~17 US$/tonnes/year (EC, 2002a; Damgaard et al., 2011) and cost forgone of abating pollutant discharge from management of food wastes.

The offset of emissions was quantified based on the carbon market price ranging from 0.1 to 44.8 US$/$MTCO_2E$ in 2015, with an average price of 3.3 US$/$MTCO_2E$, which is the lowest reported market value from voluntary actors since 2006 (Ecosystem Marketplace, 2016). This average is adopted to assess associated benefits and allow cost savings estimation for reducing the carbon footprint through regulated and voluntary global markets for trading or offsetting of carbon credits (El-Fadel et al., 2013). Finally, a sensitivity analysis was conducted whereby the cost of several processes (collection, composting, landfilling with energy recovery, and sludge treatment) were varied one at a time to assess their impacts on the net cost

variation. These costs were tested because the corresponding processes are reported to have a wide range in cost depending on location (Tables Table 5.4 and Table 5.5).

5.3 Results and discussion

5.3.1 Emissions Implications

The adoption of a FWD policy affected emissions depending on the tested scenario for waste and wastewater / sludge management with variation from -42 to -10% with respect to baseline scenarios (Figure 5.2). The comparison in emissions between alternative scenarios (SA.1, SA.2, and SA.3) using FWDs and baseline scenarios SB.1 (landfilling), SB.2 (composting), and SB.3 (anaerobic digestion) is depicted in Figure 5.2.

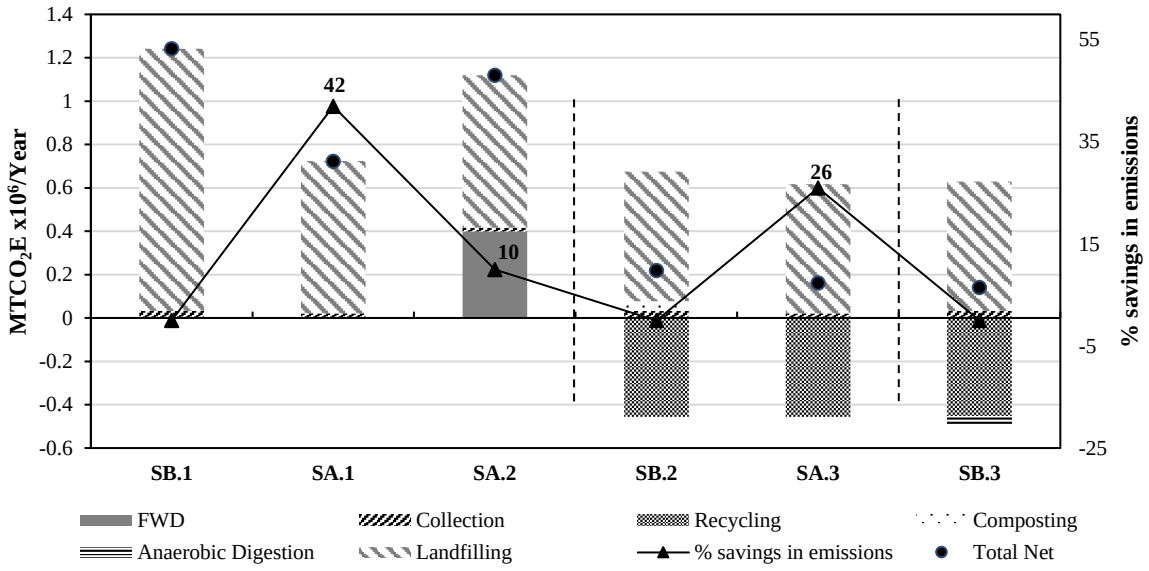


Figure 5.2. Impact of FWD on emissions

The absolute savings in emissions is calculated with respect to the existing baseline scenario whereby
Savings in emissions % = | (Old - New) / Old | x100
where Old = Net emissions from Baseline scenario (e.g. SB.1); and New = Net emissions from alternative scenario (e.g. SA.1)

Baseline Scenarios	***FWD Scenarios***
SB.1 *Collection/Landfilling*	***SA.1*** *Collection/Landfilling/Aerobic Wastewater treatment/ Anaerobic Digestion of Sludge*
	SA.2 *Collection/Landfilling/Anaerobic Wastewater treatment/ Composting of sludge*
SB.2 *Collection/Recycling/ Composting/Landfilling*	***SA.3*** *Collection/Recycling/Landfilling/ Aerobic Wastewater treatment/ Anaerobic Digestion of Sludge*
SB.3 *Collection/Recycling/ Anaerobic Digestion/ Landfilling*	

The comparison between the baseline scenario SB.1 that involves 100% landfilling of collected MSW and alternative scenarios SA.1 or SA.2 that consist of diverting 70% of the food waste into the WW management system (i.e. 74% market penetration rate and 95% of waste ground), resulted in about 42% savings in emissions. This is attributed to savings from food waste not collected and diverted away from landfilling. The results were equally affected by the WW management scheme whereby diverting the food waste from landfilling with energy recovery (SB.1) to anaerobic wastewater treatment with energy recovery in SA.2, did not add significant savings in emissions (~10% less) in comparison to the baseline scenario SB.1 (Figure 5.2). However, the treatment of additional WW using centralized aerobic processes under alternative scenario SA.1 contributed to higher savings in emissions that reached ~42% less emissions with respect to the baseline scenario SB.1 (Figure 2). Managing

the sludge using anaerobic digestion with energy recovery also contributed to lower emissions in comparison to other alternatives (Figure 5.2).

The diversion of 70% food waste from composting in the baseline scenario (SB.2) to the WW management (aerobic process) in scenario SA.3 resulted in ~ 26% reduction in emissions. On the other hand, savings were not significant in comparison to baseline scenario SB.3 with anaerobic digestion of food waste and energy recovery (Figure 5.2).

Consistent with literature findings (Boldrin et al., 2011), scenario SB.3 that considers anaerobic digestion of food waste with energy recovery resulted with up to 36% less emissions in comparison to scenario SB.2 for composting (Figure 5.2). Naturally, the results are dependent on several variables such as the FWD market penetration rate that can range between 25 to 75% and the amount of food waste grinded at the household level that can range between 75 to 95%. For instance, considering a lower range of 25% market penetration rate and 75% of food waste ground would lower emission savings to ~2% with respect to baseline scenarios. Equally important is the impact of waste composition that may vary with location and noticeably different between developed and developing economies (Figure 5.2). For instance, a comparison between a baseline scenario (SB.1) that consists of 100% landfilling in a developed economy, characterized with a lower organic fraction of ~30% (more representative of a developing economy region), and an alternative scenario (SA.1) that considers the use of FWDs (75% market penetration rate and 95% of food waste ground) to divert 70% of food waste would result in only ~24% savings in net emissions. Similarly, net emissions reduction decreased from 10% for a developing economy to 3% for a developed economy when comparing scenarios SA.2 (use of FWDs with anaerobic WW treatment and composting of sludge) and SB.1, as well as from 26% (developing) to 14% (developed) when comparing scenarios SA.3 (recycling/use of FWD/landfilling) and SB.3 (recycling/composting or AD/ landfilling).

This chapter demonstrated that adopting a FWD policy can be a viable alternative solution to reduce emissions. However, several potential negative impacts have been reported (Table 5.1) such as the increase in organic load coupled with higher energy demand at the WW treatment plant (Bolzonella et al., 2003); increase in the oil and grease load with a higher risk on the WW collection network and treatment system (Nilsson et al., 1990); increase in treatment costs (Galil and Yaacov, 2001); increase in water consumption (Constantinou, 2007); increase in sludge produced (Bolzonella et al., 2003; Nilsson et al., 1990); and increase in electricity consumption (CECED, 2003). In contrast, other studies reported positive impacts (Table 5.1) from the use of FWD such as a reduction in the amount of waste to be collected and landfilled (Battistoni et al., 2007; Nilsson et al., 1990); reduction in the amount of landfill leachate formation and treatment (Iacovidou et al., 2012a); increase in landfill capacity and age (Marashlian and El-Fadel, 2005); reduction in emissions (Yi & Yoo, 2014; Evans, 2007; Diggelman & Ham, 2003; Lundie & Peters, 2005; Bernstad Saraiva, 2016; Wainberg et al., 2000); an improvement in the WW treatment process whereby the increase in the organic content of WW can enhance the C/N and C/P ratio, which result with a better removal of nutrients and reduced need for external carbon sources (Bolzonella et al., 2003; Bernstad Saraiva , 2016; Battistoni et al., 2007; De Koning and van der Graaf, 1996; Raunkjaer et al., 1995); an increase in WW biogas production and cost savings (Bolzonella et al., 2003; Galil and Yaacov, 2001; Battistoni et al., 2007; Iacovidou et al., 2012a,b); with minimal impact on water, energy consumption, and on the sewer network (Diggelman and Ham, 2003; De Koning and van der Graaf, 1996; Marashlian and El-Fadel, 2005; Nilsson et al., 1990). Therefore, it is imperative to assess both negative and positive impacts when considering the adoption of a policy to introduce a FWD in the waste management system particularly in the context of the quantity and quality of grinded food waste with its potential influence on the WW management system.

5.3.2 Economic Implications

The results of baseline scenarios (SB.1, SB.2 and SB.3) were used to test the impact of policy options on decreasing emissions. Table 5. *6* presents the details of the analysis upon integrating the FWDs into the management of MSW and WW. It also depicts variations in cost achieved under each scenario as percentages of existing costs under the baseline scenarios (without a FWD policy) based on average conventional and environmental costs. The equivalent economic impact varied from -17 to -28% including environmental externalities (carbon credit and sludge treatment), respectively, depending on the adopted management schemes (Table 5. *6*).

Table 5. 6. Policy scenario analysis: Economic implications

Scenario		**Cost variation (%)**	**Avoided emissions (%)**
SA.1	*Collection/Landfilling/Aerobic Wastewater treatment/ Anaerobic Digestion of Sludge*	-28[(a)]	-42
SA.2	*Collection/Landfilling/Anaerobic Wastewater treatment/ Composting of sludge*	-27[(b)]	-10
SA.3	*Collection/Recycling/Landfilling/ Aerobic Wastewater treatment/ Anaerobic Digestion of Sludge*	-26[(c)]	-26
		-17[(d)]	-1

[(a)] Cost variation with respect to baseline scenario SB.1 (Collection/Landfilling)
[(b)] Cost variation with respect to baseline scenario SB.1 (Collection/Landfilling)
[(c)] Cost variation with respect to baseline scenario SB.2 (Collection/Recycling/ Composting/Landfilling)
[(d)] Cost variation with respect to baseline scenario SB.3 (Collection/Recycling/ Anaerobic Digestion/Landfilling)
Cost variation is calculated with respect to existing costs of baseline scenario whereby *Cost variation % = [(Old- New) / Old] x100 where Old = Total cost of baseline scenario (e.g. SB.1) and New = Total cost of alternative scenario (e.g. SA.1)*
Note that the cost variation includes environmental externalities in the form of carbon credit and sludge treatment, such as the carbon credit based 3.3 US$/MTCO$_2$E (adapted from Ecosystem Marketplace, 2016).

The results indicate that the tested scenarios (SA.1, SA.2, SA.3) considering the diversion of the 70% of food waste generated through the use of FWDs, resulted with lower costs in comparison to baseline scenarios considering landfilling (SB.1), composting (SB.2), or anaerobic digestion (SB.3) of food waste, reaching a 28% cost reduction with environmental externalities of emissions reduction and sludge management, depending on WW and SW management methods (Table 5. *6*). It is worth reemphasizing that many other combinations of scenarios can be tested and only a few were presented above for illustrative purposes.

The sensitivity analysis for scenarios (SA.1, SA.2, SA.3) showed an equivalent economic impact ranging between -1 and -33%, including environmental externalities (Figure 5.3) thus emphasizing the viability of a FWD policy. Evidently, the variation in the cost of sludge management exhibited a significant impact on savings (Figure 5.3.d).

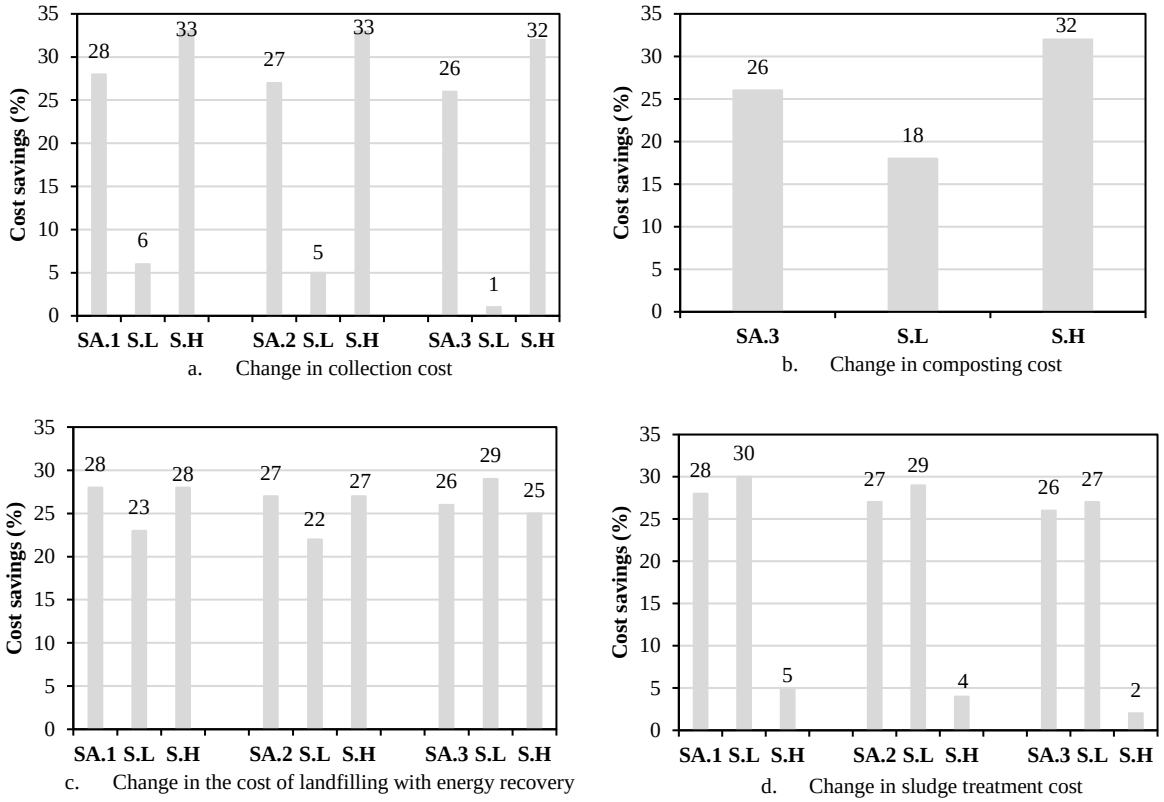


Figure 5.3. Sensitivity to change in selected economic parameters

S.L corresponds to the change in the economic parameter to its lowest value and S.H to the change in its upper value
SA.1: Collection/Landfilling/Aerobic Wastewater treatment/ Anaerobic Digestion of Sludge
SA.2: Collection/Landfilling/Anaerobic Wastewater treatment/ Composting of sludge
SA.3: Collection/Recycling/Landfilling/ Aerobic Wastewater treatment/ Anaerobic Digestion of Sludge

5.4 Conclusion

This chapter revealed that integrating FWDs in a developing economy characterized with a high fraction of food waste can be a viable alternative solution to reduce emissions for carbon trading. The results indicated that adopting a FWD policy reduced emissions by about 42% at cost savings reaching ~28% when environmental externalities are considered including sludge management. While increasing the market penetration and the fraction of food waste ground contribute to a decrease in net emissions depending on wastewater and sludge management processes, the system remains economically attractive even when adding the wastewater and sludge management costs. The sensitivity analyses on processes with a wide range in costs showed an equivalent economic impact thus emphasizing the viability of a FWD policy although the variation in the cost of sludge management exhibited a significant impact on savings.

CHAPTER 6

LIFE CYCLE ASSESSMENT FOR SOLID WASTE MANAGEMENT IN LEBANON: ECONOMIC IMPLICATIONS OF CARBON CREDIT

6.1 Introduction

Population growth, development, and limited land resources around urban areas are curtailing waste management efforts in cities with inadequate planning policies (Arena et al., 2003). As such, solid waste is increasingly raising serious challenges and environmental concerns due to inefficient systems particularly in developing countries where landfilling remains the preferred route due primarily to economic factors or lack of technical expertise in other alternatives (e.g. biological and/or thermal treatment). The impacts of conventional landfilling have been long documented (El-Fadel et al., 1997). Its contribution to greenhouse gas (GHG) emissions reaches ~3% worldwide and up to 15% in developing economies (Blanco et al., 2014). Thus, the proper selection of waste processing technologies through an integrated waste management system (IWMS) with minimal impacts and reduced emissions is imperative. In this context, the Life Cycle Assessment (LCA) approach has been recognized as a valuable tool widely adopted as an internationally standardized method (ISO, 2006a, b) that is effective in quantifying environmental impacts of management alternatives thus contributing towards the decision-making process through the comparison of various systems. Accordingly, several LCA-based accounting tools have been developed for estimating emissions from such systems (Gentil et al. 2010; Laurent et al. 2014a, b). Recent efforts targeted LCA applications in the environmental assessment of waste management. Many of these efforts focused on developed economies (Di Maria & Sisani, 2017; Ripa et al., 2017; Thomsen et al., 2017; Di Maria et al.,

2016; Tunesi et al., 2016; Herva et al., 2014) with limited applications on economies in transition or developing economies (Liu et al., 2017a, b; Noya et al., 2018; Othman et al., 2013) where waste composition is different and management practices still concentrate on landfilling or open dumping (Laurent et al., 2014a, b). In addition, most studies emphasized individual processes instead of integrated systems (Tabata et al., 2010; Laurent et al., 2014a, b). Accordingly, this study aims to fill a gap in the developing context particularly with respect to the effect of waste composition and integrated systems. For this purpose, an LCA approach is adopted at the system level to identify alternatives with minimal environmental impacts and reduced emissions. An economic valuation, sensitivity analysis, and comparative assessment defined economically attractive scenarios taking into consideration related carbon credit.

6.2 Materials and methods

6.2.1 LCA analysis

The LCA was conducted in accordance with the ISO 14040 standards (ISO, 2006a, b) and ILCD Handbook (EC, 2010). Accordingly, four main steps can be considered in the LCA study: goal and scope definition, inventory analysis, impact assessment and interpretation.

6.2.1.1 Goal and scope

The LCA methodology was used in this study to compare waste management alternatives and assess corresponding environmental impacts. The test area (Beirut, Lebanon) considered in this study encompasses 297 municipalities (Figure 6.1.) with > 2M inhabitants generating 2,800 - 3,000 tonnes of municipal solid waste (MSW) daily with an average waste composition as presented in Table 6.1. Waste is collected daily by a fleet of 332 collection vehicles that consume an average volume of diesel equivalent to 6.3 L/tonne of waste generated. The latter was calculated based on an overall diesel fuel consumption per year from various types of collection vehicles used in the test area. The data was used to estimate the average fuel consumed per tonne of waste collected. A comparison between international fuel

consumption data and results from the test area shows that on average the resultant fuel consumption is comparable to the average reported value of 6 L/tonne of waste (Tanskanen and Kaila, 2001; Chen and Lin, 2008). When compared with other literature reported values from developed economies (Larsen et al., 2009) the consumption fell at the lower end of reported ranges of 6.3 and 10.1 L/tonne of waste from rural areas. However, these studies did not provide adequate details on their tested areas to allow a more accurate quantitative comparison particularly that some studies suggest a lower range of 2.8 to 3.6 L/tonne of waste for high density urban areas (Nguyen and Wilson, 2010). In the local context, several factors may influence the higher fuel consumption, mainly traffic congestion that affect the number of traffic-related stops, location of transfer stations, as well as inefficient routes, and age of vehicles (Sonesson, 2000).

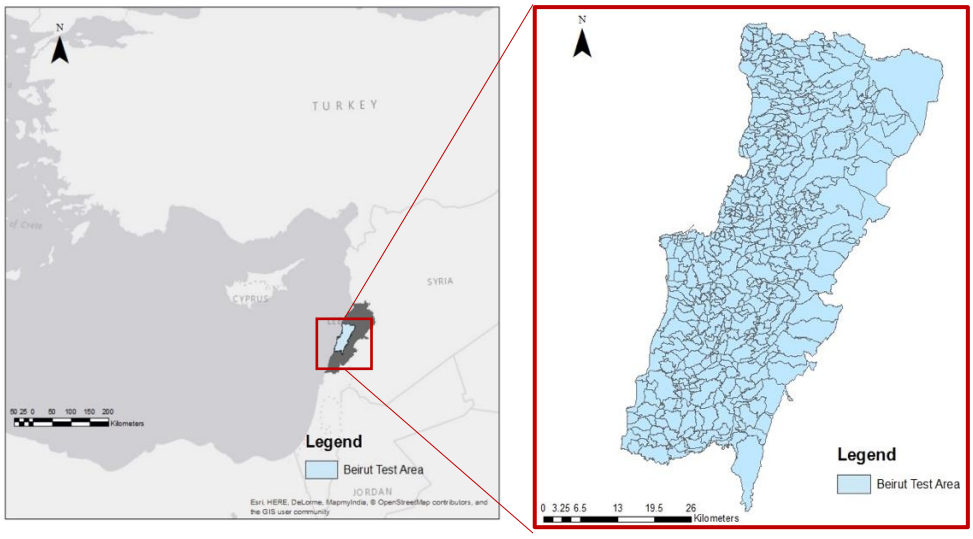


Figure 6.1.General location of test area

Table 6.1. MSW composition
(Data extracted from Laceco/Ramboll 2012)

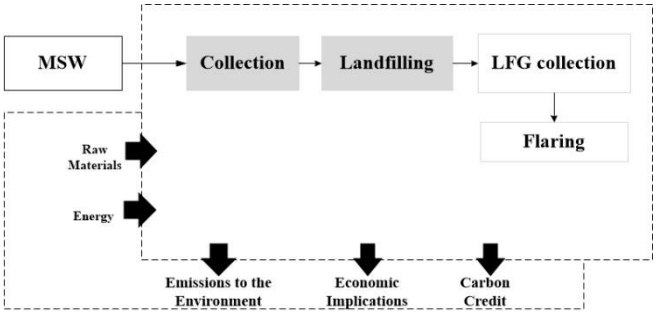
Waste category	**(%)**
Food	53.4
Glass	3.4
Metals	2
Nappies	3.6
Papers	15.6
Plastics	13.8
Textiles	2.8
Wood	0.8
Others	4.6
Total	*100*

6.2.1.2 Functional unit and system boundaries

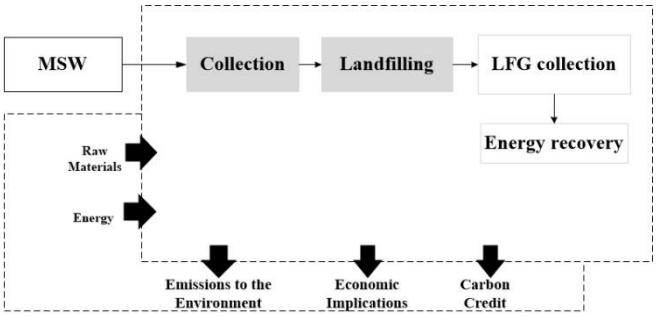
The functional unit (FU) was the management of 1 tonne of waste generated in the test area considered. The FU was also assumed as the reference flow on which the analysis was performed. Several scenarios were simulated to compare alternative management systems while considering the emissions to the environment, economic implications, and carbon credit. The waste treatment processes and systems' boundaries (defined inside the frame boundaries) of the five scenarios are depicted in Figure 6.2.

The systems' boundaries include emissions from waste management including indirect upstream emissions arising from inputs of materials and energy (electricity & fuel), direct operational emissions from systems' operation such as onsite operating equipment and waste degradation, and indirect downstream emissions related to energy generation, materials substitution, management of residues, and carbon storage.

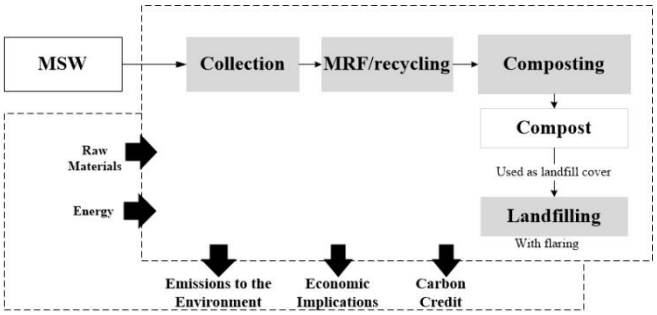
The baseline scenario (S1) reflects a policy towards landfilling of all the waste with LFG flaring because economic considerations render landfills as most attractive. Scenario S2 replaces flaring in S1 with energy recovery. However, land availability is continuously constraining landfilling particularly in urban areas. Hence, three other scenarios targeted the minimization of landfilling and optimization of recycling and biological treatment (composting-S3 or anaerobic digestion-S4) or incineration with energy recovery (S5).



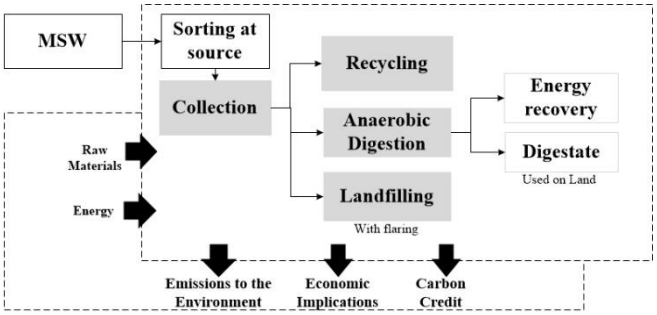
S1: Baseline scenario: Landfilling all waste with flaring



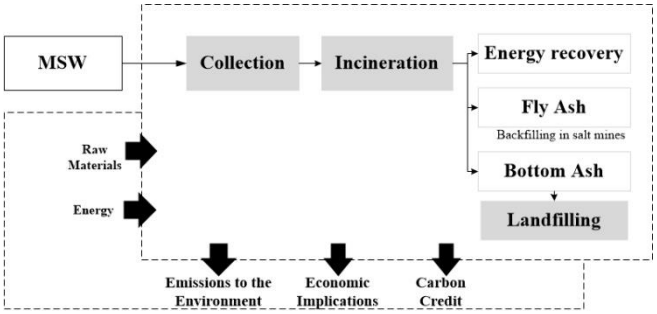
S2: Upgrade LFG capture system in S1 + energy recovery



S3: Material recovery facility (MRF)+ Max recycling & composting + landfilling



S4: Max recycling & anaerobic digestion + landfilling



S5: Incinerate all waste + energy recovery

Figure 6.2. Systems' boundaries

6.2.1.3 Life cycle inventory

In a review of 220 waste-LCA studies, Laurent et al. (2014b) reported that around half of them favored the use of dedicated waste-LCA models instead of general LCA models. As such, in this study, the dedicated waste-LCA EASETECH software (Clavreul et al., 2014; Laurent et al., 2014b; Liu et al., 2017 a, b) was tailored to reflect the test area characteristics (waste composition, electricity mix, waste management, etc.) and applied to assess the environmental impacts of various waste management processes and estimate corresponding emissions. The data was collected from annual reports of facility operations, published sources, face-to-face interviews, site visits and field observations, and supplemented with reported literature. Table 6.2 synthesizes the input data of tested scenarios for assessing waste management processes.

The tested system includes the collection of mostly commingled waste that is transported to a material recovery facility (MRF) with a recycling program, biological treatment (composting-anaerobic digestion), incineration, and landfilling (Table 6.2) with energy recovery when applicable. Input data for landfilling are summarized in Table 6.3 with several modules combined to represent this process: (1) construction and operation of the landfill; (2) gas generation using first order decay for a 100 years-time horizon and natural oxidation (in daily, intermediate, and final covers); (3) leachate generation (without treatment); (4) stored substances in the landfill contribute to eco-toxicity and sequestrated carbon. The Life Cycle Inventory (LCI) data for individual waste treatment processes, extracted from the EASETECH database, were detailed in Appendix D Waste collection was simulated for all scenarios based on an overall fuel consumption per year. Electricity requirements were derived using the Ecoinvent database (Ecoinvent, 2017) based on the country's national electricity mix, that rely primarily on oil-fired power plants (95.5%) and a small contribution from hydropower plants (4.5%) (MoE/UNDP/GEF, 2015a).

Table 6.2. Main features of the analysed scenarios

ID	Scenario	Description
S1	Landfilling (100%) with flaring	– Landfilling of waste with LFG collection and flaring at 18% (MoE/UNDP/GEF, 2015b)
S2	Landfilling (100%) with energy recovery	– Increase in the LFG collection efficiency up to 52% (DTU, 2017) with energy recovery facilities
S3	MRF with recycling (15%) + composting (50%) + landfilling (35%) with flaring	– Commingled waste collected and transferred to two MRFs prior to treatment for material recovery (cardboard/paper (43.22%), metal (16.85%), glass (6.47%), plastic (33.46%) Laceco/Ramboll (2012)) and recycling or the removal of inert residues – The 15% recovery rate is consistent with studies implementing successful recycling programs (Di Maria et al., 2015, 2013). Existing MRFs in the study area achieved a recycling recovery rate close to 10% by mechanical separation using bag openers, trommel screens, and magnetic separation with manual sorting (CDR-LACECO, 2014). With separation at source, it is expected that this rate can reach the 15% adopted in this study – 81% of the recovered recyclables are sold for recycling industries the rest is sent for landfilling (CDR-LACECO, 2014) – Simulated recycling processes include: shredding and reprocessing of mixed paper and cardboard material; shredding and reprocessing of plastic materials; re-melting of glass cullet and forming of glass bottles (substituting virgin bottle production); and shredding and reprocessing of steel scrap. Note that inventory data of avoided emissions form recycling processes were adopted from the EASETECH database (DTU, 2017) – Food waste is treated using open windrow composting at 300 tonnes / day with air supplied by mechanical turning using wheeled loader – The resulting compost has a C/N ratio of 16.5, pH of 7.3, average density of 470 kg/m^3, average Nitrogen content of 1.44%, average organic content of 52.6%, and moisture content of 48% by wet weight (CDR-LACECO, 2014). However, the compost has not been well-accepted with farmers mostly because it contains a small fraction of glass due to no separation of waste at source. – Non-sold compost used as a cover material at the landfill – The consumption of electricity and fuel during operation are 0.02 kWh/tonne of waste composted and 3.28 Liters/tonne of waste composted, respectively (Boldrin et al. 2009) – The remaining waste stream is landfilled with LFG flaring
S4	Sorting at source + recycling (15%) + AD (50%) with energy recovery + landfilling (35%) with flaring	– Waste management involves waste sorting-separation at source prior to the AD treatment. The source separation efficiency by individual waste component is presented in the supplementary material (Table C2) – The average electricity production is around 244 kWh/tonne (based on a range of 184-299 kWh/tonne) with engine conversion efficiency of 36% (Møller et al., 2009) – Digestate used in land application and as a substitute for fertilizer production – Methane leakage rate from AD assumed at 10% (DTU, 2017) – Energy consumption at 49 kWh of electricity and 0.9 L of diesel / tonne of organic waste (DTU, 2017) – The LCI data of AD is adopted from the EASETECH database (DTU, 2017) based on an average biogas plant in Europe using a one stage wet thermophilic anaerobic digestion process for the treatment of the organic fraction of municipal solid waste (see Supplementary Material Table C5)
S5	Incineration (100%) with energy recovery	– All MSW is collected for incineration coupled with energy recovery – LCI data of incineration is adopted from a typical incineration plant in Denmark, which was selected from the EASETECH database (DTU, 2017) and adjusted using the waste composition and specific data of the test area (see Supplementary Material Table C6) – Given the higher moisture of the MSW in the test area, an 18% net electricity output efficiency (after subtracting the operational energy consumed within the facility) was adopted (Liu et al., 2017a, b; Chen & Christensen, 2010; Di Maria et al., 2018; Di Maria & Pavesi 2006; Münster & Lund, 2010) – Average calorific value of the waste in the test area is 6.9 MJ/Kg Laceco/Ramboll, 2012), which is at the edge of the upper range of the LHV that is 5-6 MJ/ kg required to sustain the burning process (Chen and Christensen, 2010; Di Maria et al., 2018) and avoided auxiliary equipment (Zhao et al., 2012) – Diesel fuel consumed for the operation of the facility was considered at 1 Liter/tonne of waste incinerated (Astrup et al., 2009) – Requires electricity input of 70 kWh / tonne of waste and generates 20% ash (Yay, 2015; Fernandez-Nava et al., 2014) – Fly ash exported and used as backfilling in salt mines and bottom ash delivered to inert landfilling without energy recovery

L: landfilling; I: incineration: R: recycling; C: composting; MRF: material recovery facility; AD: anaerobic digestion; MSW: municipal solid waste; LFG: landfill gas; LHV: lower heating value; LCI: life cycle inventory

Table 6.3. Data on consumption, expressed per tonne of waste for landfilling *(Manfredi et al., 2009; 2014; DTU, 2017)*

Consumption	**Value**
Electricity (kWh)	8
Diesel fuel consumption (L)	2
Steel sheets (tonne)	0.00014
Aluminum (tonne)	$5.8x10^{-8}$
Polyvinylchloride resin (tonne)	10^{-5}
Polyvinylchloride fibers (tonne)	$4x10^{-8}$
Polyethylene high density granulate (tonne)	0.00023
Gravel (tonne)	0.18
Clay (tonne)	0.082
Copper (tonne)	$9.87x10^{-9}$

6.2.1.4 Impact assessment

The ILCD (EC-JRC, 2011) impact assessment method was used in this study and is further described in Hauschild et al. (2013). The impact assessment was conducted for all scenarios using several indicators including climate change (global warming potential of 100 years-time horizon), photochemical oxidant formation, stratospheric ozone depletion, acidification, depletion of abiotic resources, and freshwater and marine eutrophication. The model results are presented in the form of characterized and normalized potential impacts. The latter is expressed as Person Equivalent (PE), which is the contribution to an indicator of an average person in a given reference year as outlined in Table 6.4.

Table 6.4. Normalization references for the selected environmental impact categories *(DTU, 2017; Laurent et al., 2013)*

Impact category	**Characterization unit**	**Global Normalization reference[(a)] (2010 or 2013)**
Climate change, GWP-100	Kg CO_2-equivalents	8100
Photochemical oxidant formation	Kg NMVOC	56.7
Stratospheric ozone depletion	Kg CFC-11-equivalents	0.0414
Acidification	Kg SO_2-equivalents	74
Freshwater eutrophication	Kg P-equivalents	0.62
Marine eutrophication	Kg N-equivalents	9.38
Depletion of abiotic resources	MJ	$6.24x10^4$

[(a)] Characterization unit/ person/year: Person Equivalent (PE) = Characterized impact category (characterization unit) / Normalized reference (characterization unit/ person/year)

[(b)] Note that the global warming potential-GWP_{100} that was selected in this study follows the IPCC (2013) reference, including climate-carbon feedbacks.

6.2.2 Economic analysis

The cost of MSW management alternatives was estimated using an average cost for the total amount of waste managed under each scenario (Table 6.5). The wider range in Table 6.5 is considered in the sensitivity analysis to define breakeven points that could assist in defining zones of enhanced economic viability. Inflation was not included, and values were considered at a constant year zero. The offset of emissions was quantified based on the carbon market. The latter ranged from 0.5 to 50 US$ per metric tonne of CO_2 equivalent ($MTCO_2E$) in 2016, with an average price of 3 US$/$MTCO_2E$, which is the lowest reported market value from voluntary actors since 2006 (Ecosystem Marketplace, 2017). The average value was used to assess associated benefits and allows for the estimation of minimal savings when the carbon footprint is reduced through regulated and voluntary global markets for trading or offsetting of carbon credits. A wider range is considered in the sensitivity analysis to define breakeven points that could assist in decision making and policy planning.

Table 6.5. Average cost of MSW management (US$/ tonne of waste)

	Collection and transport	Sorting	Composting	Anaerobic digestion with energy Recovery	Landfilling	Landfilling with energy Recovery	Incineration with energy Recovery
MoE/UNDP/ECO DIT (2011); CDR (2010)	33	23	25	…	46	…	…
Maalouf and El-Fadel (2017)	20-250	26-28	5-90	20-150	10-100	13-67	…
Assamoi and Lawryshyn (2012)	…	…	…	…	18	…	38
Dijkgraaf and Vollebergh (2004)	…	…	…	…	45	…	97
EC (2002)	…	…	…	80	62	58	88
Jamasb and Nepal (2010)	…	26	…	…	15	13	70
Rabl et al. (2008)	…	…	…	…	45	40	92
Tsilemou & Panagiotakopoulos (2006)	…	…	17-73	22-67	12-50	…	117
World bank (2012)	20-250	…	5-90	20-150	10-100	…	120
Wrap (2016)	…	28	27	44	21	…	94
Range (US$/tonne)	*20-250*	*23-28*	*5-90*	*20-150*	*10-100*	*13-67*	*38-120*
Adopted average (US$/tonne)	**33**[(a)]	**23**[(a)]	**25**[(a)]	**85**[(b)]	**46**[(a)]	**57**[(c)]	**90**[(d)]

[(a)] Current costs in the test area applied for the baseline scenario (S1). Costs exclude leachate treatment.
[(b)] Anaerobic digestion includes energy recovery but excludes cost of residue sale or disposal.
[(c)] Average including an additional ~17 US$/tonne of waste for onsite leachate and gas management (EC, 2002; Damgaard et al., 2011).
[(d)] Includes energy recovery but excludes disposal cost of bottom and fly ash.

6.2.3 Sensitivity analysis

A one-factor-at-a-time sensitivity analysis was conducted to assess key parameters by varying the main assumptions whereby each parameter was varied one at a time to assess its impact on emissions. These parameters included the fraction of the LFG collected, food waste fraction, land application of compost, and net energy recovery efficiency of waste incineration. The cost range of carbon credit and several waste management processes (anaerobic digestion, incineration and landfilling with energy recovery) were also considered one at a time to assess their impacts on net cost variations and define breakeven points for decision making and policy planning. Note that only processes related to alternative technologies were tested for cost variation because for the baseline scenario (landfilling, sorting/recycling, and composting), the costs are already fixed.

6.3 Results and discussion

6.3.1 LCA analysis

The results of the LCA characterization analysis per functional unit (1 tonne of MSW managed) for each indicator of the tested scenarios are depicted in Figure 6.3 taking the baseline scenario-S1 as the normative reference, whereby all other scenarios are indexed to the impact of S1 per 1 tonne of MSW (S1 is 100%). Figure 6.4. depicts the contribution of each scenario disaggregated by waste management processes to the impact categories. The results show that landfilling scenarios contribute the most to all impact categories (Figure 6.3). For instance, the baseline scenario S1 that consists of landfilling all waste with flaring of LFG collected (18%) is associated with the highest climate change impact (GWP, 100) in comparison to other scenarios due to uncaptured methane emissions from landfilling (Figure 6.4.a). Note that developing economies are generally characterized by poorly operated landfills with inefficient LFG collection systems at efficiencies ranging between 28-40% (Banar et al., 2009) compared with 60 to 98% reported in developed economies (EPA/ICF, 2016). In this case, the low collection efficiency can be attributed to the high fraction of the food waste component associated with rapid decomposition leading to unstable LFG generation; the high moisture content of the waste resulting in the generation of large quantities of leachate that reduce the collection efficiency; and inefficient gas recovery systems (Liu et al., 2017 a, b; Zhan et al., 2015). The sensitivity analysis showed that a 10% increase in the food waste fraction, led to an equivalent increase of 8.6% in the climate change impact indicator (Table 6.6). This can be attributed to emissions through the landfill surface during the early stage of waste disposal (1-2 years). Similarly, an increase in the LFG collection efficiency up to 60%, contributed to a 58% decrease in the climate change impact indicator (Table 6.6). Major savings in the climate change impact were achieved under scenario S4 considering incineration with energy recovery (Figure 6.4.a). In this context, results from sensitivity analysis showed that a 10% increase in the electricity efficiency from incineration could save 53% of the climate

change impact indicators (Table 6.6).

All scenarios except S3 and S4 exhibited a similar trend for the depletion of abiotic resources due to avoided raw material usage through recycling. Moreover, waste collection has a significant impact in this category due to the use of fossil fuels (Figure 6.4.e). The waste composition exhibited different impacts on emissions from various treatment processes. For instance, the high fraction of biodegradable food waste was advantageous to the AD process whereby more biogas generation for energy recovery would result in greater savings in emissions. However, environmental benefits from these scenarios are affected by several factors such as gas leakage under AD and land application of compost or digestive residues. In the case of the test area, food waste is not separated at the source contributing to a lower efficiency of separation which translates into less emissions savings. The low-quality compost in the test area is invariably not accepted by farmers and hence used as intermediate covers in landfilling thus contributing to greater emissions. Accordingly, the separation of waste at source will contribute to about 11% savings in acidification and 7% in photochemical oxidant formation (Table 6.6). This can be attributed to the production of better quality compost that can be used on land resulting in savings from the substitution of fertilizer production and carbon storage. Further savings can also be attributed to avoided energy consumption during pretreatment.

Stratospheric ozone depletion is caused by emitting "methane bromotrifluoro-Halon 1301", which is a consequence of crude oil production, petroleum and natural gas (Yay, 2015). The best alternative against ozone depletion is scenario S5 that considers incineration with energy recovery substituting electricity production from oil-fired power plants in the case of the test area (Figure 6.4.c). However, the benefits of waste incineration can be compromised by the high organic fraction and moisture content that decrease the net energy recovery in comparison to developed economies at 30-31% (Gohlke and Martin, 2007; Murer et al., 2011).

Table 6.6. Sensitivity to key input parameters

Parameter		Climate change [a]	Acidification [b]	Photochemical oxidant formation [c]	Comments
Food waste fraction	Initial value (53.4 %) Increase 10%	859.5 (S1) 932.9	0.27 (S1) 0.28	0.63 (S1) 0.67	The change in the food waste fraction had a great impact on emissions (Liu et al., 2017a)
	% Change	*+8.6%*	*+5.5%*	*+6.3%*	*% change considering 53.4 to 58.7*
LFG collected	Initial value (18 %) New value (60%)	859.5 (S1) 58.2	0.27 (S1) 0.27	0.63 (S1) 0.47	Collection efficiency of (60%) for a typical operating landfill with wet waste (EPA/ICF, 2016).
	% Change	*-58.3%*	*-1%*	*-24.4%*	*% change considering 18 to 60*
Land application of compost	Without With	19.9 (S3) 19.8	0.12 (S3) 0.10	-0.14 (S3) -0.15	Avoided emissions from the application of compost as a fertilizer are adapted from (DTU, 2017; Boldrin et al. 2009) assuming the waste is co-composted with a nutrient material
	% Change	*-0.5%*	*-11.1%*	*-7.1%*	*% change from not considering land application of compost to considering it*
Net energy recovery efficiency of waste incineration	Initial value (18 %) Increase 10%	-90.5 (S5) -138.5	*-5.71* (S5) *-6.36*	-0.81(S5) -1	Given the higher moisture of MSW characteristic of the test area, the net electricity output efficiency was set at 18% (Liu et al., 2017a, b; Chen & Christensen, 2010; Di Maria et al., 2018).
	% Change	*-53%*	*-11.4%*	*-23.2%*	*% change considering 18 to 19.8*

S1: Baseline scenario: Landfilling with flaring; S3: Max recycling & composting + landfilling; S5: Incinerate + energy recovery
The percent change is calculated with respect to the total initial value of the different impact categories (expressed depending in the characterization unit)
[a] Impact on climate change expressed in Kg CO_2E/tonne of waste
[a] Acidification potential expressed in Kg SO_2E/tonne of waste
[c] Photochemical oxidant formation expressed in Kg NMVOC/tonne of waste

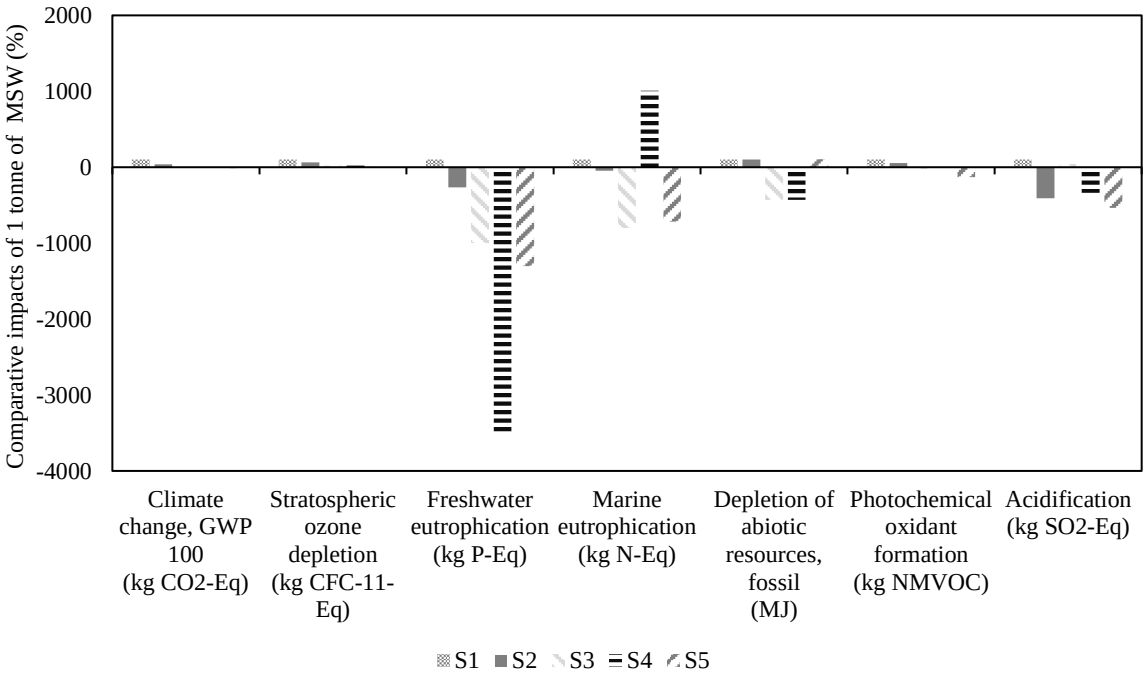


Figure 6.3. Life cycle characterization per 1 tonne of MSW in the test area

S1 is the normative reference, whereby all other scenarios are indexed to the impact of S1 per 1 tonne of MSW (S1 is 100%)

S1: Baseline scenario: Landfilling all waste with flaring
S2: Upgrade LFG capture system in S1 + energy recovery
S3: Max recycling & composting + landfilling
S4: Max recycling & anaerobic digestion + landfilling
S5: Incinerate all waste + energy recovery

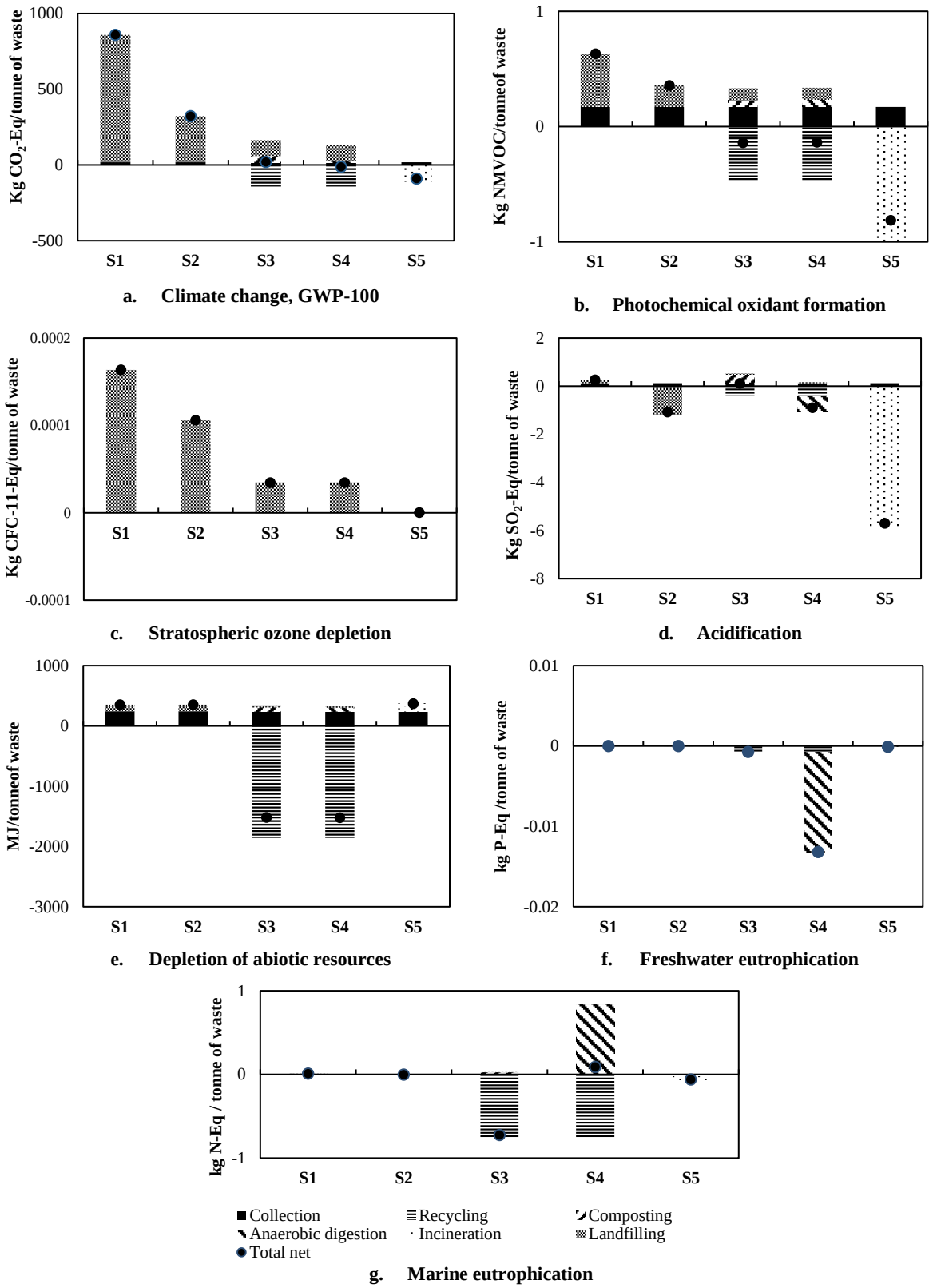


Figure 6.4. Contribution of each scenario to the impact categories

S1: Baseline scenario: Landfilling all waste with flaring
S2: Upgrade LFG capture system in S1 + energy recovery
S3: Max recycling & composting + landfilling
S4: Max recycling & anaerobic digestion + landfilling
S5: Incinerate all waste + energy recovery

Landfilling scenarios contribute mostly to the photochemical oxidant formation due to

methane emissions with savings achieved under scenarios S3 and S4 considering the minimization of landfilling in comparison to the baseline scenario S1 (Figure 6.3). Waste collection also contributes to the effect of photochemical oxidation due to Sulfur dioxide emissions (Figure 6.4.b). The acidification potential is measured by its capacity to form H^+ ions relative to SO_2 (Banar et al., 2009; Yay, 2015). Energy recovery is the best alternative to reduce this impact due to savings in NO_x emissions from equivalent electricity generation. Accordingly, incineration coupled with energy recovery (Scenario S5) achieved the most savings from this category followed by upgrading the LFG collection system from flaring to energy recovery (Scenario S2) (Figure 6.3). Alternative scenario S4 that considers anaerobic digestion also reduced the impact of acidification due to the use of biogas for energy recovery and digestate in farming as a substitute for fertilizer production (Figure 6.4.d). Similarly, this scenario achieved significant savings in freshwater eutrophication due to avoided fertilizer production and energy recovery (Figure 6.4.f). However, S4 might affect marine eutrophication due to nitrate run-off from the application of digestate on land (Yay, 2015; Hansen et al., 2006).

In summary, the comparison of scenarios (Figure 6. *5*) using the normalized potential impacts showed that the integrated MSW management system is contributing most to the climate change indicator. While, the highest impact arises from landfilling all waste (scenario-S1), mainly due to uncaptured methane emissions, the greatest benefits can be derived from incinerating all the waste (scenario-S5) due to energy recovery. Similarly, maximizing recycling and composting or anaerobic digestion in scenarios S3 and S4 contributed to significant savings in all impact categories.

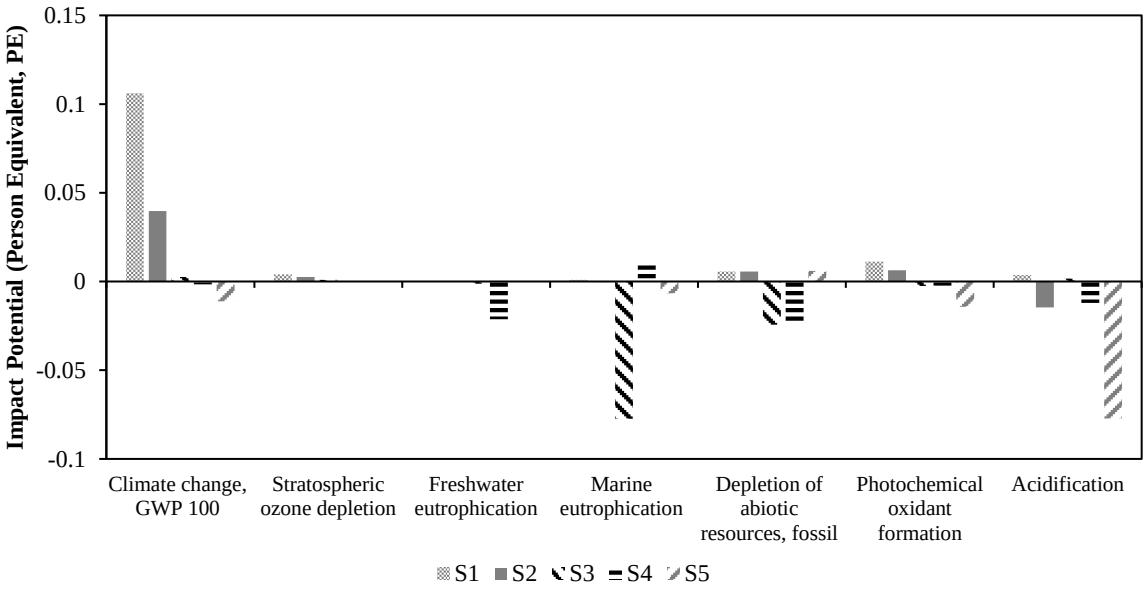


Figure 6. 5. Normalized potential non-toxic impacts from the treatment of 1 tonne of MSW

S1: Baseline scenario: Landfilling all waste with flaring
S2: Upgrade LFG capture system in S1 + energy recovery
S3: Max recycling & composting + landfilling
S4: Max recycling & anaerobic digestion+ landfilling
S5: Incinerate all waste + energy recovery

6.3.2 Economic analysis

Reducing emissions from alternative scenarios can be subject to economic constraints depending on the technology adopted and whether reductions are considered in the economic valuation (Table 6.7). In the context of the existing waste management system, maximizing waste recycling and composting with minimal landfilling decreases the management cost most (-21% with carbon credit). Optimizing emissions reduction through incineration and energy recovery (S5) reduces emissions most at the expense of an overall increase in cost (+52% with carbon credit) (Table 6.7). The breakeven analysis defines when the carbon credit could enhance the economic viability of adopting a favorable policy towards a technology change. Holding the same value for all parameters, Figure 6. 6 depicts the breakeven points for all scenarios taking into consideration only the reported cost range of carbon credit (0.5 to 50 US$/MTCO$_2$E). While scenarios S2 and S4 become economically viable at a carbon credit cost of 21 US$/MTCO$_2$E, scenario S3 stipulating maximum recycling and composting with landfilling remains profitable under the entire range of carbon credit. On the other hand,

incineration with energy recovery under scenario S5 requires the highest capital investment and cannot achieve overall economic attractiveness except under the highest carbon credit cost of ~50 US$/MTCO$_2$E albeit showing the greatest potential for emissions reduction.

Table 6.7. Economic implications of scenario analysis

Scenario	Description	Avoided emissions (%)	Cost variation (%)
S1	Collection + Landfilling with gas flaring	0 (Baseline)	0 (Baseline)
S2	Collection +Landfilling + with landfill gas energy recovery	-63	12
S3	Collection + Max Recycling & Composting + Landfilling with gas flaring	-98	-21
S4	Collection + Max Recycling & Anaerobic digestion + Landfilling with gas flaring	-101	17
S5	Collection + Incineration + Energy recovery	-111	52

Avoided emissions (climate change impact indicator) is calculated with respect to existing total emissions of baseline scenario (S1) whereby *Avoided emissions % = [(Old – New)/Old]* where Old= Total emissions of baseline scenario (S1) and New= Total net emissions of alternative scenario S_i where i = 2 to 5

Cost variation is calculated with respect to existing costs of baseline scenario (S1) whereby *Cost variation % = [(Old- New) / Old] x100* where Old = Total cost of baseline scenario (S1) and New = Total cost of alternative scenario S_i where i = 2 to 5

Note that the cost variation includes environmental externalities in the form of carbon credit that is based on 3 US$/MTCO$_2$E (Ecosystem Marketplace, 2017).

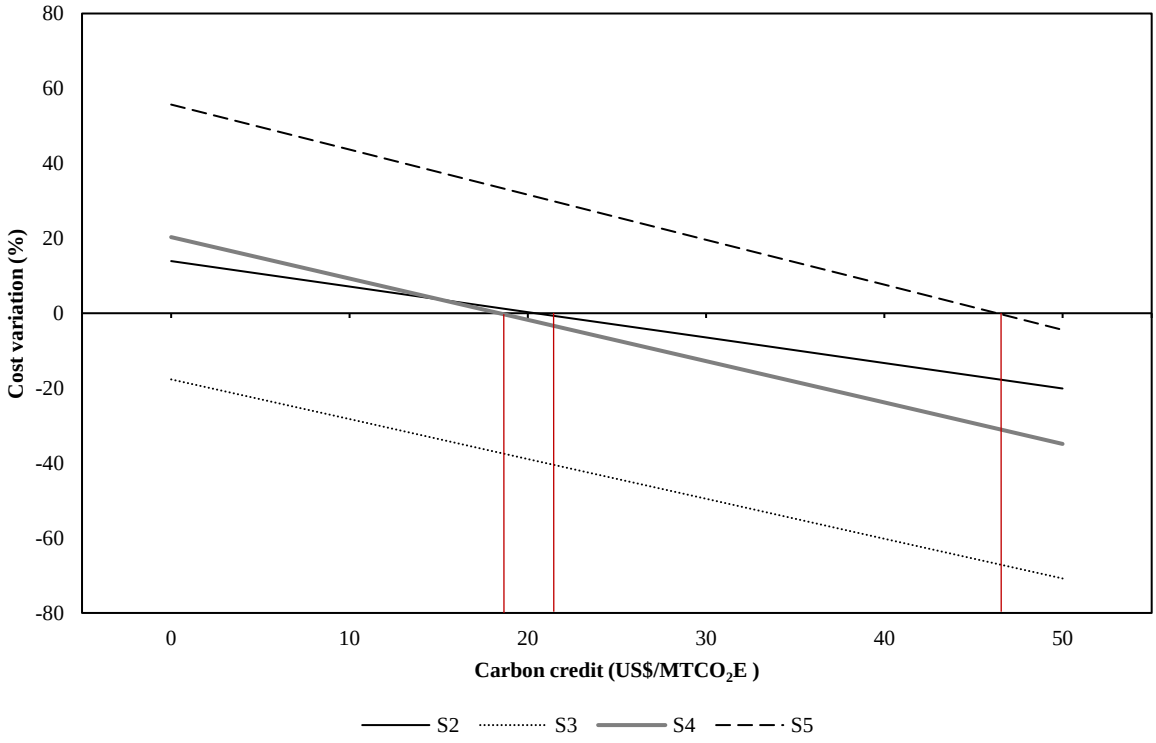


Figure 6. 6. Breakeven point analysis for carbon credit with respect to baseline scenario S1

S1: Baseline scenario: Landfilling all waste with flaring
S2: Upgrade LFG capture system in S1 + energy recovery
S3: Max recycling & composting + landfilling
S4: Max recycling & anaerobic digestion+ landfilling
S5: Incinerate all waste + energy recovery

The sensitivity analysis for all scenarios showed a significant change in the economic impact ranging from –76% to +93% at a cost range of carbon credit of 0.5 and 50 US$/MTCO$_2$E (Figure 6. 7). Incineration with energy recovery contributed most to the variation in cost that ranged between –70% and 93% with respect to the baseline scenario S1 (Figure 6. 7). Energy recovery of LFG collected instead of flaring (S1) can contribute to significant savings in cost that reached 76% with respect to S1 at a higher carbon credit exchange rate (Figure 6. 7). Similar to incineration, the cost of anaerobic digestion with energy recovery varies widely from –76% to 61% with respect to the baseline scenario S1 (Figure 6. 7).

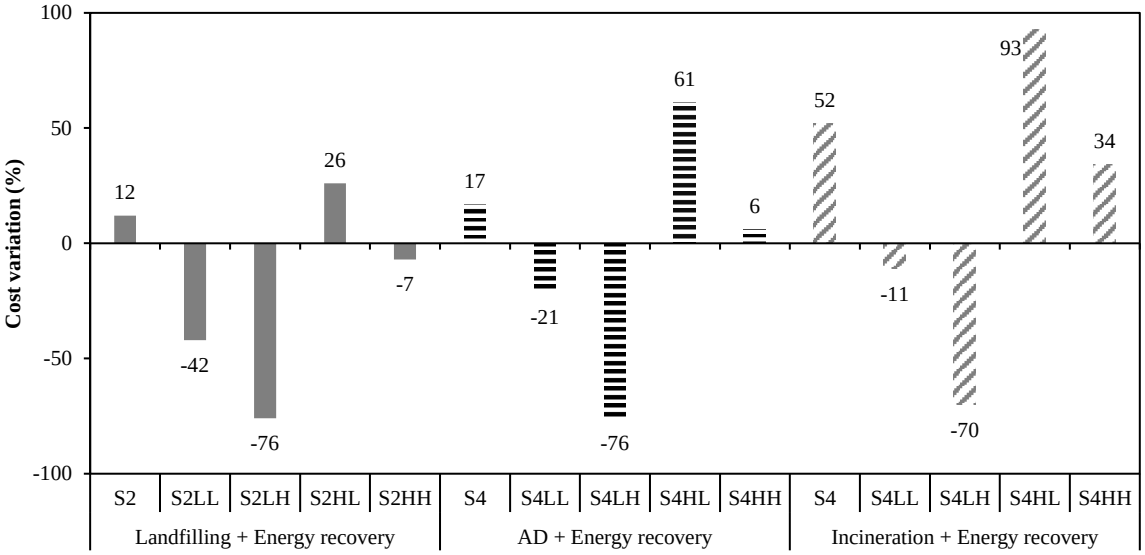


Figure 6. 7. Sensitivity to the cost of waste management processes and carbon credit (comparison with baseline scenario S1)

SLL: Process and carbon credit costs set at their lowest values
SLH: Process cost set at its lowest value *and carbon credit at its highest*
SHL: Process cost set at its highest value and carbon credit at its lowest
SHH: Process cost set and carbon credit set *at* their *highest* values
S1: Baseline scenario: Landfilling all waste with flaring
S2: Upgrade LFG capture system in S1 + energy recovery
S4: Max recycling & anaerobic digestion+ landfilling
S5: Incinerate all waste + energy recovery

It is worth mentioning that other externalities (e.g. real estate depreciation, air and groundwater pollution with potential health impacts) may affect the economic valuation of various scenarios. Another limitation is related to time factor considerations that affect both

costs and emissions (e.g. time required for the construction of different waste facilities). Moreover, current and future offsets of electricity were assumed similar.

6.4 Conclusion

A Life Cycle Assessment was applied in evaluating waste management alternatives towards defining optimal integrated systems. The highest environmental impacts were associated with scenarios that include landfilling with minimal material and energy recovery. Environmental benefits can be achieved under scenarios that maximize recycling and composting whereby savings in emissions reached up to 98%. Incineration with energy recovery reduced equivalent emissions most at a varying cost of –70% to +93% depending on the selected technology and the value of carbon credit. Despite the decrease in emissions, increased operational and investment costs favor alternatives that consider maximizing recycling and composting with residual landfilling when land is available. The sensitivity analysis suggested that greater savings in emissions can be achieved with improved landfill gas collection efficiency, application of produced compost and energy recovery during incineration. Finally, the breakeven analysis showed that maximizing material recovery and landfilling remains profitable under the entire range of carbon credit (0.5 to 50 US$/MT$CO_2$E). While the results provide guidelines for policy and decision makers on the economic viability of investment in carbon credit, potential changes in costs due to the dynamics of economy of scale and other externalities should be considered in the economic analysis.

CHAPTER 7
CONCLUSIONS

7.1 Major Conclusions

The goal of this dissertation was to develop a comprehensive model for decision support in quantifying and improving emissions estimation from integrated waste and wastewater/sludge management systems while considering upstream, operating, and downstream processes. The model provides guidelines towards a credible national inventory as well as a policy planning and decision making about process viability for investing in carbon credit. In addition to the current state of practice in developed economies, the model included emissions from waste management processes still practiced in developing economies (such as open dumping, open burning of waste, and poorly operated landfills with flaring systems as well as auxiliary fuel needed to satisfy the LHV during waste incineration) commonly not considered in most LCA models. It can disaggregate emissions by source (waste processes from collection to final disposal), or type (direct and indirect), or gas (CH_4, CO_2, N_2O) and offers users the flexibility to select processes or modify input parameters while examining their impact on uncertainty in model simulations. Equally important is a clarity in deriving and applying emission factors used to quantify emissions from waste management systems. A comparison of worldwide adopted methods for country accounting showed a wide variability reaching several folds depending on the source, gas or type of emissions. The observed variability can be attributed to differences between methods in approaches and default parameters. These differences can have serious implications on mitigation measures and reporting targets under the UNFCCC agreements or influence reduction targets using carbon credits to meet Nationally Determined Contributions under the Paris Agreement. The study

presents a framework model to address limitations in existing methods with emphasis on increased flexibility in allowing the user to modify default approaches and parameters.

The model was tested in the context of both developed and developing economies to assess the impact of waste composition, management processes, energy consumption and other input parameters on variations in emissions. A scenario analysis demonstrated that best practices through recycling, biological treatment, food waste diversion, and / or energy recovery can contribute to significant savings in emissions that ranged between 24 and 95%, depending on the tested system. In contrast, improper waste management such as open dumping or burning instead of controlled landfilling or incineration (with energy recovery) can increase the total equivalent emissions by ~30% and ~295%, respectively. This study revealed that integrating FWDs in a developing economy characterized with a high fraction of food waste can be a viable alternative solution to reduce emissions for carbon trading. The results indicated that adopting a FWD policy reduced emissions by about 42% at cost savings reaching ~28% when environmental externalities are considered including sludge management. While increasing the market penetration and the fraction of food waste ground contribute to a decrease in net emissions depending on wastewater and sludge management processes, the system remains economically attractive even when adding the wastewater and sludge management costs. The sensitivity analyses on processes with a wide range in costs showed an equivalent economic impact thus emphasizing the viability of a FWD policy although the variation in the cost of sludge management exhibited a significant impact on savings.

7.2 Recommendations for Future Work

In this work, a software was developed under a Matlab-based Graphical User Interface (GUI) that allows users to control the operation of the system. It provides a user-friendly platform with the flexibility to select processes or modify input parameters. It also encompasses a built-in Monte Carlo simulation to check on the variability in emissions by

varying key parameters. The software can provide tools for technical, economic, policy, and sensitivity analysis. The objective is to optimizing the life cycle of emissions or costs considering carbon credits to assist decision makers to allocate expenditures for emissions mitigation measures. Further improvements to the developed model include:

- Complementing the climate change impact category (GHG emissions) with other social and environmental impacts.
- Enhance the emission assessment component related to energy systems and leachate treatment.
- Development of a database to facilitate the model application.

APPENDIX A
MONOGRAM

Maalouf, A., El-Fadel, M. Optimizing Emissions and Carbon Credit from Integrated Waste and Wastewater Management: A MATLAB-based model with a Graphical User Interface. 2019. *Monogram in Environmental Modeling and Assessment.* American University of Beirut and Lebanese National Council for Scientific Research, Beirut Lebanon

A. 1. INTRODUCTION

Waste management models have been developed worldwide since the 1970s (Figure A.1) to assist decision makers in defining cost-effective and environmentally sound management alternatives. These models were established by a range of universities, environmental protection agencies, or consultancy firms, mainly in the US, Europe, and Canada, with limited applications in developing economies (Figure A.1).

Model	Location	'74	'75	'85	'94	'96	'97	'98	'99	'00	'02	'04	'05	'06	'07	'09	'10	'13	'14	'18	Source
Mathematical model	USA																				Walker et al. (1974)
Simulation model	USA																				Clark and Gillean (1975)
Computational model	GER																				Gottinger (1988)
CORINAIR 90	EU																				EEA (1994)
MIMES/WASTE	SW																				Sundberg et al. (1994)
IPCC-1996	Worldwide																				IPCC (1996)
MADS	CA																				Rubinstein (1997)
SWDSS	CA																				Huang et al. (1997)
LCA-LAND	DK																				Nielsen et al. (1998a,b)
MSWI	GER																				Ciroth (1998)
MSWFLOW	USA																				Haith (1998)
WARM	USA																				EPA (2006); EPA/ICF (2016)
Theoretical model	UK																				Daskalopoulos et al. (1998)
ARES	GER																				Schwing (1999)
EUGENE	GVA																				Berger et al. (1999)
IWM	CA																				Haight (1999, 2004)
WISARD	UK																				Ecobilan (1997)
IWM-2	UK																				McDougall (2001)
Life cycle inventory	USA EU																				Cambreco et al. (1999)
DST	USA																				Thorneloe et al. (2007)
HMA	FI																				Tanskanen (2000)
MWS model	SW																				Ljunggren (2000)
I-LCA	IT																				Baldo & Pretato (2001)
Decision Support Tool	USA																				Harrison et al. (2001)
MSW-DST	USA																				Solano et al. (2002a,b)
DG JRC	EU																				AOO (2002)
ORWARE	SW																				Dalemo et al. (1997); Eriksson et al. (2002)
Computer-based interface	LB																				Abou Najm and El-Fadel (2004)
SSWMSS	JP																				Tanaka et al. (2004); Tanaka (2008)
EpE	FR																				EpE (2013); Ranganathan et al. (2004)
LCA-IWM	EU																				Den Boer et al. (2005a, b ; 2007)
WASTED	CA																				Diaz and Warith (2006)
EASEWASTE	DK																				Kirkeby et al. (2006; 2007)
HOLIWAST	SW																				European Commission (2007)
IPCC-2006	Int.																				IPCC (2006)
WAMPS	SW																				Moora et al. (2006)
WRATE	UK																				Coleman (2006); Thomas& McDougall (2003)
Nordic Council's tool	Nordic																				Nordic Council of Ministers (2007)
FCM-PCP	CA																				Cadena (2009)
Konsta, Martti & Petra	FI																				Anderson et al. (2010)
SIWMS	AU																				El Hanandeh and El-Zein (2010)
Simulation model	IT																				Di Maria and Micale (2013)
SWOLF	USA																				Levis et al. (2013)
CO2ZW tool	SP																				Itoiz et al. (2013)
EASETECH	DK																				Clavreul et al. (2014)
SWW	LB																				Maalouf and El-Fadel (2018b)

Figure A.1. Reported waste models, tools, protocols, and guidelines

Comprehensive reviews (Gottinger, 1988; MacDonald, 1996) showed that early models focused on individual waste management processes (e.g. optimizing collection routes). In the 1980s, the focus expanded to cover municipal solid waste (MSW) management at the system level with interactions between various alternate processes (Tanskanen, 2000; Morrissey and Browne, 2004). The increasing complexity of integrated management modelling and the need for location-specific data, led to the inevitable independent development of existing models and subsequently a common lack of consistency and harmonization amongst models. In this context, Maalouf and El-Fadel (2019; 2018a) assessed the variability in estimating emissions from waste management when using commonly adopted international methods including the country level accounting with reference to the United Nations Intergovernmental Panel on Climate Change (IPCC), life cycle assessment (LCA) modelling, and organizational reporting. The assessment reflected a wide variability across methods in estimating total aggregated emissions when using default model parameters. Invariably, these methods depended on location-specific parameters where a particular method was developed. The IPCC Guidelines in particular were advocated as a common international ground under the United Nations Framework Convention on Climate Change (UNFCCC) and were intended to alleviate this limitation. However, the main parameters and waste-specific data remain largely not available for most countries with a frequent trend to still use data reported at locations/countries with different characteristics thus negating the very purpose for which the guidelines were developed. Moreover, the IPCC guidelines, do not consider indirect (upstream or downstream) contributions from processes within the waste management sector. These differences can affect emissions reporting targets under the UNFCCC commitments or reduction targets and decision making when relying on carbon credit to meet Nationally Determined Contributions (NDCs) under the Paris Agreement (United Nation, 2015) for instance.

Noteworthy, the IPCC guidelines consider emissions from waste and wastewater management systems under the same category but independently. In fact, none of the existing emissions' accounting models (Figure A.1) were designed to combine these systems or to evaluate policies that integrate them such as introducing a food waste disposer (FWD) policy at the household level for grinding food waste and diverting it towards the wastewater (WW) collection and management system. Such a policy was found in many cases to be an effective and economically viable alternative for waste and emissions reduction (Maalouf and El-Fadel, 2017; Yi and Yoo, 2014). This can be particularly important in developing countries where the food waste fraction can exceed 60% of the waste stream. As such, we proposed and implemented a modeling framework to address gaps discerned in existing methods by allowing the user to modify default parameters and by providing input data from both developed and developing economies (Maalouf and El-Fadel, 2017; 2018a, b; 2019).

In this study, we build on past experience and limitations to develop a MATLAB-based graphical user interface software towards optimizing emissions and carbon credit from integrated waste and wastewater management including several tools for technical, economic, and policy analysis on carbon trading. A built-in Monte Carlo simulation checks on the variability in emissions by varying key parameters with in-depth disaggregation of emissions by source (from collection to final disposal), or type (direct and indirect), or main gases (CH_4, CO_2, N_2O) and a user- flexibility to select processes or modify input parameters. In what follows, we describe the overall software development, input data, and new economic and policy analysis tools (Section 2). In Section 3, we demonstrate the model / software applicability in the context of developed and developing economies and in Section 4, we emphasize the specificities and novelties of the overarching model/software with potential policy implications and a comparative discussion of its advantages.

A.2. Material and methods

This section describes the overall software development methods, waste management processes, functioning of tools, and an application/demonstration example. Input data are presented as averages by default when lacking or modifiable by the user through a graphical interface. Interactions between various tools are depicted in Figure A.*2* and the tools are explained and elaborated below.

A.2.1. Software development

The software was developed under a Matlab-based Graphical User Interface (GUI) that allows users to control the operation of the system. Matlab is universally accepted as one of the most powerful data processing platforms. Its connectivity with many advanced programming languages (like C, Java, and VB) and availability of a wide range of toolboxes makes it popular among the scientific and research community (MATLAB, 2017). The software development can be divided into two phases: (1) hidden programming for data collection and model formulation based on Matlab code, and (2) interface initialization built and executed over the Matlab code using GUI tools. The interface allows the user to select data and input parameters as well as visualize outputs by displaying various forms of plots. The Matlab-based software provides an efficient way to operate and manipulate the data and automatically store results in excel files. The software is strengthened with a user-friendly and interactive GUI platform to perform easy operations and visualize tested scenarios towards optimizing emissions and costs of integrated solid waste and wastewater management systems. Figure A. 3 depicts the model / software user interface at the starting mode.

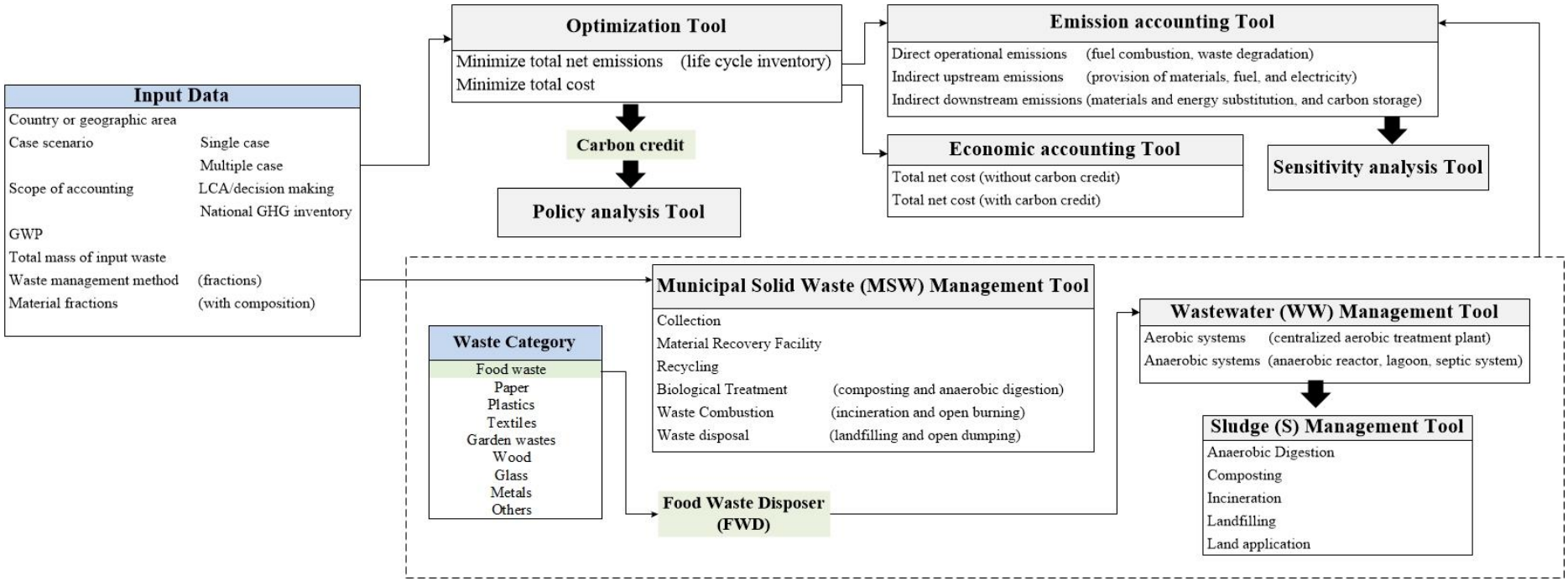


Figure A.2. Software structure with user input data and various interacting tools

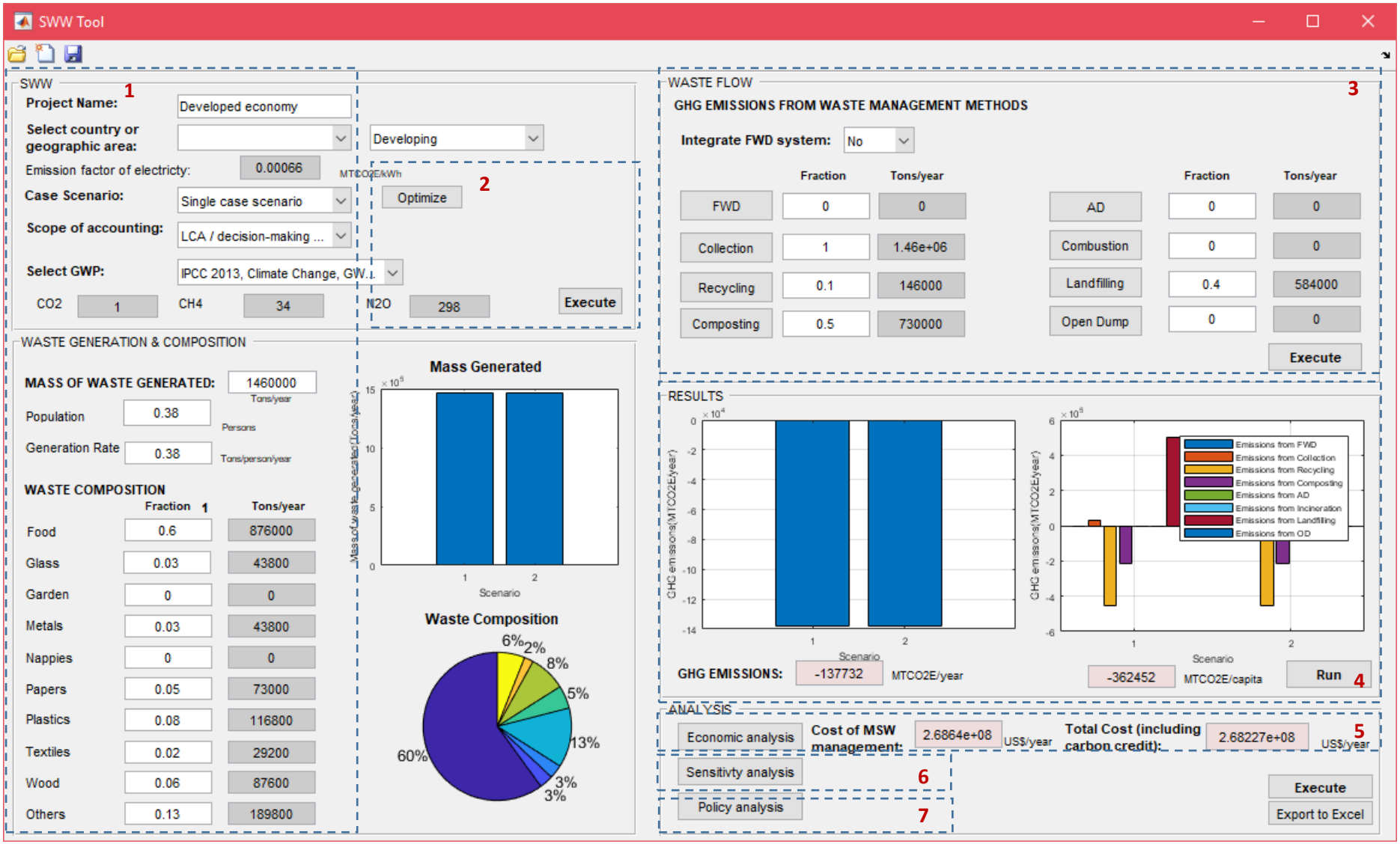


Figure A. 3. Software user interface

1: Data input; 2: Optimization tool; 3: Emission accounting tool; 4: Results display; 5: Economic analysis tool; 6: Sensitivity analysis tool; 7: Policy analysis tool

A.2.2. Input data

A.2.2.1. Country or geographic area

Some data such as the electricity generation mix (e.g. the share of coal, fuel oil, natural gas, nuclear, and renewable electricity generation) and fuel consumption are related to geographical conditions while others such as the performance of equipment at waste facilities (e.g. efficiency factors) are location-independent. Accordingly, it is imperative for the user to provide location-specific data to ensure representative results ("Part 1" in Figure A. 3). When data is not available, the model / software offers average default data for emission factors (EFs) of electricity depending on the selected geographic area or country (IEA, 2014).

A.2.2.2. Scenario definition

The user has the option to select whether to conduct: 1) a single case scenario that considers emissions' estimation from a predetermined waste management system; or 2) a multiple case scenario that considers a wide range of possible combinations to optimize the integrated waste and wastewater management system based on minimum emissions or costs (see Section 2.5) while considering the carbon credit ("Part 1" in Figure A. 3).

A.2.2.3. Scope of accounting

The model / software disaggregates emissions by type (direct or indirect), which allows the user to select the scope of reporting whether for national GHG inventories (accounting for direct emissions) or LCA/planning and decision-making purposes (accounting for direct and indirect emissions) ("Part 1" in Figure A. 3). Therefore, the results of total emissions are displayed in the main window ("Part 4" in Figure A. 3) according to the selected scope of accounting.

A.2.2.4. GWP

The global warming potential (GWP) comprises a GWP_{20}, GWP_{100} and GWP_{500}, for a time horizon of 20, 100 and 500 years respectively (Solomon et al., 2007). All reporting mechanisms use GWP values provided by the IPCC based on the effects of GHGs over a 100-year time horizon (GWP100). The latter has evolved three times since the Second Assessment Report (SAR) published by the IPCC (1995) until the last one (Fifth Assessment Report-AR5) published in 2013 due to improvements in calculations and an increase in the amount of atmospheric GHGs during this period. For instance, the GWP100 of methane, which is the most significant in the waste sector, has increased by around 62% since 1995.

Selecting the GWP reference can vary depending on the scope of reporting. For national GHG inventories under the UNFCCC, all reporting protocols from 2015 use the GWP_{100} of the Fourth Assessment Report (AR4) (IPCC, 2007). Under the Kyoto Protocol reporting period, the SAR GWP_{100} was used by all reporting parties to ensure a common GHG metrics for all trading mechanisms and UNFCCC reporting targets. However, for LCA/planning and decision-making purposes, the most scientifically updated reference (e.g. GWP_{100} from IPCC AR5 in 2013) is used.

Regardless of the scope of reporting, it is essential to indicate the time horizon (e.g. 20, 100, 500 years) and the reference of the GWP used to ensure transparency (Gentil et al., 2009). In this context, when values are not available, the model / software provides default GWP100 values based on IPCC references (e.g. IPCC, 1990, 1995, 2001, 2007, 2013) ("Part 1" in Figure A. 3). Equally important is that EFs (e.g. $MTCO_2E$/tonne of waste managed) used in the intermediary calculations of the model, which are linked automatically to the GWP reference selected by the user to ensure a consistent reporting of emissions.

A.2.2.5. Waste generation and composition

Data related to waste generation and composition constitutes the starting point for calculating emissions and costs. The total amount of waste generated (tonnes/year) is provided by the user or extrapolated from the population (persons/year) based on per capita generation rate (tonne/person/year) for a general study area and inventory year ("Part 1" in Figure A. 3). The user also enters the waste composition fractions (food, glass, garden, metals, nappies, papers, plastics, textiles, wood, etc.) for estimating emissions.

A.2.3. Emission accounting tool

The modelling domain integrates the MSW and WW management systems under a single framework upon introducing a FWD policy at the household level for grinding food waste and diverting it into the WW collection system. In this context, the model/software accounts for emissions from various MSW management processes including collection, sorting/recycling, biological treatment (e.g. composting and anaerobic digestion), incineration (with and without energy recovery), landfilling (with and without landfill gas collection for flaring or energy recovery), open dumping[17] or burning1. The WW management system may consist of aerobic (e.g. centralized aerobic treatment plant) or anaerobic processes (e.g. anaerobic lagoon, septic system) with several sludge management (SM) options including anaerobic digestion, composting, landfilling, incineration, or land application.

When the user selects a single case scenario option, the model offers emission accounting tools to calculate emissions from individual waste management processes. The user first defines in the main window the amount (tonnes /year) or fraction of MSW managed under each process ("Part 3" in Figure A. 3). Figure A. *3* shows an example for modelling emissions

[17] Still commonly practiced in developing economies

from landfilling by using the "process-specific tool" template. Additional screenshots of the interface are displayed for individual process-specific tools (Figures Figure A.*4* Figure A.10). The model keeps track of all the mass and material flows specific for each process ("Part a" in Figures Figure A.*4* Figure A.10). It also offers the flexibility of allowing the user to select or modify process specific-input parameters. For instance, for the case of landfilling the user enters specific data related to fuel and material provision for the landfill construction; amount of electricity and fuel consumed by onsite-daily operating equipment; fraction of landfill gas (LFG) collected (for flaring or energy recovery); as well as the net electricity conversion efficiency. Default averages are provided when data is not available. The process-specific emissions are disaggregated by type of emissions (direct-operating, indirect-upstream, indirect-downstream) and gas (CH_4, CO_2, N_2O) also displayed graphically to allow the user to visualize the independent percentage contribution from direct and indirect processes or from each gas to the total emissions ("Part b" in Figures Figure A.*4* Figure A.10). In this context, the net total emissions from waste management processes are estimated in metric tonnes of CO_2 equivalents ($MTCO_2E$) and equal to the difference between gross[18] and avoided[19] emissions. Indirect upstream emissions arise from inputs of materials (e.g. provision of material for landfill construction); electricity provision (emissions occur offsite and depend on the current electricity generation mix selected by the user); and fuel provision (pre-combustion emissions associated with the extraction, processing, producing, storage, and transport of fuel). Direct operational emissions from system's operation are related to fuel combustion of onsite operating equipment and waste degradation as a result of physical, chemical, or biological processing (e.g. LFG emissions). Indirect downstream emissions (or savings) are associated to

[18] Indirect-upstream and direct-operating
[19] Indirect-downstream

avoided emissions from energy generation (depending on the selected electricity generation mix), materials substitution (e.g. recyclable materials that offsets production from virgin materials), and carbon storage.

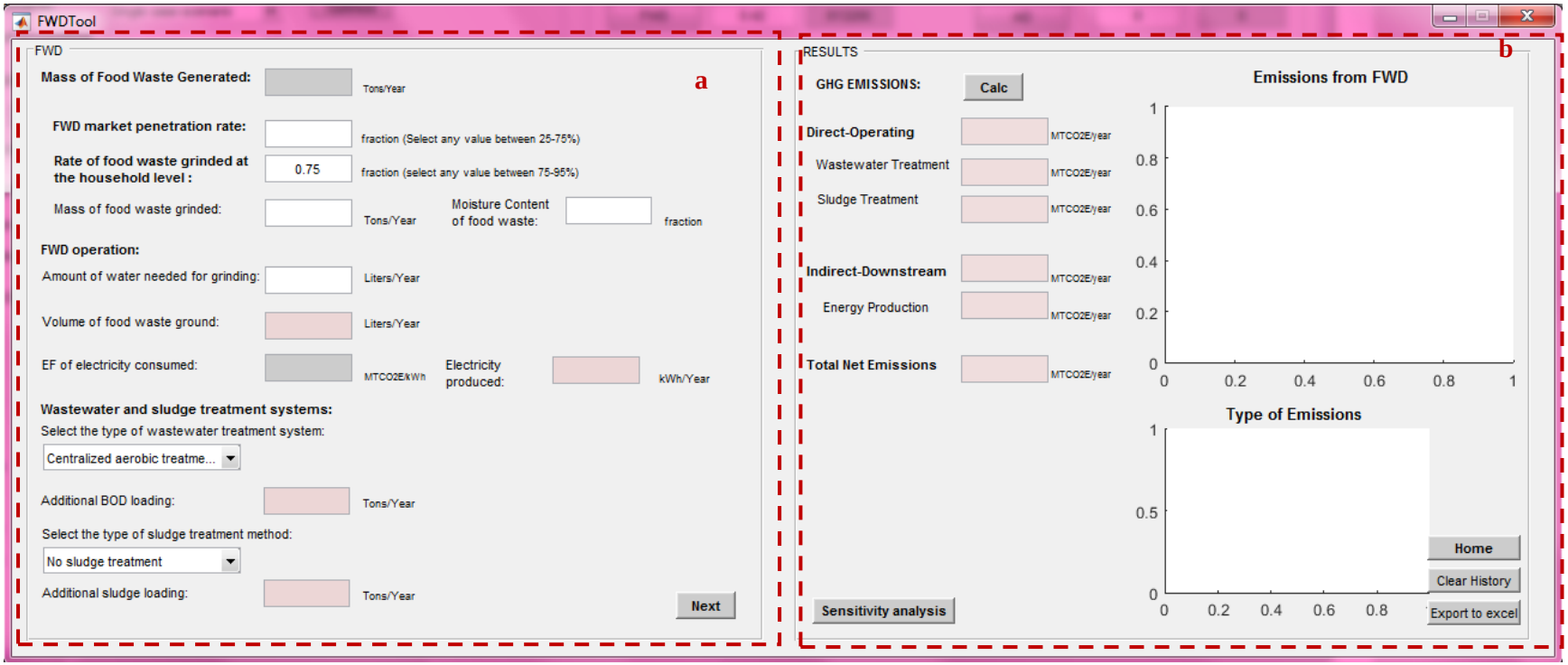


Figure A.4. Food waste disposer (FWD) tool
a: Input-specific data; b: Process-specific emissions results

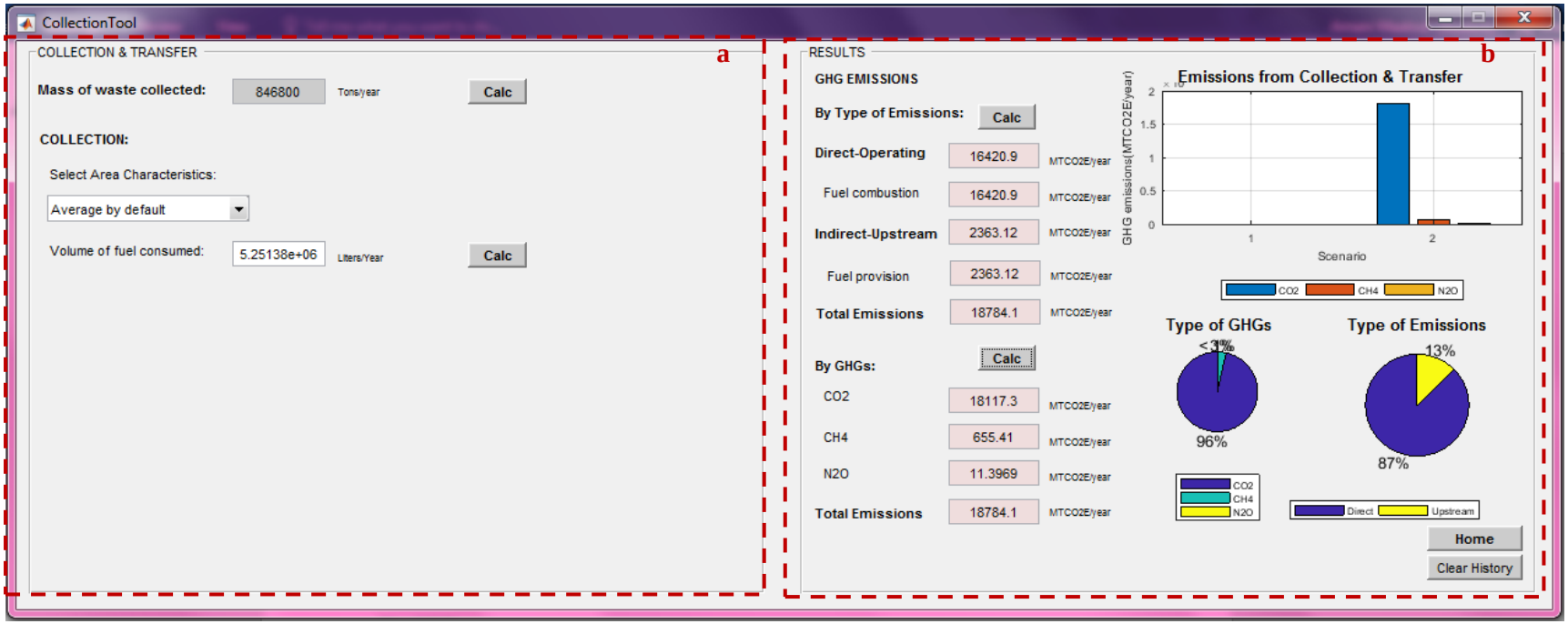


Figure A.5. Waste collection tool
a: Input-specific data; b: Process-specific emissions results

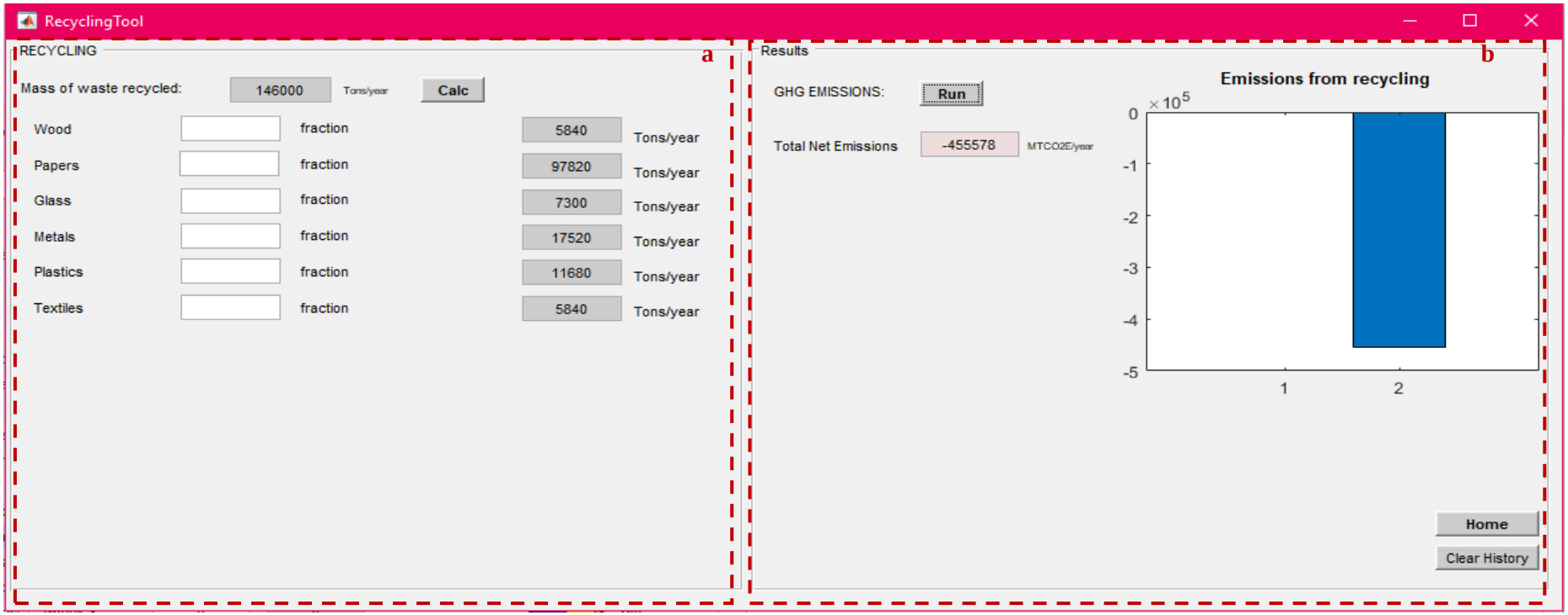


Figure A.6. Recycling tool
a: Input-specific data; b: Process-specific emissions results

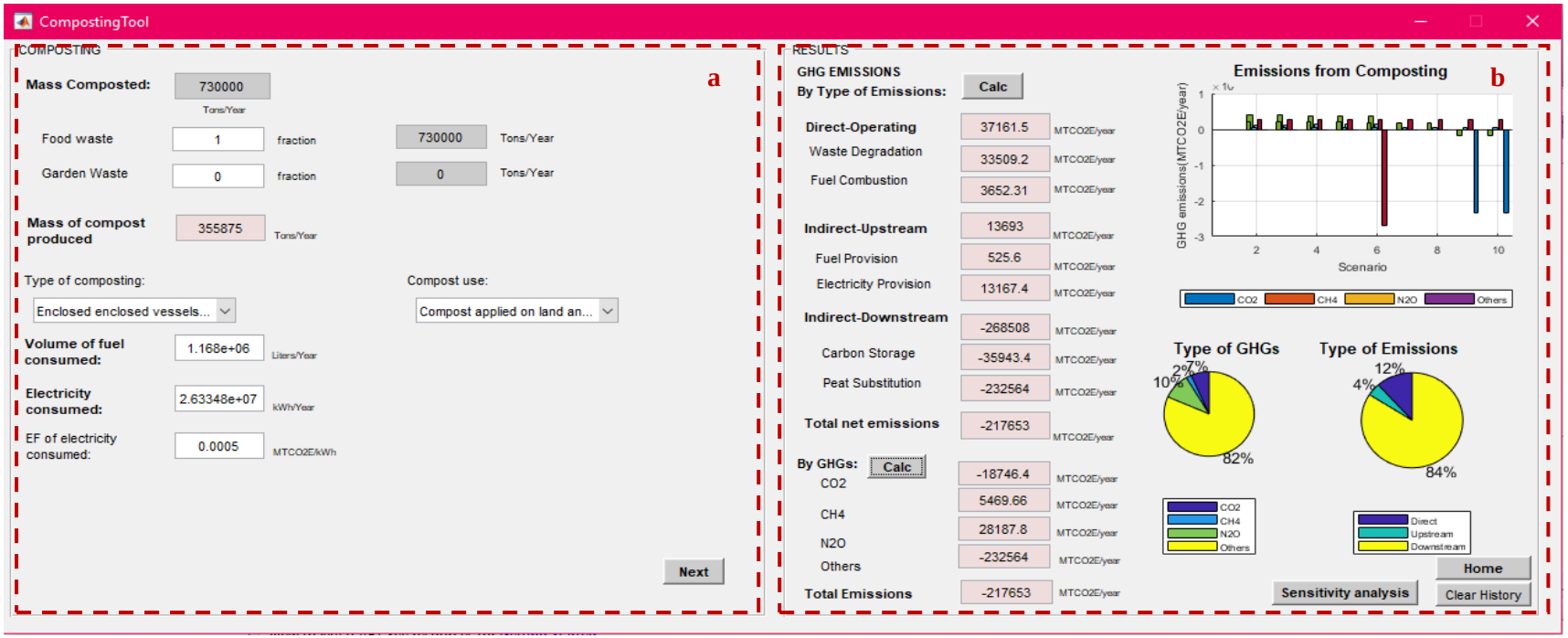


Figure A.7. Composting tool
a: Input-specific data; b: Process-specific emissions results

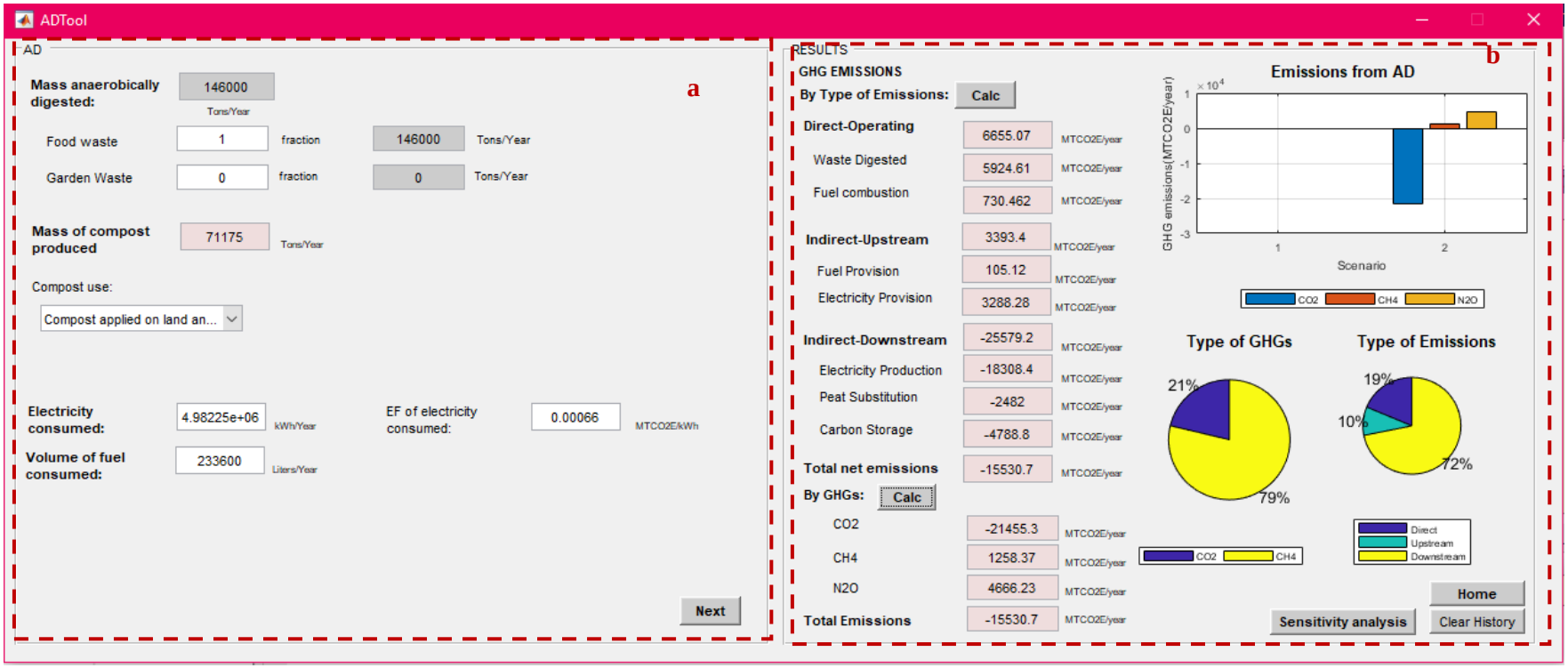


Figure A.8. Anaerobic digestion (AD) tool
a: Input-specific data; b: Process-specific emissions results

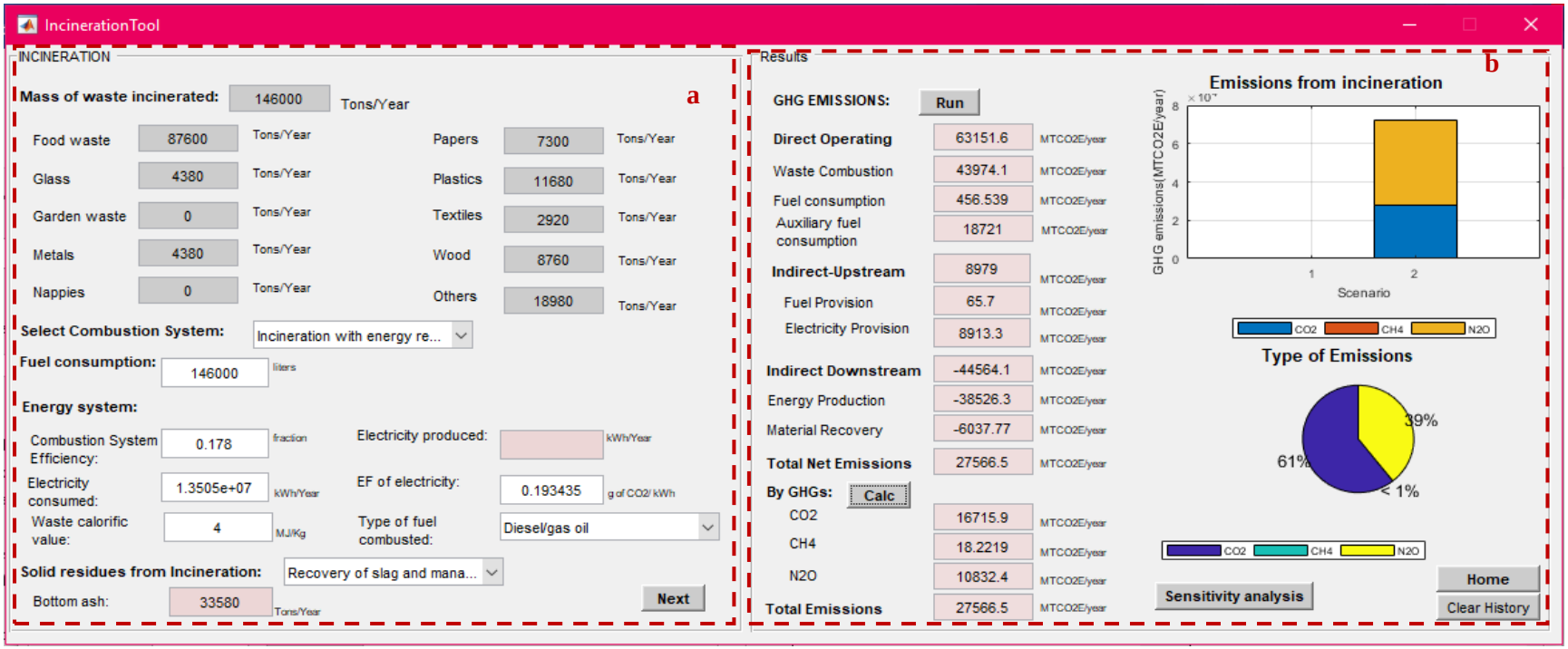


Figure A. 9. Incineration & Open burning tool
a: Input-specific data; b: Process-specific emissions results

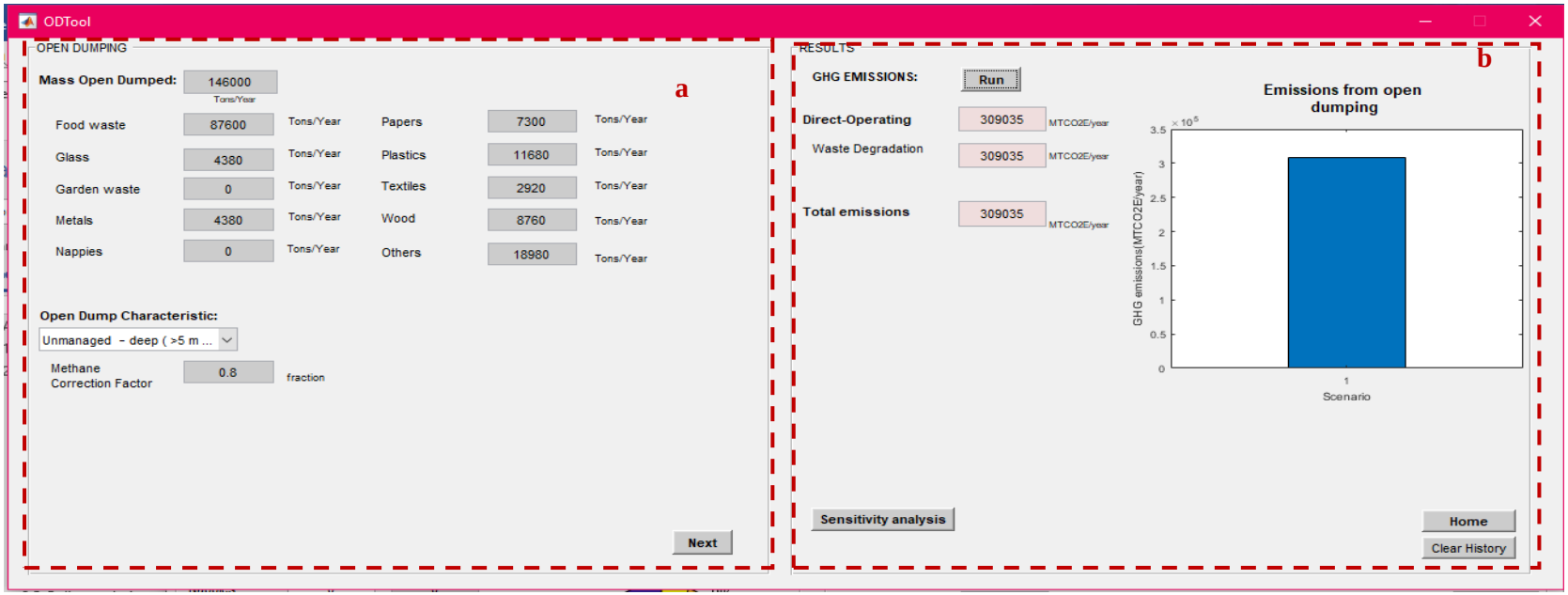


Figure A.10. Open dumping tool
a: Input-specific data; b: Process-specific emissions results

A.2.4. Economic analysis tool

Economic associations targeted the analysis of conventional (direct) and environmental (indirect) costs-savings for tested waste management systems. The conventional costs include capital and operating costs associated with waste management processes. The model provides default average operating costs (US$ per tonne of waste) of these processes (Maalouf and El-Fadel 2017) if data is not available ("Part a" in Figure A. 11). The user can enter capital costs of new facilities that are considered as part of a new waste management decision with the exception of landfilling whereby capital (e.g. construction) costs are amortized into their operating costs because they are considered as an ongoing construction process. The cost of MSW management is estimated by multiplying the average costs (US$ per tonne) of alternatives by the total amount of waste managed ("Part b" in Figure A. 11). The model also allows the user to visualize tested scenarios and shows the percentage contribution of each waste management process to the total cost ("Part b" in Figure A. 11). The cost of introducing FWDs includes (1) capital/operating costs, (2) costs of managing additional wastewater and sludge loads, and (3) the cost of increased consumption of domestic water for grinding the food waste ("Part a" in Figure A. 11) with electricity cost for operation of FWDs being negligible. Environmental savings comprised costs forgone due to the decrease in requirements for managing food wastes diverted from the waste stream such as leachate and gas management (Maalouf and El-Fadel 2017).

The offset of emissions was quantified based on the carbon market. The latter ranged from 0.5 to 50 US$ per $MTCO_2E$ in 2016, with an average price of 3 US$/$MTCO_2E$, which is the lowest reported market value from voluntary actors since 2006 (Ecosystem Marketplace, 2017). The model allows the user to define the average price or to select from different values reported by the Ecosystem Marketplace (from 2010 to 2017) ("Part a" in Figure A. 11). The

average value is used to assess associated benefits and allows the estimation of minimal savings when the carbon footprint is reduced through regulated and voluntary global markets for offsetting of carbon credits.

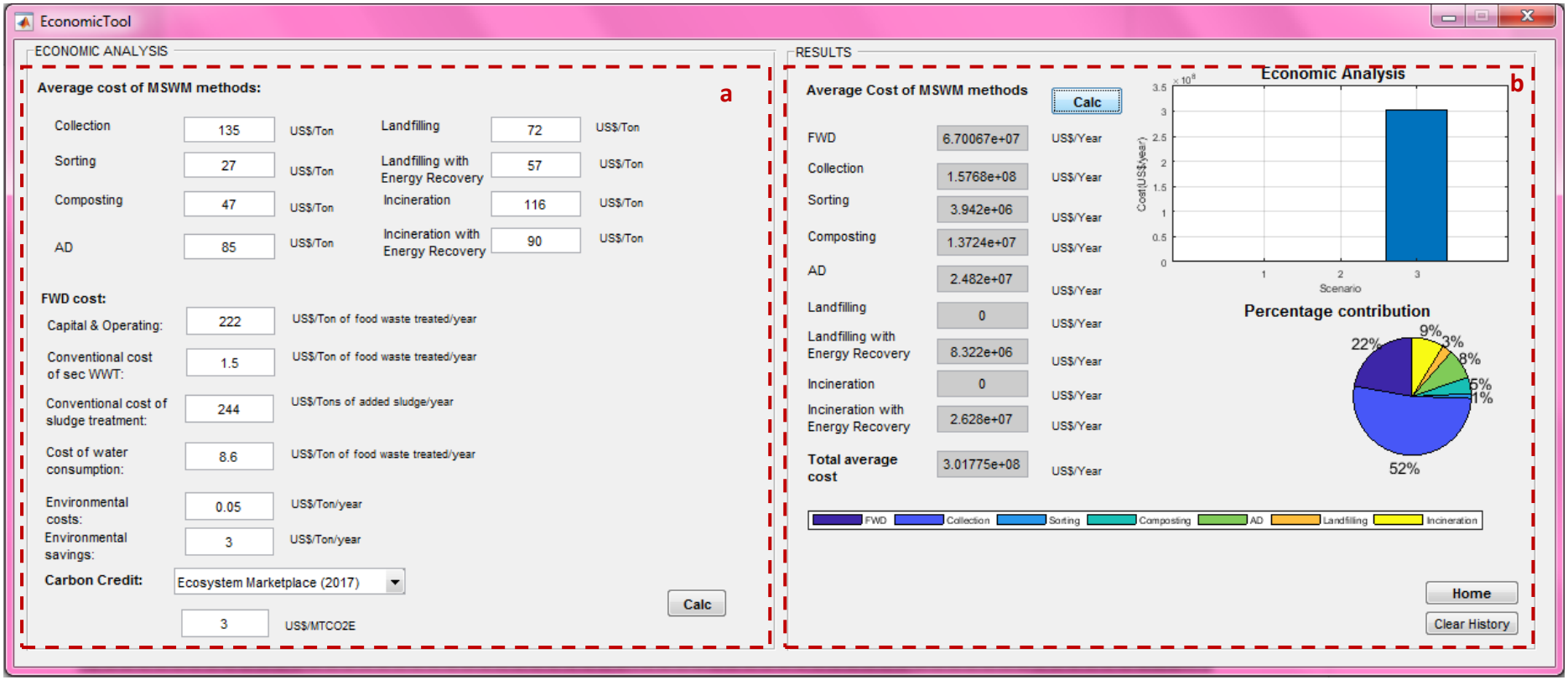


Figure A. 11. Economic analysis tool
a: Average costs of municipal solid waste management processes (MSW) and costs of food waste disposer (FWD); b: Costs results

A.2.5. Optimization tool

The model offers an optimization tool using linear programming (LP) to provide decision-makers with optimum integrated waste management systems for any region. The flow network of the waste stream is divided into two routes for the MSW and WW streams (Figure A. 12). The structure allows the model to optimize following a life cycle inventory approach, while considering economic implications including carbon credit, costs of future MSW and WW processes, and related policies. The user has the option to conduct the optimization based on minimum total emissions or minimum cost with consideration to carbon credit (Figure A.13).

Depending on the inventory year (t), the formulation of the objective function calculates the minimum total net emissions (Equation 1) or total costs (Equation 2) of waste management processes. In this context, the total net emissions, ET, during an inventory year (t), is the summation of emissions from all MSW management processes from waste collection, recycling, composting, incineration, landfilling, open dumping, and open burning that are defined as C, R, Co, I, Lf, OD, and OB as well as emissions from WW and S management, defined as FWD, which are associated with introducing a FWD policy. On the other hand, the total costs, CostT, is the summation of MSW costs from collection to landfilling and the cost of introducing a FWD policy, assuming no costs for open dumping or burning of waste.

$$Minimize\ E_T = \sum_{k=FWD}^{OB} E_k\,; \quad k \in \{FWD;\ C; R; Co; AD; I; Lf; OD; OB\} \tag{1}$$

$$Minimize\ Cost_T = \sum_{k=FWD}^{Lf} Cost_k\,; \quad k \in \{FWD;\ C; R; Co; AD; I; Lf\} \tag{2}$$

Where

E_T Total net emissions from MSW and WW management system in inventory year *t* ($MTCO_2E/yr$)
E_K Emissions for waste management method *k* [Food waste disposer (FWD), collection (C), recycling (R), composting (Co), anaerobic digestion (AD), incineration (I), landfilling (Lf), open dumping (OD), and open burning (OB)] in inventory year t ($MTCO_2E/yr$)
$Cost_T$ Total costs from MSW and WW management system in inventory year *t* (US$/yr)
$Cost_K$ Costs for waste management method *k* [Food waste disposer (FWD), collection (C), recycling (R), composting (Co),

anaerobic digestion (AD), incineration (I), landfilling (Lf)] in inventory year t (US$/yr)

The decision variable selected in the optimization process consists of the amount of waste (M_k) sent to a management alternative (k) that is in function of the fraction of waste, f_k,, managed under a method k multiplied by the total waste generated (M_T) (Equations 3 and 4).

$$M_k = f_k * M_T \quad \sum_{K=R}^{Lf} f_k = 1\,;\; k \in \{FWD; R; Co; AD; I; LF; OD; OB\} \tag{3}$$

$$f_{k\,min} \leq f_k \leq f_{k\,max} \tag{4}$$

Where

M_T Total mass of waste generated in year t (Tons/yr)
f_k Fraction of waste under management method k
$f_{k\,min}$ Minimum fraction of waste under management method k
$f_{k\,max}$ Maximum fraction of waste under management method k
M_K Mass of waste managed under method k [food waste disposer (FWD), recycling (R), composting (Co), anaerobic digestion (AD), incineration (I), landfilling (Lf), open burning (OB), or open dumping (OD)] in year t (Tons/yr)

The model constraints include mass balance, material limitations, capacity and policy implementations. The mass balance necessitates that the amount of waste distributed to various management alternatives must equal the amount of waste generated. At the material level, not all the amount of waste reaching a processing facility can be biologically treated for instance. Accordingly, the user has to define the minimum and maximum fraction that can be diverted to biological treatment (composting or anaerobic digestion), depending on the degradable organic fraction (Figure A.13). Similarly, the optimum integrated waste management system can be constrained through policy-implementation whereby waste separation at source can be imposes through minimum and maximum targets. For instance, the fraction of waste diverted to the WW stream when using a FWD is dependent on the amount of food waste grinded at the household level that typically ranges between 75 to 95% and the FWD market penetration that varies between 25 to 75% (Maalouf and El-Fadel, 2017). Note that these ranges are based on literature reported values/guidelines and past experience (Galil and Yaacov, 2001; Wainberg et al., 2000), yet, the model is not constrained by these ranges and the user can select any value according to the targeted policy.

Similarly, the fraction of waste recycled and biologically treated can be encouraged through policies setting diversion targets to reduce landfilling. The user can eliminate an alternative through regulatory compliance. Economic constraints are equally considered by increasing diversion to certain processes to maximize gains or minimize costs. Users can also allocate minimum capacity constraint for specific processes to ensure the operation of an existing facility. For instance, an incineration facility usually requires a minimum utilization fraction of 80% to operate efficiently (Levis et al., 2013). Accordingly, user-specified constraints can be introduced by setting minimum (fk min) and maximum (fk max) fraction of waste under a management method k (Equation 4) to optimize specific policies (Figure A.13).

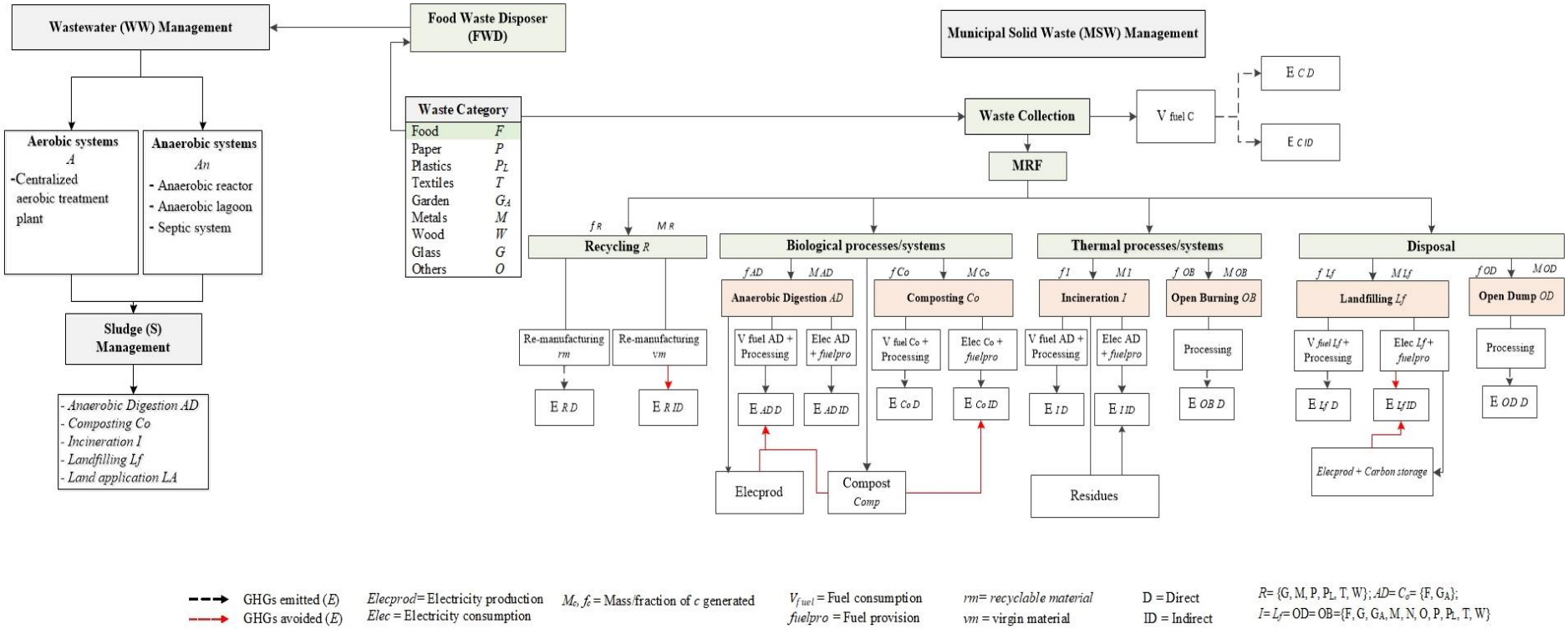


Figure A. 12. Solid waste and wastewater stream flow network

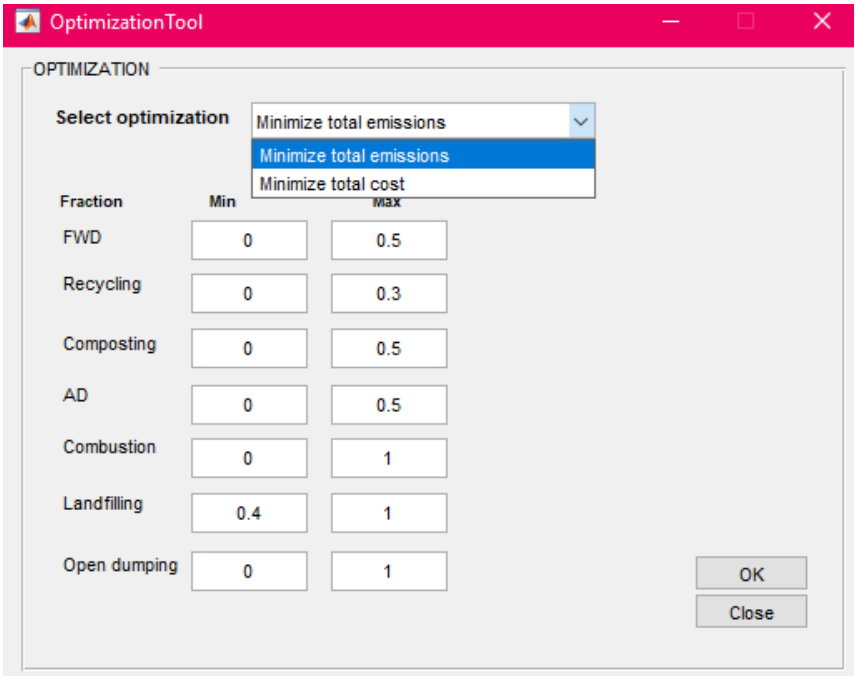


Figure A.13. Optimization tool

A.2.6. Policy analysis tool

The decision analysis for integrated waste management is multi-objective whereby progress in one way might hinder progress in others. For instance, open dumping or burning of waste appear as best alternatives in terms of cost minimization, yet, they have the highest potential emissions impacts. Therefore, decision-makers should trade-off costs in one area against benefits in another or vise-versa. In this context, the model addresses the problem of multi-objectivity by considering environmental valuation in the form of carbon externalities. Accordingly, the carbon credit expressed in US$ per $MTCO_2E$ were assigned to environmental emissions. The ultimate objective is to evaluate scenarios based on minimizing total net emissions or costs while considering implications in terms of carbon credit for both cases (Figure A. 14). This can be particularly important at the policy planning level whereby it can influence reduction targets or affect mitigation measures using carbon credits to meet NDCs under the Paris Agreement.

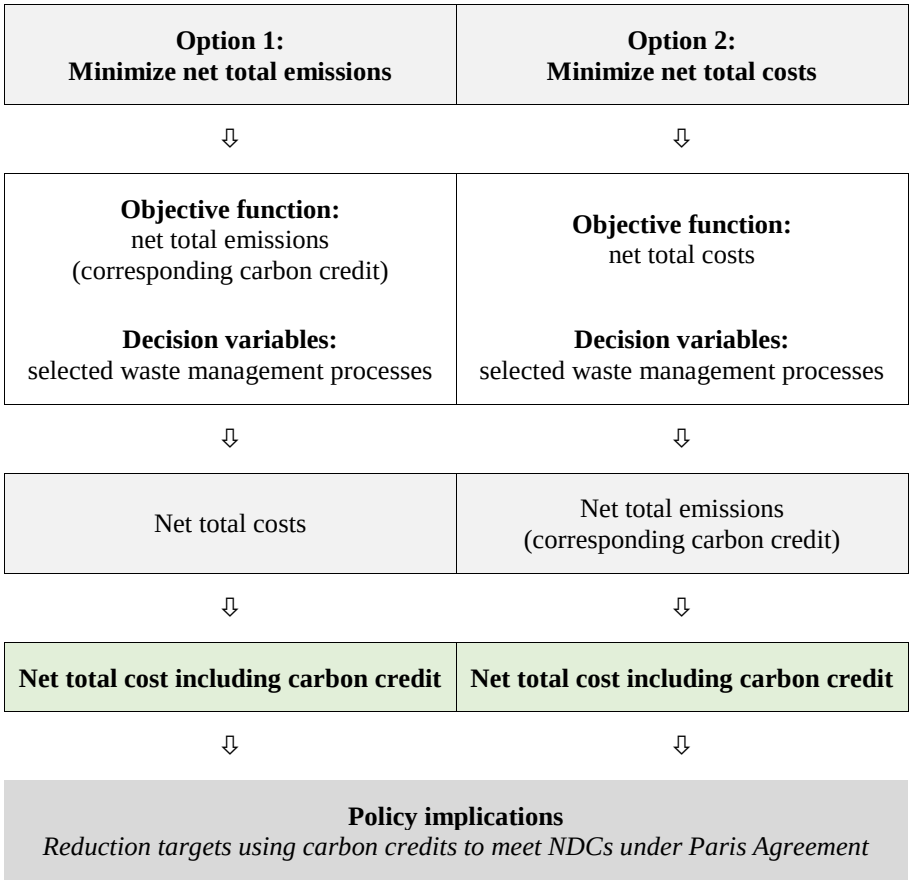


Figure A. 14. Policy implications in terms of carbon credit based on optimizing total net emissions or costs

NDC: Nationally Determined Contributions

A.2.7. Sensitivity or uncertainty analysis tool

The model allows the user to select key parameters for sensitivity and uncertainty assessment whereby each parameter can be individually modified to assess its impact on emissions by following two methods: 1) The One-at-a-time (OAT) sensitivity analysis whereby the user specifies the percent increase or decrease of the initial value of the parameter with the results displayed as percent change in emissions to assess the influence of each parameter based on the same initial variation, or 2) Monte Carlo analysis whereby parameters are simulated as a normal distribution around their means with a standard deviation of 5% (or 95% confidence interval at +/-10%) with the results displayed graphically as depicted in Figure A. 15.

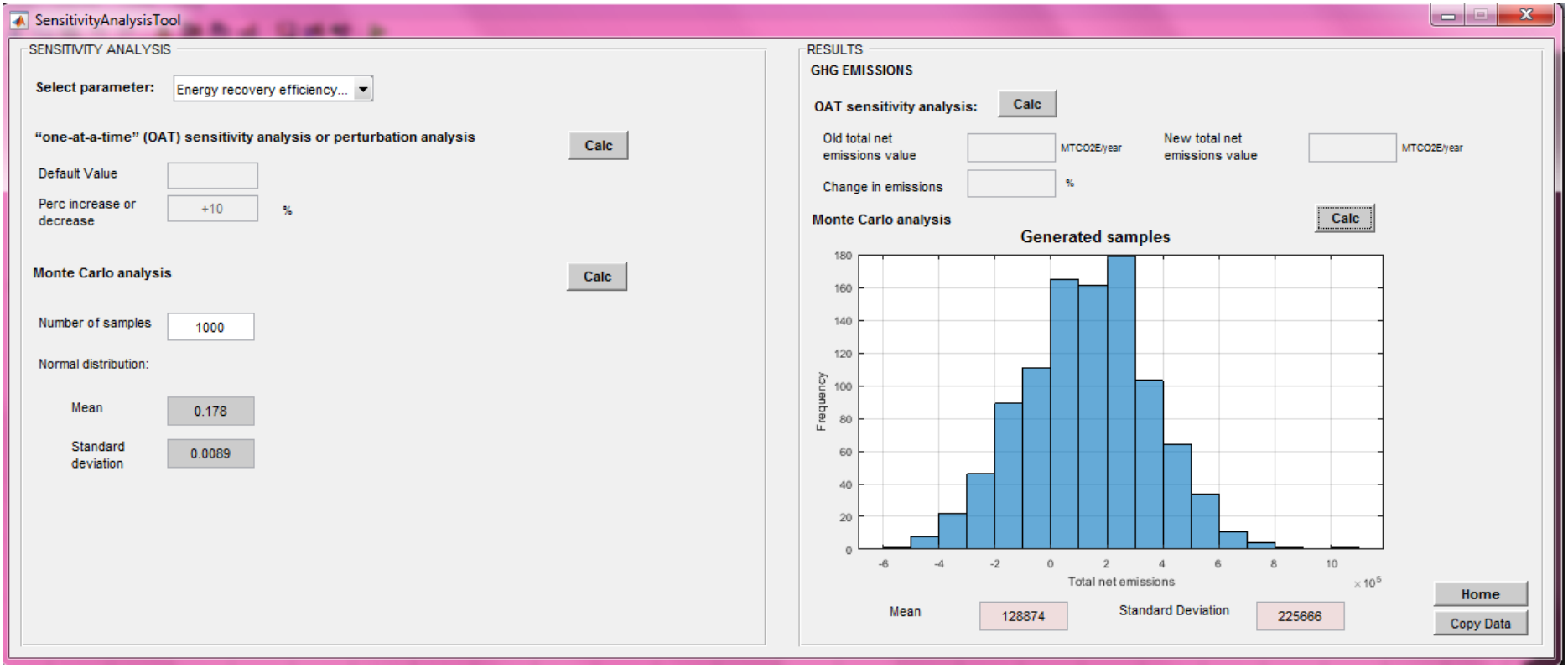


Figure A. 15. Sensitivity analysis tool

A.2.8. Application – Software Demonstration

We present a demonstration of the model/software on how optimization can be performed and what type of results can be extracted from it under two cases. In the first case, we considered optimizing an integrated waste management system with the objective to minimize total net emissions in the context of developed and developing economies. The second case targeted minimal total net costs. The scenarios were then evaluated when considering carbon credit. The variation in waste composition between developed and developing economies is reflected in the analysis. The specifications used to define the optimization model constraints include:

- An amount of waste that can reach 4000 tonnes per day (Table A.1 presents all input data). The results are presented on the basis of 1 tonne of waste managed.
- At 75% market penetration and 95% of food waste ground, up to 42% of MSW in a developing economy and 21% in a developed economy can be diverted through a food waste disposer (FWD) to the wastewater (WW) stream for aerobic treatment while sludge is treated anaerobically;
- Similarly, up to 42% of MSW in a developing economy and 21% in a developed economy can be biologically treated (anaerobic digestion or composting) while considering savings in emissions from land application of produced compost due to substituting the production of mineral fertilizers;
- Up to 13% of MSW in a developing economy and 23% in a developed economy can go to recycling;
- Incineration requires more than 80% of its capacity to operate efficiently (Levis et al., 2013);
- Waste can be landfilled with 60% of landfill gas (LFG) for flaring or energy recovery; and

– Open dumping or burning were excluded from the comparative analysis since they are not expected to be practiced under a developed economy.

Table A.1. Average model input data for developing vs developed economies scenarios

Input Parameter	Value		Reference
Waste composition (%)	**Developed economy**	**Developing economy**	World Bank (2012); IPCC (2006)
	– Food (30) – Papers (31) – Plastics (11) – Textiles (3) – Wood (5) – Glass (7) – Metals (6) – Others (7)	– Food (60) – Papers (5) – Plastics (8) – Textiles (2) – Wood (6) – Glass (3) – Metals (3) – Others (13)	
Waste data Total mass of waste generated (tonnes/year)	1		
Global warming potential-GWP$_{100}$	$CO_{2\ biogenic}$=0; CO_2=1 CH_4=34; N_2O=298		IPCC (2013)
Energy Emission factor for electricity grid mix ($MTCO_2E$/kWh)	**Developed economy** $4x10^{-4}$ [(a)]	**Developing economy** $6.6x10^{-4}$ [(b)]	IEA (2014)
Emission factor for fuel provision $EF_{fuelpro_{CO_2}}$(extraction, processing, storage, and transportation of the fuel) ($MTCO_2E$ /L of diesel fuel)	$EF_{fuel_{CO_2}} = 4.5x10^{-4}$		Fruergaard *et al.* (2009)
Emission factor for CO_2, CH_4, and N_2O ($EF_{fuel_{CO_2}}$, $EF_{fuel_{CH_4}}$, $EF_{fuel_{N_2O}}$) from fuel combustion ($MTCO_2E$ /L of diesel fuel)	$EF_{fuel_{CO_2}} = 0.003$ $EF_{fuel_{CH_4}} = 1.2x10^{-4}$ $EF_{fuel_{N_2O}} = 2.2x10^{-6}$		Fruergaard *et al.* (2009) McDougall *et al.* (2001) McDougall *et al.* (2001)
Waste treatment facilities			
Collection			
Fuel consumption (L/tonne)	6		Larsen et al. (2009); Tanskanen and Kaila (2001)
Cost (US$/tonne)	135		Maalouf & El-Fadel (2017)
Composting (open windrow + land application of produced compost)			
Electricity consumed (kWh/tonne)	32		McDougall et al. (2001)
Fuel consumption (L/tonne)	3.2		Boldrin et al. (2009)
Cost (US$/tonne)	47		Maalouf & El-Fadel (2017)
Anaerobic Digestion (land application of produced digestate)			
Electricity consumed (kWh/tonne)	35		Møller et al. (2009)
Fuel consumption (L/tonne)	1.6		Møller et al. (2009)
Cost (US$/tonne)	85		Maalouf & El-Fadel (2017)
Incineration			
Efficiency of combustion process (%)	17.8		Di Maria et al. (2018)
Waste calorific value (MJ/kg)	5.5		Zhao et al. (2012)
Electricity consumed (kWh/tonne)	92.5		Astrup et al. (2009)
Fuel consumption (L/tonne)	1		Astrup et al. (2009)
Cost of incineration with energy recovery (US$/tonne)	90		Maalouf & El-Fadel (2017)
Cost of incineration with no energy recovery (US$/tonne)	116		Maalouf & El-Fadel (2017)
Landfilling			
Amount of material used during landfilling (tonnes /yr)	Cells: 0.001 Drainage system: 0.1		Manfredi et al. (2009)

Input Parameter	Value	Reference
Electricity consumed (kWh/tonne)	7	Manfredi et al. (2009)
Fuel consumption (L/tonne)	2	Manfredi et al. (2009)
Landfill gas collected for energy recovery (%)	60	EPA/ICF (2016)
Efficiency of electricity generated (%)	85	EPA/ICF (2016)
CH_4 energy content (kW h/Tons of CH_4 generated)	4325	EPA/ICF (2016)
Cost of landfilling with energy recovery (US$/tonne)	57	Maalouf & El-Fadel (2017)
Cost of landfilling with no energy recovery (US$/tonne)	72	Maalouf & El-Fadel (2017)

[(a)]Considered high income countries (e.g. OECD region)
[(b)]Considered low income countries (e.g. Africa region)

A.3. Results and discussion

A.3.1. Case study under a developed economy condition

Table A.2 presents the total net emissions and cost variation with and without carbon credit for each case on a per tonne of waste managed under a developed economy condition. It also provides a description of the resulting optimal integrated waste management scenarios with minimal emissions or costs, depending on the selected objective of the optimization, with corresponding MSW fractions (variables) under each waste management process. Figure A.16 depicts the simulation results of scenarios under the developed economy condition with corresponding contribution to total net emissions, further disaggregated by source (e.g. FWD, collection, recycling, composting, incineration, or landfilling), gas (e.g. CO_2, CH_4, N_2O), and type (e.g. direct-operating, indirect-upstream, indirect-downstream).

Table A.2. Total net emissions and cost (with and without carbon credit) in the context of a developed economy condition

Case description	Scenario	Percent waste distribution by management process	Total net emissions ($MTCO_2E$/tonne of waste managed/yr) [a]	Emissions variation (%)	Cost variation without carbon credit (%)	Cost variation including carbon credit (%)
Baseline condition [b]	S1	20% Recycling, 12% composting, 25% incineration, and 43% landfilling	-0.27			
Minimize emissions	S2	20% Recycling, 20% FWD, and 60% incineration with energy recovery	-0.52	-93	11	10
Minimize cost	S3	20% Recycling and 80% landfilling	-0.02	92	-4	-3

[a] A negative value indicates net savings from avoided emissions due to material recycling (remanufacturing), energy recovery (incineration, landfilling), substituting fertilizers production from the application of compost on land (composting and anaerobic digestion), and carbon storage (landfilling).

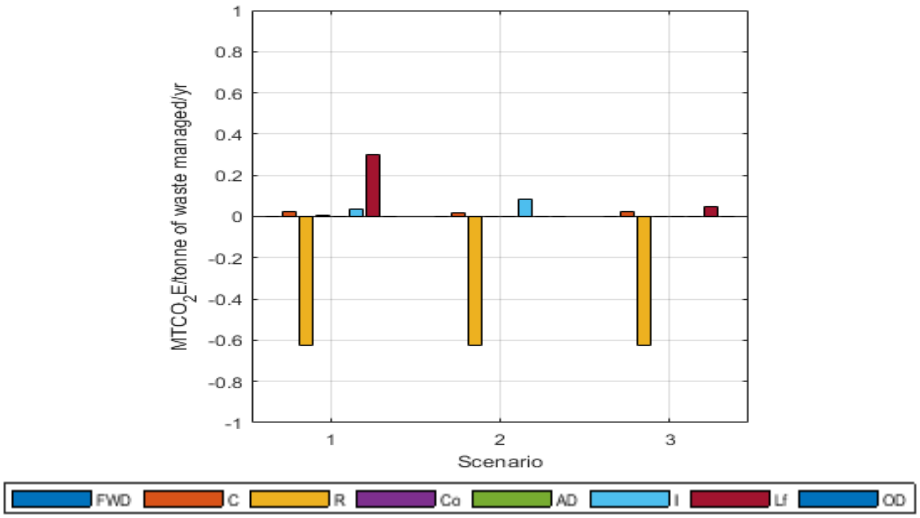
[b] Baseline scenario describing the conditions under a developed economy were extracted from the World Bank (2012)
The variation in emissions is calculated with respect to the baseline condition (S1) whereby Emissions variation is expressed as: %=[(Old−New)/Old] ×100. Where Old=Net emissions from Baseline scenario (S1); and New=Net emissions from optimized scenario (S2).
The cost variation is calculated with respect to existing costs of baseline condition whereby Cost variation is expressed as: % = [(Old _ New)/Old] x 100 where Old = Total cost of baseline scenario (S1) and New = Total cost of optimized case scenario (S2).
Note that the cost variation including carbon credit is based on 3 US$/$MTCO_2E$ (Ecosystem Marketplace, 2017).
Global warming potential-GWP_{100} follows the IPCC (2013) reference, including climate-carbon feedbacks.
The functional unit (FU) was the management of 4000 tonnes per day of waste generated in developed economies.
The reference flow on which the analysis was performed is 1 tonne of waste managed.

The baseline scenario (S1) describes the condition under a developed economy whereby MSW is collected with 20% recovered for recycling and 12% biologically treated (composting). Due to high land costs and low land availability in some regions of developed economies, incineration coupled with an energy recovery system is prevalent (25%) in these areas with environmental and emissions control reaching at times more than three times the landfilling cost per tonne of waste (World Bank, 2012). The residual 43% of MSW is landfilled with LFG collected (60%) for energy recovery. Nevertheless, opening new landfills is often problematic in developed economies because of stricter regulations and increased social concerns and refusal when siting.

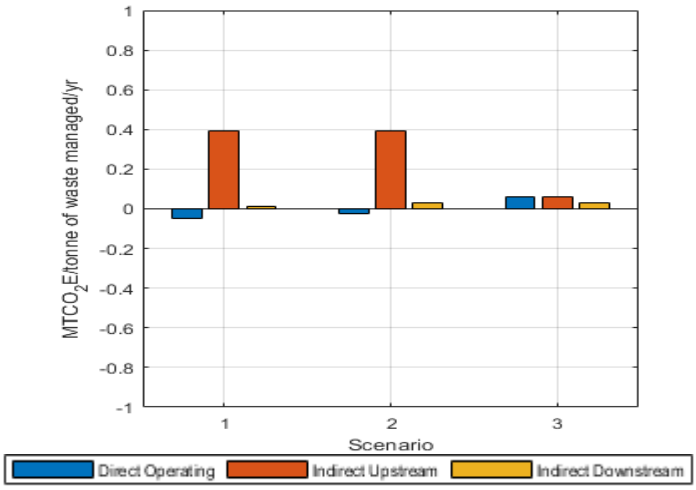
The optimization showed that for the minimum emissions case scenario under a developed economy condition (S2), the maximum possible fractions of MSW recovered

for recycling remained at ~20%. However, the recovered organic fraction of MSW increased, reaching 20% of MSW. The latter is diverted through a FWD (at 75% market penetration and 95% of food waste ground) to the WW stream for aerobic treatment while sludge is treated using anaerobic digestion. The remaining 60% of MSW is incinerated with energy recovery, leading to 93% savings in emissions with respect to baseline condition at a 10% increase in cost when carbon credit is considered (Table A.2).

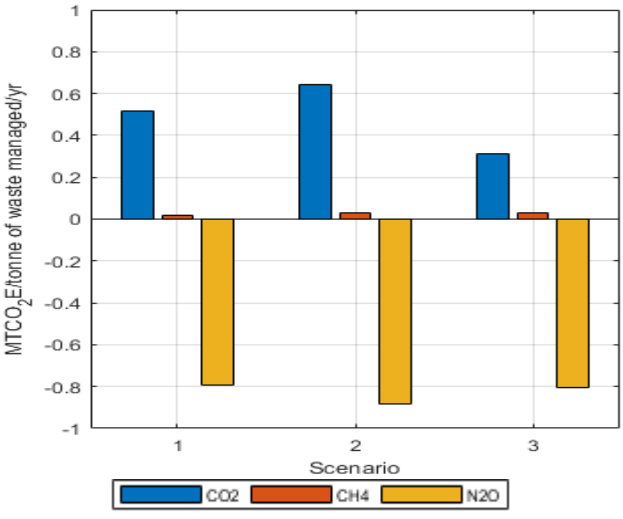
In the minimum cost scenario, the maximum possible mass that can be recycled (~20% of MSW) was processed with the remaining 80% landfilled with LFG collected for energy recovery under a developed economy condition (S3), leading to the least cost (Table A.2). Cost savings with respect to the baseline scenario (S1) reached 3% when including carbon credit (at 3 US$/MTCO$_2$E). The breakeven analysis defines when the carbon credit could enhance the economic viability of adopting a favorable policy towards a technology change. Holding the same value for all parameters, Figure A.17 shows the breakeven points for all case scenarios taking into consideration the reported cost range of carbon credit (0.5 to 50 US$/MTCO$_2$E). Cost variations are calculated to check on savings in costs when including the carbon credits. The results showed that savings in the total cost can reach 10% for the minimum cost scenario S3. On the other hand, the minimum emissions case scenario S2 requires the highest capital investment and cannot achieve overall economic attractiveness except under the highest carbon credit cost of ~50 US$/MTCO$_2$E albeit showing the greatest potential for emissions reduction (Figure A.17).



a. Source



b. Gas



c. Type

Figure A.16. Total net emissions under a developed economy condition disaggregated by source, gas, and type of various scenarios

Scenario: S1 baseline condition; S2 min emissions; S3 min costs

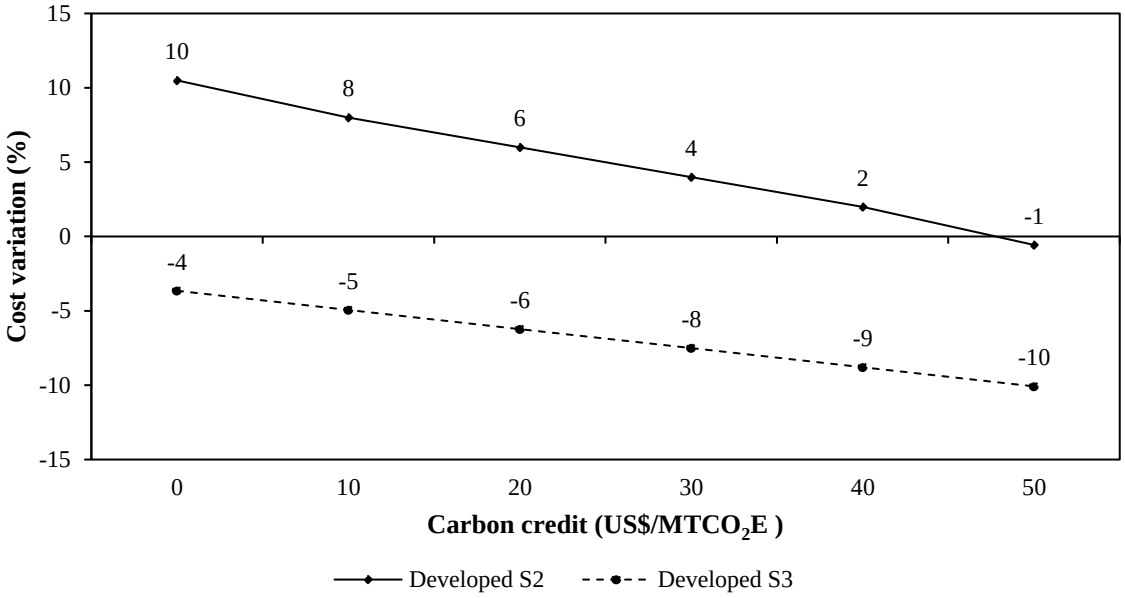


Figure A.17. Breakeven point analysis for carbon credit for all case scenarios under a developed economy condition

Scenario: S2 min emissions; S3 min costs
Cost variation, %= [(Old- New)/Old x100]
where Old = Net total cost of baseline conditions, US$/tonne of waste managed/yr; and
New = Net total cost including carbon credits of optimized scenario, US$/tonne of waste managed/yr

A.3.2. Case study under a developing economy condition

Baseline conditions in developing economy locations are characterized with inefficient waste recycling (3%) or biological treatment such as composting (2%) despite that the waste stream includes a high fraction of biodegradable organic material. Developing economies still depend heavily on open dumping (17%) or poorly operated landfills (78%) with LFG occasionally flared (World Bank, 2012).

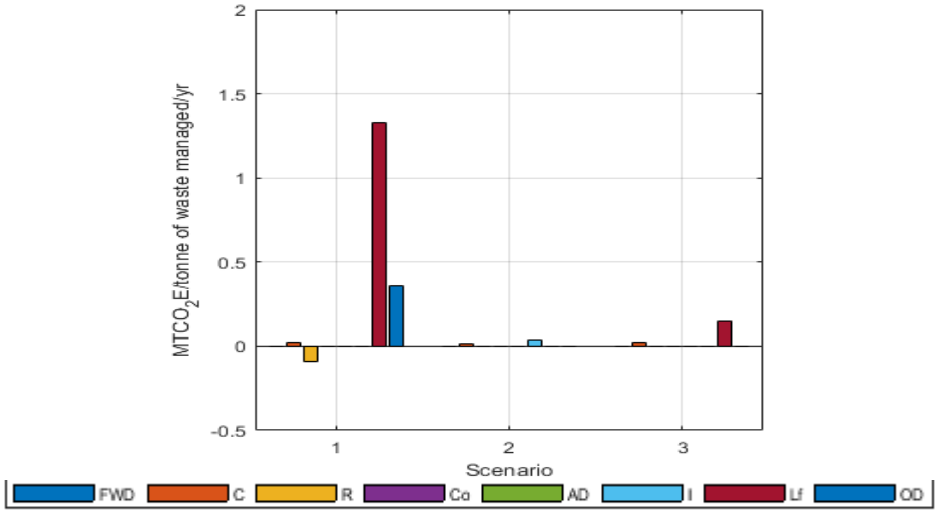
Optimizing existing conditions in developing economy with the objective of minimizing emissions (S1) showed that 40% of MSW is diverted to the WW stream through the use of a FWD with the remaining 60% diverted for incineration with energy recovery (Table A.2). Accordingly, scenario S1 resulted with about 96% savings in emissions with respect to baseline conditions under a developing economy condition due to savings achieved from incineration with energy recovery (Figure A. 18a) that

contributes to indirect-downstream processes (Figure A. 18.c). When all waste is landfilled with energy recovery, minimum costs were achieved for the case of the developing economy, leading to 2% savings in costs when including carbon credit (*Table A.*3). However, the breakeven point analysis under developing economy (Figure A. 19) showed that the minimum emissions case scenario started to become profitable at 30 US$/MTCO$_2$E and can reach 18% savings in cost with respect to baseline conditions (at 50 US$/MTCO$_2$E). On the other hand, the minimum cost case scenario remained profitable under the entire range of carbon credit costs (0 to 50 US$/MTCO$_2$E) and reached 10% savings in cost (at 50 US$/MTCO$_2$E) (*Table A.*3).).

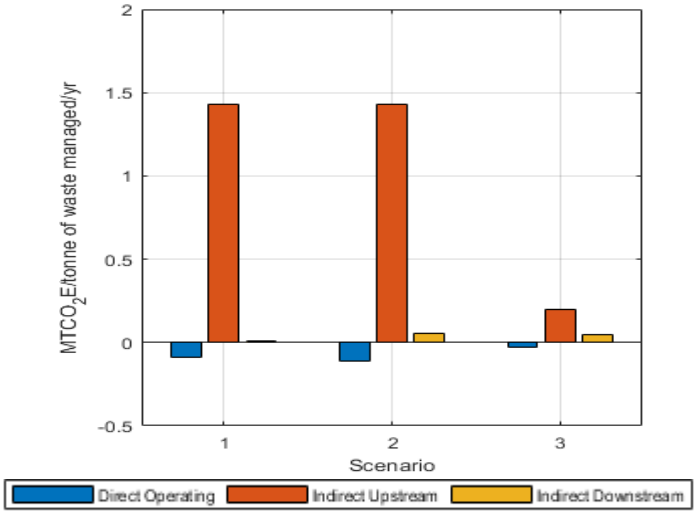
Table A.3. Total net emissions and cost (with and without carbon credit) in the context of a developing economy condition

Case description	Scenario	Percent waste distribution by management process	Total net emissions (MTCO$_2$E/tonne of waste managed/yr) [a]	Emissions variation (%)	Cost variation without carbon credit (%)	Cost variation including carbon credit (%)
Baseline condition [a]	S1	3% Recycling, 2% composting, 78% landfilling, and 17% open dumping	1.44			
Minimize emissions	S2	40% FWD, and 60% incineration with energy recovery	0.06	-96	18	15
Minimize cost	S3	Landfilling all waste	0.85	-41	-0.5	-2

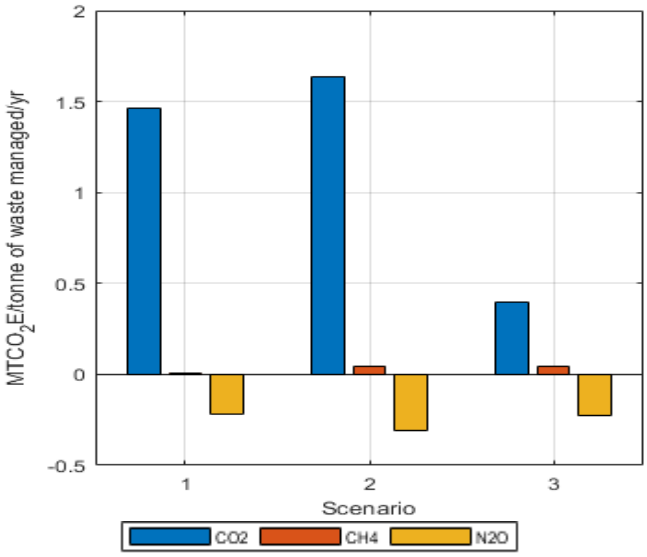
[a] Baseline scenario describing the conditions in developing economy were extracted from World Bank (2012)
The variation in emissions is calculated with respect to the baseline condition (S1) whereby Emissions variation is expressed as: %=[(Old−New)/Old] × 100. Where Old=Net emissions from Baseline scenario (S1); and New=Net emissions from optimized scenario (S2).
The cost variation is calculated with respect to existing costs of baseline condition whereby Cost variation is expressed as: % = [(Old _ New)/Old] x 100 where Old = Total cost of baseline scenario (S1) and New = Total cost of optimized case scenario (S2).
Note that the cost variation including carbon credit is based on 3 US$/MTCO$_2$E (Ecosystem Marketplace, 2017).
Global warming potential-GWP$_{100}$ follows the IPCC (2013) reference, including climate-carbon feedbacks.
The functional unit (FU) was the management of 4000 tonnes per day of waste generated in developing economies.
The reference flow on which the analysis was performed is 1 tonne of waste managed.



a. Source



b. Gas



c. Type

Figure A. 18. Total net emissions under a developing economy condition

disaggregated by source, gas, and type of various scenarios
Scenario: S1 baseline condition; S2 min emissions; S3 min costs

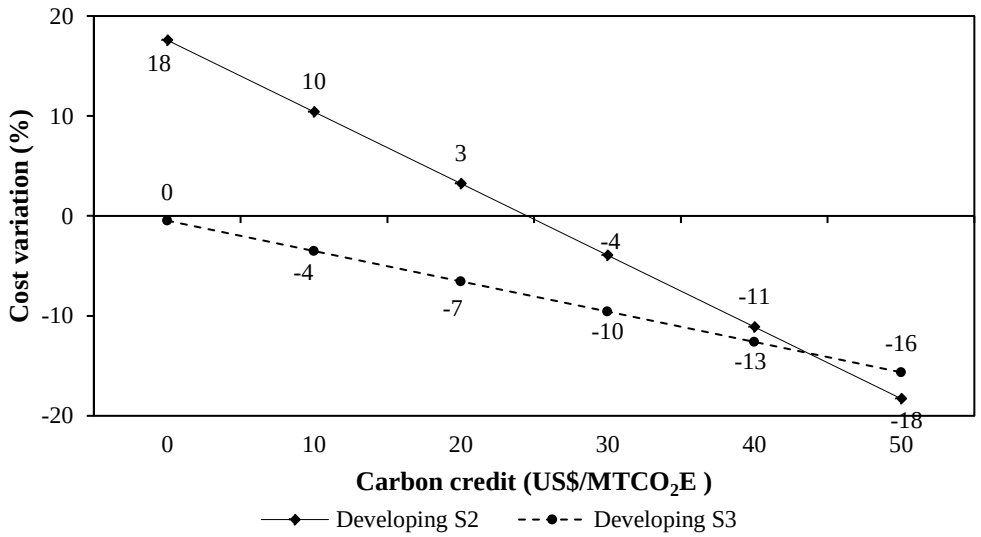


Figure A. 19. Breakeven point analysis for carbon credit under a developing economy condition

Scenario: S2 min emissions; S3min costs
Cost variation, %= [(Old- New)/Old x100]
where Old = Net total cost of baseline conditions, US$/tonne of waste managed/yr; and
New = Net total cost including carbon credits of optimized scenario, US$/tonne of waste managed/yr

A.3.3. Comparison and sensitivity analysis

The results highlight savings in costs achieved under both developed and developing economies when considering the optimization of the integrated waste management systems based on minimizing emissions. Savings in cost were particularly significant for the case of developing economy and ranged between -18% and +18%, depending on the carbon credit cost, with respect to baseline conditions when optimizing based on minimum emissions. Certainly, the overall results are influenced by several factors such as waste composition, considered environmental externalities, type of fuel and energy consumed, and efficiencies of waste technologies that are different between both economies. For instance, the variation in waste composition showed a considerable impact on total emissions and resulting waste management processes in the context of developed and developing economies. The FWD policy was more attractive under a developing economy condition (~40 % of MSW) given the higher fraction of

biodegradable fraction in its waste composition in comparison to developed economies. On the other hand, recycling was more profitable (~ 20% of MSW) in the case of a developed economy given its higher fraction of recyclable material in comparison to developing economies.

As indicated above, the sensitivity analysis was conducted to determine the impact of key parameters on the net total emissions through a Monte Carlo analysis. Accordingly, two parameters were selected, depending on the resulting waste management processes. For instance, the net energy recovery efficiency (~18%) of waste incineration was considered for the minimum emissions scenarios S2 in developed and developing economies while the fraction of the LFG collected (~60%) was considered for minimum costs scenarios S3 equally in developed and developing economies. Figure A. 20 illustrates the frequency distributions of all scenarios' impact upon varying input parameters. The impact on the total net emissions of scenarios S2 in both economies exhibited standard deviations of 0.22 and 0.16 ($MTCO_2$/tonne of waste managed/yr) around their means of -0.37 and 0.22 ($MTCO_2$/tonne of waste managed/yr), respectively, while scenarios S3 in both economies have standard deviations of 1.36 and 1.57 ($MTCO_2$/tonne of waste managed/yr) around their means of 1.43 and 2.5 ($MTCO_2$/tonne of waste managed/yr), respectively (Figure A. 20). This analysis indicates that changing the net energy recovery efficiency of waste incineration for the S2 scenarios in developed and developing economy resulted with savings in emissions 99% and 26% of cases, respectively (Figure A. 20). Moreover, changing the fraction of LFG collected for the S3 scenarios also in developed and developing economy resulted with savings in emissions 13% and 5% of cases, respectively (Figure A. 20).

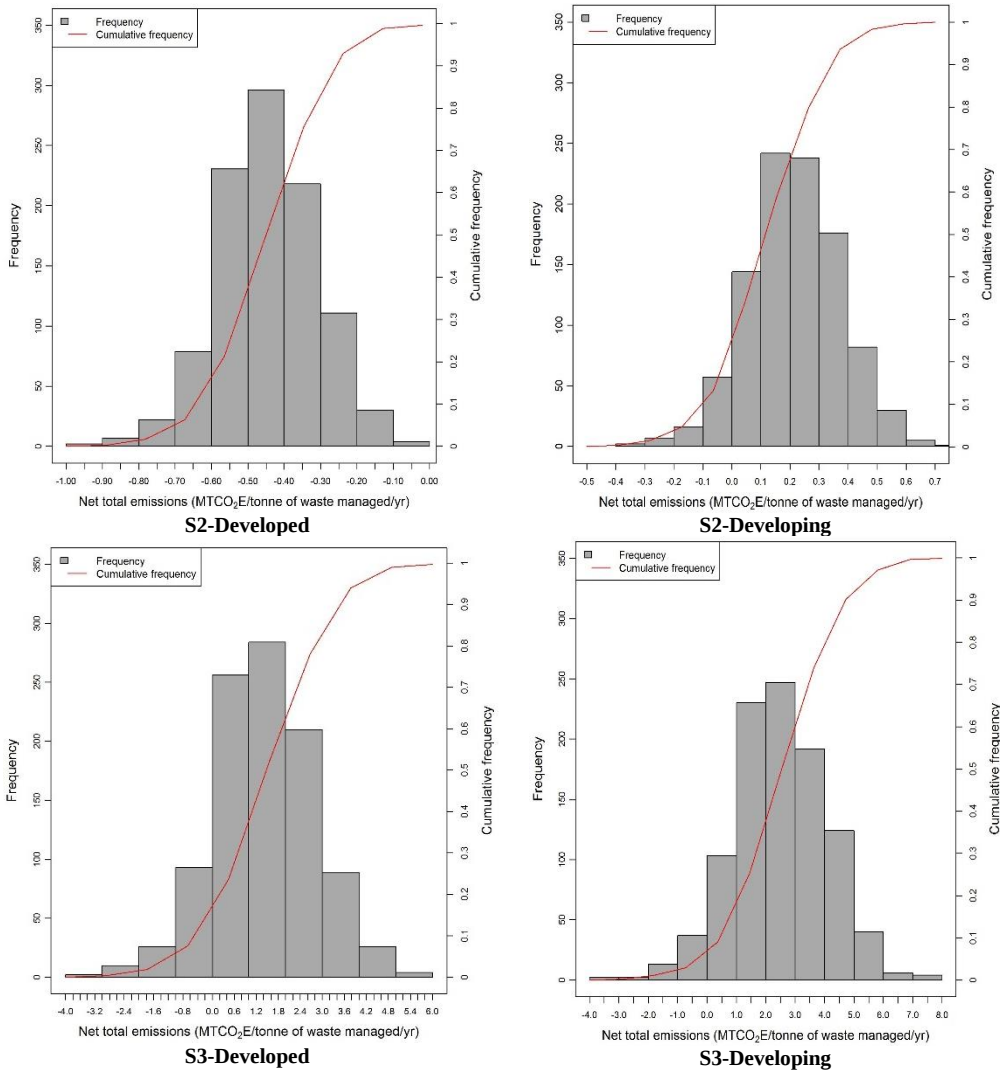


Figure A. 20. Sensitivity analysis using Monte Carlo simulations (1000 runs) for the impact on net total emissions

Scenario: S2 min emissions; S3 min costs

Negative values indicate savings in net total emissions

The tested scenarios demonstrate how the model / software responds and show that carbon credit could enhance the economic viability of adopting a favorable policy towards a technology change. Users can consider more complex scenarios including several policy targets by modifying the model constraints (such as the FWD market penetration rate, recovery rate of MSW material for recycling or biological processes, adopting waste to energy policy or incineration, and minimizing the disposal of MSW in the form of landfilling or open dumping).

Noteworthy, the LCA approach identified integrated waste management systems with minimal impacts by comparing various waste management alternatives and technologies. Accordingly, using a waste-LCA based model/software allows decision-makers to define an integrated waste management policy by selecting the option with the least impact among predetermined policies. However, the optimum policy might be a combination that was not predetermined and thus was not assessed by an LCA-based model. Therefore, coupling LCA modeling to an optimization framework would allow decision-makers to evaluate various economic trade-offs and environmental impacts related to waste management policies. The SWW model/software is innovative because it offers an optimization tool that considers a wide range of possible combinations whether predetermined or not. This would allow the definition of an optimum integrated solid waste and wastewater management system while quantifying emissions based on an LCA approach.

A.4. Conclusion

A MATLAB-based graphical user interface model / software was developed to assist in evaluating the impact on carbon emissions from integrated waste and wastewater management systems under a new user-friendly platform with tools that allow the evaluation of complex processes while visualizing results quickly and efficiently. Special emphasis was exercised to increase the software flexibility by allowing the user to select processes or modify input parameters. The output can be disaggregated by source (waste processes), or type (direct-operating, indirect-upstream, and indirect-downstream), or gas (CH_4, CO_2, N_2O). The novelty of the SWW model /software stems from integrating both solid waste and wastewater management systems under a single framework offering a fixed emissions-accounting tool structure across all processes that are clearly defined for

process-specific input data and calculation outputs. It is equally innovative in embedding an optimization tool that considers a wide range of combinations that allow the definition of an optimum integrated waste and wastewater management system while quantifying costs and emissions based on an LCA approach.

The model / software examines solid waste management processes within a wide context from collection, recycling, composting, anaerobic digestion, incineration, to landfilling, and open dumping or open burning (commonly still practiced in developing economies). On the other hand, the WW management system may consist of aerobic or anaerobic processes with several sludge management options including anaerobic digestion, composting, landfilling, incineration, or land application. In addition, the model / software offers policy and economic tools for the analysis of direct conventional costs-savings and indirect environmental externalities for tested management systems including carbon credits for future policies. This can be particularly important in allocating expenditures for emissions mitigation measures and country reporting on NDCs following the Paris Agreement. The model / software keeps track of mass and material flows specific for each process and offers default values for lacking data. It allows the user to strengthen the results through a sensitivity analysis tool that varies one-parameter-at a time or uses a Monte Carlo simulation analysis.

We tested the new model/software interface in the context of developed and developing economies and demonstrated its applicability as a credible decision-making tool to define economically viable management alternatives with minimal environmental externalities and optimal carbon

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APPENDIX B

Aggregated and Disaggregated Emission Factors

Table B.1. Aggregated and disaggregated emission factors per management process and tonne of waste type

Method [a]	Type of EFs	Recycling	Composting	Anaerobic Digestion	Incineration	Landfilling
			By Waste Type			
Food						
IPCC-2006	Aggregated					0.436
	Disaggregated					$EF_{Lf\,F\,CH_4}$=1.09
EpE						
IWM	Aggregated		0.066		-0.04	0.496
	Disaggregated		$EF_{fuel\,CO_2}$=0.0027 $EF_{elec\,CO_2}$=1.1x10^{-3} $EF_{elec\,CH_4}$=0.02x10^{-3} $EF_{elec\,N_2O}$ =0.6x10^{-3}		$EF_{D\,I\,N_2O}$=0.33 $EF_{ID\,I}$ = -0.37	$EF_{Lf\,F\,CH_4}$ =1.21 $EF_{Lf\,F\,N_2O}$=0.013 $EF_{fuel\,CO_2}$=0.0027 $EF_{elec\,CO_2}$=1.14x10^{-3} $EF_{elec\,CH_4}$ =0.02x10^{-3} $EF_{elec\,N_2O}$= 0.6x10^{-3}
IWM-2	Aggregated		0.012	3.x10^{-7}	0.573	0.832
	Disaggregated		$EF_{fuel\,CO_2}$=0.003 $EF_{fuel\,CH_4}$=7.7x10^{-5} $EF_{fuel\,N_2O}$ =2.2x10^{-6} $EF_{elec\,CO_2}$=6.44x10^{-5} $EF_{elec\,CH_4}$=1.6x10^{-6} $EF_{elec\,N_2O}$ =4x10^{-7}	$EF_{D\,AD\,CO_2}$ =0.44 $EF_{fuel\,CO_2}$=0.003 $EF_{fuel\,CH_4}$=7.7x10^{-5} $EF_{fuel\,N_2O}$ =2.2x10^{-6} $EF_{elec\,CO_2}$=6.44x10^{-5} $EF_{elec\,CH_4}$=1.6x10^{-6} $EF_{elec\,N_2O}$ =4x10^{-7} $EF_{ID\,AD}$=-0.1	$EF_{D\,I\,FCO2}$=0.79 $EF_{ID\,I}$=-0.05	$EF_{Lf\,F\,CH_4}$=2.063 $EF_{fuel\,CO_2}$=0.003 $EF_{fuel\,CH_4}$=7.7x10^{-5} $EF_{fuel\,N_2O}$ =2.2x10^{-6} $EF_{elec\,CO_2}$=6.44x10^{-5} $EF_{elec\,CH_4}$=1.6x10^{-6} $EF_{elec\,N_2O}$ =4x10^{-7}
WARM	Aggregated		-0.184		-0.12	0.578
	Disaggregated		$EF_{fuel\,CO_2}$ = 0.003 $EF_{Co\,F\,CH_4}$= 0.00462 $EF_{Co\,F\,N_2O}$=0.041 $EF_{Co\,F\,CS}$=-0.24		$EF_{I\,F\,N_2O}$=0.04 $EF_{ID\,I\,F}$=-0.16	$EF_{fuel\ \ F\,CO_2}$=0.003 $EF_{Lf\,F\,CH_4}$=1.63 $EF_{Lf\,F\,CS}$=-0.08
Paper						
IPCC-2006	Aggregated				0.034	1.590
	Disaggregated				$EF_{D\,I\,P\,CO_2}$=0.015 $EF_{D\,I\,P\,N_2O}$=0.017 $EF_{D\,I\,P\,CH_4}$=0.001	$EF_{Lf\ \ P\,CH_4}$ =3.975
EpE						
IWM	Aggregated	-0.83			-1.1	0.684
	Disaggregated				$EF_{D\,I\,N_2O}$ =0.33 $EF_{D\,I\,P}$=-1.43	$EF_{Lf\,P\,CH_4}$ =1.68 $EF_{Lf\ \ N_2O}$ =0.013 $EF_{fuel\,CO_2}$=0.0027 $EF_{elec\,CO_2}$=1.14x10^{-3} $EF_{elec\,CH_4}$=0.02x10^{-3} $EF_{elec\,N_2O}$ =0.6x10^{-3}
IWM-2	Aggregated		0.012	3.x10^{-7}	1.24	0.832
	Disaggregated		$EF_{fuel\,CO_2}$=0.003 $EF_{fuel\,CH_4}$=7.7x10^{-5} $EF_{fuel\,N_2O}$ =2.2x10^{-6} $EF_{elec\,CO_2}$=6.44x10^{-5} $EF_{elec\,CH_4}$=1.6x10^{-6} $EF_{elec\,N_2O}$ =4x10^{-7}	$EF_{D\,AD\,CO_2}$ =0.44 $EF_{fuel\,CO_2}$=0.003 $EF_{fuel\,CH_4}$=7.7x10^{-5} $EF_{fuel\,N_2O}$ =2.2x10^{-6} $EF_{elec\,CO_2}$=6.44x10^{-5} $EF_{elec\,CH_4}$=1.6x10^{-6} $EF_{elec\,N_2O}$ =4x10^{-7} $EF_{ID\,AD}$=-0.1	$EF_{D\,I\,P\,CO_2}$=1.28 $EF_{ID\,I}$=-0.05	$EF_{Lf\,P\,CH_4}$= 2.063 $EF_{fuel\,CO_2}$=0.003 $EF_{fuel\,CH_4}$=7.7x10^{-5} $EF_{fuel\,N_2O}$=2.2x10^{-6} $EF_{elec\,CO_2}$=6.44x10^{-5} EF_{elecCH_4}=1.6x10^{-6} $EF_{elec\,N_2O}$ =4x10^{-7}

Method [a]	Type of EFs	Recycling	Composting	Anaerobic Digestion	Incineration	Landfilling
			By Waste Type			
WARM	Aggregated Disaggregated	-3.520			-0.42 $EF_{I\ P\ CO_2}$ =0.03 $EF_{ID\ I\ P}$= -0.45	0.036 $EF_{fuel\ CO_2}$ =0.003 $EF_{Lf\ P\ CH_4}$=2.1 $EF_{Lf\ P\ CS}$=-0.81
Plastics						
IPCC-2006	Aggregated Disaggregated				2.219 $EF_{D\ I\ PL\ CO_2}$ =2.2 $EF_{D\ I\ PL\ N_2O}$=0.017 $EF_{D\ I\ PL\ CH_4}$ =0.001	0
EpE						
IWM	Aggregated Disaggregated	-4.530			-1.71 $EF_{D\ I\ N_2O}$=0.33 $EF_{D\ I\ CO_2}$ =0.98 $EF_{ID\ I\ PI}$= -3.02	0.020 EF_{fuelCO_2}=0.0027 EF_{elecCO_2}=1.14x10^{-3} EF_{elecCH_4} =0.02x10^{-3} EF_{elecN_2O}= 0.6x10^{-3}
IWM-2	Aggregated Disaggregated	-1.203			2.652 $EF_{D\ I\ PL\ CO_2}$ =2.74 $EF_{ID\ I}$ = -0.05	
WARM	Aggregated Disaggregated	-0.980			1.56 $EF_{I\ PL\ CO_2}$ =2.4 $EF_{ID\ I\ PL}$=- 0.84	0.006 $EF_{fuel\ CO_2}$ =0.003 $EF_{Lf\ PL\ CH_4}$=0 $EF_{Lf\ PL\ CS}$=0
Textiles						
IPCC-2006	Aggregated Disaggregated				0.253 $EF_{D\ I\ T\ CO_2}$ =0.235 $EF_{D\ I\ T\ N_2O}$=0.017 $EF_{D\ I\ T\ CH_4}$ =0.001	0.954 $EF_{Lf\ T\ CH_4}$ =2.385
EpE						
IWM						
IWM-2	Aggregated Disaggregated	-5.869			1.24 $EF_{D\ I\ TCO_2}$ =1.28 $EF_{ID\ I}$ = -0.05	0.832 $EF_{Lf\ T\ CH_4}$= 2.063 $EF_{fuel\ CO_2}$=0.003 $EF_{fuel\ CH_4}$=7.7x10^{-5} $EF_{fuel\ N_2O}$=2.2x10^{-6} $EF_{elec\ CO_2}$=6.44x10^{-5} $EF_{elec\ CH_4}$=1.6x10^{-6} $EF_{elec\ N_2O}$=4x10^{-7}
WARM	Aggregated Disaggregated	-2.370			1.23 $EF_{D\ I\ T\ CO_2}$ =1.67 $EF_{ID\ I\ T}$= -0.44	0.006 $EF_{fuel\ CO_2}$ =0.003 $EF_{Lf\ T\ CH_4}$=0 $EF_{Lf\ T\ CS}$=0
Garden						
IPCC-2006						0.663 $EF_{Lf\ GA\ CH_4}$ =1.657
EpE						
IWM			0.066			
IWM-2						
WARM	Aggregated Disaggregated		-0.155 $EF_{fuel\ CO_2}$ =0.003 $EF_{Co\ GA\ N_2O}$ =0.06 $EF_{Co\ GA\ CS}$=-0.24		-0.19 $EF_{ID\ I\ GA}$= -0.19	0.988 $EF_{fuel\ CO_2}$ =0.003 $EF_{Lf\ GA\ CH_4}$=0.88 $EF_{Lf\ GA\ CS}$=0.63
Wood						
IPCC-2006						2.016 $EF_{Lf\ W\ CH_4}$ =5.04
EpE						
IWM						
IWM-2						
WARM	Aggregated Disaggregated	-2.460			-0.4 $EF_{D\ I\ W\ N_2O}$=0.04 $EF_{ID\ I\ W}$= -0.44	-0.614 $EF_{fuel\ CO_2}$ =0.003 $EF_{Lf\ W\ CH_4}$=1.3 $EF_{Lf\ W\ CS}$=-1.14
Glass						
IPCC-2006						
EpE						
IWM	Aggregated Disaggregated	-0.92			0.376	0.020

Method [a]	Type of EFs	Recycling	Composting	Anaerobic Digestion	Incineration	Landfilling
			By Waste Type			
					$EF_{D\,I\,N_2O}$=0.98 $EF_{D\,I\,CO_2}$ =0.34 $EF_{ID\,I\,G}$= 0.05	EF_{fuelCO_2}=0.0027 EF_{elecCO_2}=1.14x10^{-3} EF_{elecCH_4} =0.02x10^{-3} EF_{elecN_2O}= 0.6x10^{-3}
IWM-2	Aggregated Disaggregated	-0.087			0.094 $EF_{D\,I\,G\,CO_2}$ =0.059 $EF_{ID\,I}$ = -0.05	
WARM	Aggregated Disaggregated	-0.280			0.025 $EF_{ID\,I\,G}$=0.025	0.006 $EF_{fuel\,CO_2}$ =0.003 $EF_{Lf\,G\,CH_4}$=0 $EF_{Lf\,G\,CS}$=0
Metals						
IPCC-2006						
EpE						
IWM	Aggregated Disaggregated	-1.994			0.5 $EF_{D\,I\,N_2O}$=0.33 $EF_{ID\,I\,M}$= 0.17	0.020 $EF_{fuel\,CO_2}$=0.0027 $EF_{elec\,CO_2}$=1.14x10^{-3} $EF_{elec\,CH_4}$ =0.02x10^{-3} $EF_{elec\,N_2O}$= 0.6x10^{-3}
IWM-2		-4.553				
WARM	Aggregated Disaggregated	-3.970			-0.02 $EF_{ID\,I\,M}$=-0.02	0.006 $EF_{fuel\,CO_2}$ =0.003 $EF_{Lf\,M\,CH_4}$=0 $EF_{Lf\,M\,CS}$=0
Nappies						
IPCC-2006	Aggregated Disaggregated					1.013 $EF_{Lf\ N\,CH_4}$ =2.532
EpE						
IWM						
IWM-2						
WARM						
Others or commingled MSW						
IPCC-2006	Aggregated Disaggregated		0.177[b] $EF_{Co\,CH_4}$= 0.084 $EF_{Co\,N_2O}$= 0.093	0.021 $EF_{AD\,CH_4}$= 0.021	0.022 $EF_{D\,I\,O\,CO_2}$ =0.003 $EF_{D\,I\,O\,N_2O}$=0.017 $EF_{D\,I\,O\,CH_4}$ =0.001	
EpE[c]	Aggregated Disaggregated		0.175[d] $EF_{Co\,CH_4}$= 0.107 $EF_{Co\,N_2O}$= 0.065 $EF_{fuel\,CO_2}$=0.0026 $EF_{elec\,CO_2}$=5x10^{-4}	0.045[e] $EF_{AD\,CH_4}$ =0.009 $E_{fuel\,CO_2}$=0.0026 $EF_{elec\,CO_2}$=5x10^{-4}	0.382[f] $EF_{I\,CO_2}$= 0.332 $EF_{I\,N_2O}$= 0.01 $EF_{fuel\,CO_2}$=0.0026 $EF_{elec\,CO_2}$=5x10^{-4}	0.009[g] $EF_{fuel\,CO_2}$=0.0026 $EF_{elec\,CO_2}$=5x10^{-4}
IWM	Aggregated Disaggregated				-0.58 $EF_{D\,I\,N_2O}$=0.33 $EF_{ID\,I\,O}$= -0.91	0.020 $EF_{fuel\,CO_2}$=0.0027 $EF_{elec\,CO_2}$=1.14x10^{-3} $EF_{elec\,CH_4}$ =0.02x10^{-3} $EF_{elec\,N_2O}$= 0.6x10^{-4}
IWM-2	Aggregated Disaggregated			0.345[h]	1.24 $EF_{D\,I\,O\,CO_2}$ =1.28 $EF_{ID\,I}$ = -0.05	
WARM	Aggregated Disaggregated				-0.01 $EF_{D\,I\,O}$=0.38 $EF_{ID\,I\,O}$= -0.35	1.242 $EF_{fuel\,CO_2}$ =0.003 $EF_{Lf\,O\,CH_4}$=3.64 $EF_{Lf\,O\,CS}$=-0.22

[a] Methods:

- **Entreprises pour l'Environnemnent (EpE):** It accounts for gross and net direct emissions, as well as indirect emissions (e.g. electricity consumption) and avoided emissions from the recovery of energy and material. In order to calculate direct emissions from waste degradation in landfills, the user selects a common method and refers to the regulatory methodologies recommended by the authorities of the country where the site(s) is (are) located.
- **Integrated Waste Management Model for municipalities IWM:** accepted by Environment Canada to evaluate the environmental performance of various elements of an ISWM system [1,2].
- **The Integrated Waste Management Model-2 (IWM-2)** developed by McDougall et al. (2001) [3] is based on ISWM and is to the International Standards ISO 14040 series on LCA.
- **The U.S. Environmental Protection Agency (EPA) Waste Reduction Model (WARM):** is used to estimate emissions reductions in climate change impact assessment [4]. In its last release, WARM (v. 14) included 54 materials, products and mixed categories [5].

[b] Considers total mass of MSW treated.

(c) EpE considers for recycling only: Direct emissions from permanent combustion facilities and on-site mobile equipment; direct emissions from refrigerant / fluorinated gases released because of WEEE leakages / dismantling process and indirect emissions from electricity or purchased heat consumption [6].
(d) Considers CH_4 emissions from Organic fraction of MSW and N_2O emissions from MSW.
(e) Considers VFG (vegetable, fruit and garden wastes) continuous process, as well as emissions from fuel and electricity consumption, however, it does not provide a methodology to account for avoided emissions from energy recovery.
(f) Considers direct CO_2 and N_2O emissions from waste combustion as well as emissions from fuel and electricity consumption, however, it does not provide a methodology to account for avoided emissions from energy recovery.
(g) To calculate emissions from landfills the entity should refer to the regulatory methodologies recommended by the competent authorities of the country where the site(s) is (are) located. It also considers direct emissions from (Direct emissions from permanent combustion facilities and on-site mobile equipment) and indirect emissions from (Indirect emissions from electricity or purchased heat consumption) and avoided emissions from electricity and heat recovery.
(h) Organic MSW using bio gasification with electricity generation also considers biological treatment (composting or bio gasification) of paper waste.

Disaggregated EFs are by definition generic factors determined from a number of processes representing characteristics calculated per unit of activity; thus they are expressed in $MTCO_2E$ per characteristic unit (tonne of MSW treated; KWh of electricity; Liter of Diesel fuel) using a GWP_{100}, (IPCC, 1995) [7]. EFs are fixed default values within every method with the exception of EpE where the user can select EFs of recycling (adapted from- [4] and landfilling (selected FOD method adapted from IPCC-2006 Guidelines [8].
Aggregated EFs is the combined impact of disaggregated EFs expressed in $MTCO_2E$ per tonne of waste category whereby LFG (landfill gas collected) = 0.6; Electricity consumed= 32 kWh/tonne of waste composted, 70-80 kWh/tonne of waste incinerated, 68-50 kWh/tonne of waste anaerobically digested, and 8kWh/ tonne of waste landfilled; Fuel consumed = ~2 Liters/ tonne of waste landfilled, ~3.28 Liters/ tonne of waste composted, and 0.89 Liters/ tonne of waste anaerobically digested.
Co= Composting; **I**=Incineration; **Lf**=Landfilling; **D**= Direct, **ID**= Indirect; **CS**= Carbon storage; **EFfuel**=Emission factor for fuel combustion; **EFelec**=Emission factor of electricity consumed or recovered; **CS**= Carbon storage; **CO_2=**Carbon dioxide; **CH_4**=Methane; **NO_2**=Nitrous oxide; **F** = Food waste; **G** = Glass; **M** = Metals; **O** = others; **P** = Paper; **PL** = Plastics; **T** = Textiles; **W** = Wood; **GA**=Garden waste; **N**=Nappies.

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[4] EPA/ICF, Documentation for Greenhouse Gas Emission and Energy Factors Used in the Waste Reduction Model (WARM): Background Chapters (WARM V. 12), U.S. Environmental Protection Agency Office of Resource Conservation and Recovery, Washington DC, 2012.

[5] EPA/ICF, Documentation for Greenhouse Gas Emission and Energy Factors Used in the Waste Reduction Model (WARM): Background Chapters, U.S. Environmental Protection Agency Office of Resource Conservation and Recovery, Washington DC, 2016

[6] EpE, Protocol for the quantification of GHG emissions from waste management activities, Cedex, Nanterre, 2013.

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APENDIX C

Background references and values for model input data

Waste always refers to wet waste

Table C.1. Fuel consumption during waste collection and transport
(Nguyen and Wilson, 2010; Larsen et al., 2009)

Area	*Liters of diesel fuel oil per Ton of waste collected*
High density urban areas	1.6-3.6
Medium density urban areas	1.4-5.7
Low density urban areas	13.2-16
Rural areas	6.3-10.1
Default average	*6*

Table C.2. Emission Factors from Recycling
(EPA, 2006; EPA/ICF, 2016)

Waste category, c	**MTCO$_2$E / Ton of waste**
Paper, P	$EF_{R\,P} = -3.52$
Plastics, PL	$EF_{R\,PL} = -0.98$
Glass, G	$EF_{R\,G} = -0.28$
Textiles, T	$EF_{R\,T} = -2.37$
Wood, W	$EF_{R\,W} = -2.46$
Metals, M	$EF_{R\,M} = -3.97$

Table C.3. Composting technologies

Type of emissions	**Default value**	**Unit**	**Source**
Direct: operational			
Waste degradation	$EF_{Co\,F\,CH_4} = 7.48 \times 10^{-3}$ $EF_{Co\,GA\,CH_4} = 18.9 \times 10^{-3}$ $EF_{Co\,F\,N_2O} = 39.6 \times 10^{-3}$ $EF_{Co\,GA\,N_2O} = 60.9 \times 10^{-3}$	MTCO$_2$E /Ton of waste category c	EPA/ICF (2016)
Fuel consumption	Open composting: 3.2 Enclosed composting: 1.6	Liters of diesel fuel oil / Ton of waste composted	Boldrin *et al.* (2009)
Indirect: upstream			
Electricity provision	Open composting: 3.2 Enclosed composting: 37 Adopted average: 32	kWh / Ton of waste composted	Boldrin *et al.* (2009) Boldrin *et al.* (2009) McDougall *et al.* (2001)
Indirect: downstream			
Carbon storage: Land application	$EF_{Co\,Peat\,CO_2} = -0.65$	MTCO$_2$E / Ton of compost	Boldrin *et al.* (2009

Peat substitution	$EF_{Co\ CS\ CO_2}$= -0.101	$MTCO_2E$ / Ton of compost	Boldrin *et al.* (2009)

EF: Emission factor; C_O: Composting; GA: Garden; F: Food; CS: Carbon storage

Table C.4. Anaerobic digestion

Type of emissions	**Default value**	**Unit**	**Source**
Direct: operational			
Waste degradation	$EF_{AD\ F\ CH_4}$ = 8.84x10^{-3} $EF_{AD\ GA\ CH_4}$ = 43.5x10^{-3} $EF_{AD\ F\ N_2O}$ = 32.78x10^{-3} $EF_{AD\ GA\ N_2O}$ = 17.88x10^{-3}	$MTCO_2E$ /Ton of waste category *c (F, GA)*	EPA/ICF (2016)
Fuel consumption	1.6	Liters of diesel fuel oil / Ton of waste anaerobically digested	Møller *et al.* (2009)
Indirect: upstream			
Electricity provision	35	kWh/ Ton of waste anaerobically digested	Møller *et al.* (2009)
Indirect: downstream			
Electricity production	190	kWh/ Ton of waste anaerobically digested	McDougall *et al.* (2001)
Carbon storage Land application	$EF_{AD\ CS\ F\ CO_2}$ = -32.8x10^{-3} $EF_{AD\ CS\ GA\ CO_2}$ = -159x10^{-3}	$MTCO_2E$ / Ton of compost	EPA/ICF (2016)
Peat substitution	$EF_{AD\ Peat\ F\ CO_2}$ = -0.01 $EF_{AD\ Peat\ GA\ CO_2}$ = -0.006		EPA/ICF (2016) Møller *et al.* (2009)

EF: Emission factor; AD: Anaerobic Digestion; GA: Garden; F: Food; CS: Carbon storage

Table C.5. Incineration

Type of emissions	Default value	Unit	Source
Direct: operational			
Waste degradation	$EF_{I\,P\,CO_2} = 0.03$ $EF_{I\,T\,CO_2} = 1.67$ $EF_{I\,O\,CO_2} = 0.34$ $EF_{I\,F\,N_2O} = 0.038$ $EF_{I\,PL\,CO_2} = 2.42$ $EF_{I\,O\,N_2O} = 0.38$ $EF_{OB\,CH_4} = 0.2$ $EF_{OB\,N_2O} = 0.04$	$MTCO_2E$ /Ton of waste category *c*	EPA/ICF (2016) IPCC (2006)
Fuel consumption	1	Liters of diesel fuel oil /Ton of waste incinerated	Astrup *et al.* (2009) Smith *et al.* (2001)
Indirect upstream			
Electricity provision	92.5	kWh/ Ton of waste incinerated	Astrup *et al.* (2009)
Indirect downstream			
Electricity production *energy content of waste incinerated*	$Elecprod_{I\,P}$ = 4,660 $Elecprod_{I\,F}$ = 1,377 $Elecprod_{I\,PL}$= 9,086 $Elecprod_{I\,G}$ = -138 $Elecprod_{I\,T}$ = 4,455 $Elecprod_{I\,W}$ = 4,865 $Elecprod_{I\,M}$ = -205 $Elecprod_{I\,O}$ = 2,930	kWh/ Ton of waste category *c* incinerated	EPA/ICF (2016)
Residues management			
Material recovery	0.02	Tons/ Ton of waste incinerated	EPA/ICF (2016)
	$EF_{I\ residues}$ = -1.99	$MTCO_2E$ / Ton of material recovered	EPA/ICF (2016)
Treatment of residues			
	Bottom Ash: 0.23	Tons/ Ton of waste incinerated	Astrup *et al.* (2009)
	$EF_{I\ residues}$ = 0.011	$MTCO_2E$ / Ton of residues	Manfredi *et al.* (2009)

EF: Emission factor; $Elec_{prof}$: Electricity produced; I: Incineration; OB: Open Burning;
F: Food; G: Glass; M: Metals; O: Others; P: Paper; PL: Plastics; T: Textiles; W: Wood

Table C.6. Landfilling

Type of emissions	Default value	Unit	Source
Direct: operational			
Waste degradation	$EF_{Lf\,P\,CH_4}$ =3.4; $EF_{Lf\ F\,CH_4}$ =2.64 $EF_{Lf\,W\,CH_4}$ =2.1 $EF_{Lf\ O\,CH_4}$ =5.89 $EF_{Lf\,N_2O}$ =0.0128	MTCO$_2$E /Ton of waste category *c*	EPA/ICF (2016)
Fuel consumption	2	Liters of diesel fuel oil/ Ton of waste	Manfredi *et al.* (2009)
Indirect: upstream			
Electricity provision	7	kWh/ Ton of waste landfilled	Manfredi *et al.* (2009)
Material provision *for landfill construction*	$EF_{Lf\,const}$ =1.85 Cells: 0.001 Drainage system: 0.1 $Vfuel_{Lf\,const}$ = 0.75	MTCO$_2$E /Ton of material Tons of liner/ Ton of waste landfilled Tons of material/Ton of waste landfilled Liters/ Ton of waste landfilled	
Indirect: downstream			
Electricity production	$Elecprod_{Lf}$ =4,325	kWh/ Ton of CH_4	EPA/ICF (2016)
Carbon storage	$EF_{Lf\,const}$ = -0.16	MTCO$_2$E/ Ton of waste landfilled	Manfredi *et al.* (2009)

EF: Emission factor; $EF_{Lf\,const}$: Emission factor for landfill construction; $Elec_{prof}$: Electricity produced; $V_{fuel\ Lf\,const}$: Volume of fuel consumed for landfill construction; Lf: Landfilling; F: Food; O: Others; P: Mixed Paper; W: Wood; ww: wet waste

Table C.7. Methane correction factor for wastewater treatment systems *(IPCC, 2006)*

Type of treatment system and discharge pathway		Methane correction factor
Aerobic	Centralized aerobic treatment plant (well managed)	0
	Centralized aerobic treatment plant (not well managed)	0.3
Anaerobic	Anaerobic reactor	0.8
	Anaerobic shallow lagoon	0.2
	Anaerobic deep lagoon	0.8
	Anaerobic digester for sludge	0.8
	Septic system	0.5

Table C.8 Emission Factors for common sludge treatment methods *(IPCC, 2006)*

Sludge management (S)	Default value	Unit
Composting (*Co*)	$EF_{FWD\,S\,F\,CH_4}$ = 0.01	Tons of CH_4/ton of waste treated
	$EF_{FWD\,S\,F\,N_2O}$=6x10^{-4}	Tons of N_2O/ton of waste treated
Anaerobic digestion (*AD*)	$EF_{FWD\,S\,F\,CH_4}$ = 2x10^{-3}	Tons of CH_4/ton of waste treated
Incineration (*I*)	$EF_{FWD\,S\,F\,CH_4}$ = 9.7x10^{-6}	Tons of CH_4/ton of waste
	$EF_{FWD\,S\,F\,N_2O}$=900x10^{-6}	Tons N_2O/ton of waste

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APPENDIX D

Life Cycle Inventory Data

Table D.1. Source separation efficiency by individual waste component of scenario S4

	Organic	Wood	Paper	Glass	Metals	Plastics
Separation efficiency (%)	94	86	42	29	98	27

The SS efficiencies were calculated based on a successful source separation scheme implemented in a developed economy (Di Maria and Micale, 2013). Note that 15% of MSW is being recovered for recycling and 50% is biologically treated using anaerobic digestion (S4).

Table D.2. Input data specific to the test area

Type of data	Value/ description		Source
Waste composition (%)	Food Papers Plastics Textiles Wood Glass Metals Nappies Others	(53.4) (15.6) (13.8) (2.8) (0.8) (3.4) (2) (3.6) (4.6)	Laceco/Ramboll (2012)
Average cost of MSW management (US$/ tonne of waste)	Collection Sorting Composting Landfilling	(33) (23) (25) (46)	MoE/UNDP/ECODIT (2011); CDR (2010)
Produced compost quality	pH (1:5) EC (1:5) mS/cm Lead mg/kg Chromium mg/kg Cadmium mg/kg Moisture content % weight) Organic Matter % Density g/ml Nitrogen % C/N ratio Fecal Coliforms MPN/g Salmonella col/g	7.31 4.5 161 31 0.8 48 (by 52.57 0.47 1.44 16.5 <3 <10	CDR-LACECO (2014)
Average waste calorific value (MJ/Kg)	6.9		Laceco/Ramboll (2012)
Material recovery efficiency (%) at the material recovery facilities (MRFs) prior to treatment	Cardboard/ paper Metal Glass Plastic	(43.22) (16.85) (6.47) (33.46)	Laceco/Ramboll (2012)
Recycling recovery efficiency (%)	About 10% by mechanical separation using bag openers, trommel screens, and magnetic separation with manual sorting for the recovery of recyclable materials that helped in increasing the separation efficiency		CDR-LACECO (2014)
Sold recyclables (%)	About 81% of the recovered recyclables are sold for recycling industries the rest is sent for landfilling		CDR-LACECO (2014)

Table D.3. LCI of collection of 1 tonne of waste
(Data extracted from DTU 2017)

Parameter	Value	Unit	Parameter	Value	Unit	Parameter	Value	Unit
Emission to air								
1,3,5-trimethylbenzene	1.104×10^{-13}	kg	Toluene	3.512×10^{-7}	kg	Uranium-234	6.624×10^{-6}	kBg
Dioxins	2.152×10^{-14}	kg	Hexane	1.861×10^{-5}	kg	Uranium-235	2.553×10^{-5}	kBg
Acetaldehyde	4.434×10^{-7}	kg	hydrocyanic acid	9.122×10^{-11}	kg	Uranium-238	3.751×10^{-5}	kBg
Acetic acid	4.093×10^{-7}	kg	Hydrogen	4.567×10^{-6}	kg	used air	9.619×10^{-1}	kg
Acetonnee	3.965×10^{-7}	kg	Arsine	6.070×10^{-11}	kg	Vanadium	6.616×10^{-6}	kg
acid (as H+)	4.883×10^{-10}	kg	hydrogen bromide	5.156×10^{-10}	kg	Ethene	3.212×10^{-8}	kg
Acrolein	2.739×10^{-9}	kg	Hydrogen chloride	2.336×10^{-5}	kg	Heat, waste	6.781×10	MJ
Ammonia	1.711×10^{-5}	kg	Hydrogen fluoride	1.798×10^{-6}	kg	Xenon-131m	8.542×10^{-5}	kBg
ammonium	2.097×10^{-12}	kg	hydrogen iodide	4.586×10^{-13}	kg	Xenon-133	1.398×10^{-2}	kBg
anthracene	3.882×10^{-10}	kg	Hydrogen sulfide	2.705×10^{-5}	kg	Xenon-135	4.622×10^{-3}	kBg
Antimony	2.263×10^{-9}	kBq	Hydrogen-3	1.206×10^{-2}	kBg	Xenon-137	1.212×10^{-6}	kBg
Antimony-124	9.815×10^{-10}	kBq	Indeno -pyrene	1.297×10^{-10}	kg	Xenon-138	1.561×10^{-4}	kBg
Argon-41	6.189×10^{-3}	kg	Iodine-129	6.068×10^{-6}	kBg	Xylene	1.743×10^{-6}	kg
Arsenic	5.564×10^{-8}	kg	Iodine-131	9.119×10^{-7}	kBg	Zinc	3.077×10^{-7}	kg
arsenic trioxide	7.313×10^{-13}	kg	Iron	5.353×10^{-7}	kg	zinc oxide	2.268×10^{-14}	kg
Barium	8.342×10^{-6}	kg	Kryptonne-85	$1.045 \times 10+2$	kBg	CO_2 fossil	1.581×10	kg
Benzene	6.913×10^{-6}	kg	Lead	2.795×10^{-7}	kg	CH_4 fossil	1.760×10^{-2}	kg
benzo[a]anthracene	1.953×10^{-10}	kg	lead dioxide	1.303×10^{-13}	kg			
Benzo(a)pyrene	1.060×10^{-10}	kg	Manganese	2.497×10^{-8}	kg	Diethylamine	4.870×10^{-17}	kg
benzo[g,h,i]perylene	1.742×10^{-10}	kg	Mercury	1.731×10^{-8}	kg	Water	5.558×10^{-1}	kg
benzo[k]fluoranthene	3.485×10^{-10}	kg	Methanol	1.279×10^{-7}	kg	Nitrogen	2.961×10^{-4}	kg
Beryllium	6.779×10^{-10}	kg	Molybdenum	4.171×10^{-8}	kg	Cadmium	5.207×10^{-8}	kg
Boron	4.159×10^{-7}	kg	naphthalene	4.076×10^{-8}	kg	Sulfate	9.080×10^{-10}	kg
Bromine	1.155×10^{-7}	kg	Butane	3.586×10^{-4}	kg	CO, fossil	2.133×10^{-2}	kg
Butadiene	1.794×10^{-12}	kg	Nickel	1.023×10^{-6}	kg	Chromium VI	2.604×10^{-7}	kg
Cadmium	1.810×10^{-8}	kg	NO	3.959×10^{-11}	kg	Copper	5.207×10^{-7}	kg
Carbon disulfide	2.736×10^{-12}	kg	N_2O	3.652×10^{-5}	kg	Lead	5.207×10^{-7}	kg
CO, non-fossil	2.178×10^{-3}	kg	NMVOC	1.364×10^{-3}	kg	Mercury	5.207×10^{-9}	kg
Carbon-14	2.840×10^{-3}	kBq	octane	6.899×10^{-6}	kg	Nickel	5.207×10^{-6}	kg
Cesium-134	7.773×10^{-7}	kBq	oxygen	1.951×10^{-3}	kg			
Cesium-137	1.588×10^{-6}	kBq	palladium	4.351×10^{-17}	kg	Selenium	5.207×10^{-8}	kg
Methane, Trichlorofluoro-,CFC-11	1.731×10^{-8}	kg	P > 10 um	4.185×10^{-12}	kg	Sulfur dioxide	5.232×10^{-4}	kg
Methane, chlorotrifluoro-, CFC-13	2.337×10^{-9}	kg	P < 2.5 um	6.780×10^{-5}	kg	NMVOC	1.023×10^{-2}	kg
Chlorine	1.030×10^{-5}	kg	Pentane	1.219×10^{-4}	kg	Zinc	5.207×10^{-6}	kg
Chromium	1.639×10^{-7}	kg	phenanthrene	1.280×10^{-8}	kg	P < 2.5 um	3.887×10^{-3}	kg
chromium III	1.571×10^{-10}	kg	Phenol	5.367×10^{-12}	kg	tin oxide	1.134×10^{-14}	kg
Chrysene	4.797×10^{-10}	kg	Phosphine	1.454×10^{-13}	kg	Titanium	2.968×10^{-11}	kg
Cobalt	8.272×10^{-8}	kBq	Plutonium-alpha	1.823×10^{-10}	kBq	Helium	1.223×10^{-8}	kg
Cobalt-58	4.870×10^{-9}	kBq	Polychlorinated biphenyls	1.330×10^{-10}	kg	Heptane	1.254×10^{-5}	kg
Cobalt-60	1.235×10^{-7}	kg	Hydrocarbons	1.294×10^{-6}	kg	Thallium	1.537×10^{-10}	kg
Copper	1.134×10^{-7}	kg	Propane	1.729×10^{-3}	kg	Tin	3.315×10^{-8}	kg
Cyanide	2.085×10^{-7}	kg	Propene	3.484×10^{-8}	kg	HCFC-22	4.068×10^{-9}	kg
Cyclohexane	6.272×10^{-11}	kg	Propionic acid	5.220×10^{-11}	kg	Sulfur dioxide	9.200×10^{-3}	kg
dibenz[a,h]anthracene	1.086×10^{-10}	kg	Radon-222	1.524×10	kBq			
Ethane	9.560×10^{-4}	kg	rhodium	4.200×10^{-17}	kg	tellurium	2.094×10^{-11}	kg
Ethanol	1.390×10^{-7}	kg	Scandium	2.548×10^{-13}	kg	Fluorine	5.192×10^{-9}	kg
Benzene, ethyl-	4.446×10^{-7}	kg	Selenium	7.321×10^{-8}	kg	Formaldehyde	1.399×10^{-9}	kg
Ethene	1.054×10^{-7}	kg	Silver	1.248×10^{-17}	kg	fluoranthene	1.264×10^{-9}	kg
Nitrogen oxides	4.600×10^{-3}	kg	Strontium	9.810×10^{-12}	kg	fluorene	4.011×10^{-9}	kg
P > 2.5 um, and < 10um	7.408×10^{-5}	kg	Styrene	6.946×10^{-14}	kg			
hexamethylene diamine	1.055×10^{-13}		Sulfur hexafluoride	8.970×10^{-12}	kg			

Table D.4. LCI of composting of 1 tonne of waste
(Data extracted from DTU 2017)

Parameter	Value	Unit	Parameter	Value	Unit	Parameter	Value	Unit
Emission to air								
CO, fossil	1.736×10^{-1}	kg	Hexane	4.686×10^{-6}	kg	Vanadium	1.666×10^{-6}	kg
1,3,5-trimethylbenzene	2.781×10^{-4}	kg	hydrocyanic acid	2.297×10^{-11}	kg	Ethene chloro-	8.088×10^{-9}	kg
Dioxins	5.421×10^{-15}	kg	Hydrogen	1.150×10^{-6}	kg	Heat, waste	1.708 x 10	MJ
Acetaldehyde	1.117×10^{-7}	kg	Arsine	1.528×10^{-11}	kg	Xenon-131m	2.151×10^{-5}	kBq
Acetic acid	1.031×10^{-7}	kg	hydrogen bromide	1.299×10^{-10}	kg	Xenon-133	3.521×10^{-3}	kBq
Acetone	9.984×10^{-8}	kg	Hydrogen chloride	4.112×10^{-3}	kg	Xenon-135	1.164×10^{-3}	kBq
acid (as H+)	1.230×10^{-10}	kg	Hydrogen fluoride	4.529×10^{-7}	kg	Xenon-137	3.051×10^{-7}	kBq
Acrolein	6.898×10^{-10}	kg	hydrogen iodide	1.155×10^{-13}	kg	Xenon-138	3.930×10^{-5}	kBq
Ammonia	1.742×10^{-1}	kg	Hydrogen sulfide	6.811×10^{-6}	kg	Xylene	4.388×10^{-7}	kg
ammonium	5.281×10^{-13}	kg	Hydrogen-3	3.037×10^{-3}	kBq	Zinc	7.750×10^{-8}	kg
anthracene	9.775×10^{-11}	kg	Indeno-pyrene	3.265×10^{-11}	kg	zinc oxide	5.710×10^{-15}	kg
Antimony	5.698×10^{-10}	kg	Iodine-129	1.528×10^{-6}	kBq	CO_2, fossil	4.469 x 10	kg
Antimony-124	2.472×10^{-10}	kBq	Iodine-131	2.297×10^{-7}	kBq	CH_4, fossil	4.663×10^{-3}	kg
Argon-41	1.559×10^{-3}	kBq	Iron	1.348×10^{-7}	kg	Nitrogen oxides	3.504×10^{-2}	kg
Arsenic	1.401×10^{-8}	kg	Krypton-85	2.633×10^{1}	kBq	Diethylamine	1.226×10^{-17}	kg
arsenic trioxide	1.842×10^{-13}	kg	Lead	7.038×10^{-8}	kg	Water	1.400×10^{-1}	kg
Barium	2.101×10^{-6}	kg	lead dioxide	3.281×10^{-14}	kg	Nitrogen	7.456×10^{-5}	kg
Benzene	1.741×10^{-6}	kg	Manganese	6.287×10^{-9}	kg	Dioxins	1.393×10^{-10}	kg
benzo[a]anthracene	4.918×10^{-11}	kg	Mercury	4.359×10^{-9}	kg	Cobalt	3.656×10^{-9}	kg
Benzo(a)pyrene	2.669×10^{-11}	kg	Methanol	3.220×10^{-8}	kg	Hydrocarbons	8.890×10^{-8}	kg
benzo[g,h,i]perylene	4.388×10^{-11}	kg	Molybdenum	1.050×10^{-8}	kg			
benzo[k]fluoranthene	8.775×10^{-11}	kg	naphthalene	1.026×10^{-8}	kg	Benzene	4.445×10^{-9}	kg
Beryllium	1.707×10^{-10}	kg	Butane	9.031×10^{-5}	kg	Dinitrogen monoxide	2.786×10^{-7}	kg
Boron	1.047×10^{-7}	kg	Nickel	2.576×10^{-7}	kg	Nickel	2.883×10^{-8}	kg
Bromine	2.909×10^{-8}	kg	nitrogen monoxide	9.969×10^{-12}	kg	Hydrocarbons, aromatic	8.890×10^{-10}	kg
Butadiene	4.518×10^{-13}	kg	NMVOC	3.434×10^{-4}	kg	Iron	9.750×10^{-9}	kg
Cadmium	4.558×10^{-9}	kg	octane	1.737×10^{-6}	kg	Calcium	4.178×10^{-9}	kg
Carbon disulfide	6.891×10^{-13}	kg	oxygen	4.913×10^{-4}	kg	Benzopyrene	2.089×10^{-12}	kg
CO, non-fossil	5.484×10^{-4}	kg	palladium	1.096×10^{-17}	kg	Manganese	1.149×10^{-9}	kg
Carbon-14	7.151×10^{-4}	kBq	P > 10 um	1.054×10^{-12}	kg	Selenium	5.223×10^{-10}	kg
Cesium-134	1.957×10^{-7}	kBq	P > 2.5 um, and < 10um	1.764×10^{-3}	kg	Copper	4.910×10^{-9}	kg
Cesium-137	3.999×10^{-7}	kBq	P < 2.5 um	1.708×10^{-5}	kg	Molybdenum	7.939×10^{-10}	kg
Methane, trichlorofluoro-, CFC-11	4.360×10^{-9}	kg	Pentane	3.070×10^{-5}	kg	P > 10 um	5.223×10^{-7}	kg
Dinitrogen monoxide	1.170×10^{-4}	kg	phenanthrene	3.224×10^{-9}	kg	Acetone	2.177×10^{-8}	kg
Methane, chlorodifluoro-, HCFC-22	9.373×10^{-10}	kg	Phenol	1.351×10^{-12}	kg	Acetic acid	8.890×10^{-8}	kg
Methane, chlorotrifluoro-, CFC-13	5.886×10^{-10}	kg	Phosphine	3.661×10^{-14}	kg	Acetaldehyde	2.177×10^{-8}	kg
Chlorine	2.593×10^{-6}	kg	Plutonium-alpha	4.592×10^{-11}	kBq	P < 2.5 um	9.402×10^{-7}	kg
Chromium	4.127×10^{-8}	kg	Polychlorinated biphenyls	3.350×10^{-11}	kg	Sulfur dioxide	1.289×10^{-4}	kg
chromium III	3.955×10^{-11}	kg	Hydrocarbons, aromatic	3.259×10^{-7}	kg	Ethanol	4.445×10^{-8}	kg
Chrysene	1.208×10^{-10}	kg	Propane	4.354×10^{-4}	kg	Zinc	2.507×10^{-9}	kg
Cobalt	2.083×10^{-8}	kg	Propene	8.774×10^{-9}	kg	Formaldehyde	6.668×10^{-8}	kg
Cobalt-58	1.226×10^{-9}	kBq	Propionic acid	1.315×10^{-11}	kg	Cadmium	3.301×10^{-10}	kg
Cobalt-60	3.110×10^{-8}	kBq	Radon-222	3.837×10^{-1}	kBq	Hydrogen fluoride	4.192×10^{-9}	kg
Copper	2.857×10^{-8}	kg	rhodium	1.058×10^{-17}	kg	Vanadium	1.045×10^{-7}	kg
Cyanide	5.250×10^{-8}	kg	Scandium	6.416×10^{-14}	kg	Propane	4.445×10^{-9}	kg
Cyclohexane	1.580×10^{-11}	kg	Selenium	1.844×10^{-8}	kg	Chromium VI	4.335×10^{-11}	kg
dibenz[a,h]anthracene	2.735×10^{-11}	kg	Silver	3.142×10^{-18}	kg	CO, fossil	1.393×10^{-6}	kg
HCC-30	1.531×10^{-16}	kg	Strontium	2.470×10^{-12}	kg	Beryllium	7.768×10^{-12}	kg
Ethane	2.408×10^{-4}	kg	Styrene	1.749×10^{-14}	kg			
Ethanol	3.501×10^{-8}	kg	Sulfate	2.287×10^{-10}	kg	Methanol	4.445×10^{-8}	kg

Parameter	Value	Unit	Parameter	Value	Unit	Parameter	Value	Unit
Benzene, ethyl-	1.120×10^{-7}	kg	Sulfur dioxide	2.343×10^{-3}	kg	Sodium	4.178×10^{-8}	kg
Ethene	2.655×10^{-8}	kg	Sulfur hexafluoride	2.259×10^{-12}	kg	Lead	3.552×10^{-9}	kg
fluoranthene	3.184×10^{-10}	kg	tellurium	5.273×10^{-11}	kg	P > 2.5 um, and < 10um	6.268×10^{-7}	kg
fluorene	1.010×10^{-9}	kg	Thallium	3.871×10^{-11}	kg	PAH	2.507×10^{-10}	kg
Fluorine	1.307×10^{-6}	kg	Tin	8.347×10^{-9}	kg	Hydrogen chloride	6.282×10^{-9}	kg
Formaldehyde	3.522×10^{-7}	kg	tin oxide	2.855×10^{-15}	kg	Mercury	2.089×10^{-11}	kg
			Titanium	7.474×10^{-12}	kg	Nitrogen oxides	3.343×10^{-5}	kg
Helium	3.081×10^{-9}	kg	Toluene	8.845×10^{-8}	kg	Ammonia	1.393×10^{-8}	kg
Heptane	3.158×10^{-6}	kg	Uranium-234	1.668×10^{-6}	kBq	Chromium	1.179×10^{-9}	kg
hexamethylene diamine	2.657×10^{-14}	kg	Uranium-235	6.430×10^{-6}	kBq	Arsenic	6.895×10^{-10}	kg
Hexane	4.686×10^{-6}	kg	Uranium-238	9.445×10^{-6}	kBq	CH_4 non-fossil	1.203 x 10	kg
hydrocyanic acid	2.297×10^{-11}	kg	used air	2.422×10^{-1}	kg	CO_2 non-fossil	$1.444 \times 10^{+2}$	kg
Emission to water								
1,2-dibromoethane, ground	3.710×10^{-15}	kg	Fluoride, ground	7.846×10^{-6}	kg	VOC, ocean	6.744×10^{-9}	kg
1,2-dichloropropane, ground	2.133×10^{-17}	kg	fluorine, ground	2.063×10^{-9}	kg	Heat, waste, ground	7.360×10^{-2}	MJ
Tetrachlorodibenzo-p-dioxin, ground	3.669×10^{-22}	kg	Hexane, ocean	1.770×10^{-13}	kg	Xylene, ocean	1.920×10^{-6}	kg
Acenaphthene, ocean	1.453×10^{-8}	kg	Hexane, ground	2.323×10^{-13}	kg	zinc, ground	7.738×10^{-8}	kg
Acenaphthylene, ocean	5.535×10^{-9}	kg	Hydrocarbons, ground	5.123×10^{-9}	kg	Iron, ground	1.165×10^{-5}	kg
Acetic acid, ocean	4.575×10^{-8}	kg	hydrogen chloride, ground	4.000×10^{-11}	kg	Nickel, ground	1.477×10^{-7}	kg
Acidity, ground	6.224×10^{-9}	kg	hydrogen fluoride, ground	3.304×10^{-10}	kg	Zinc, ion, ocean	7.122×10^{-6}	kg
Acrylonitrile, ground	1.560×10^{-12}	kg	Hydrogen-3, Tritium, ground	1.041 x 10	kBq	Iron, ion, ocean	4.372×10^{-6}	kg
AOX, ocean	6.114×10^{-13}	kg	Hydroxide, ground	1.039×10^{-9}	kg	Ammonium, ion, ground	3.659×10^{-6}	kg
Aluminium, ocean	7.235×10^{-12}	kg	Iodine-129, ground	1.019×10^{-4}	kBq	Calcium, ion, ground	3.091×10^{-6}	kg
Americium-241, ground	7.046×10^{-7}	kBq	Iodine-131, ground	5.228×10^{-9}	kBq	Copper, ion, ocean	3.985×10^{-7}	kg
ammonia, ocean	2.150×10^{-10}	kg	Lead, ocean	7.923×10^{-8}	kg	Nickel, ion, ocean	2.853×10^{-7}	kg
anthracene, ocean	3.750×10^{-9}	kg	Magnesium, ocean	1.208×10^{-7}	kg	Arsenic, ion, ocean	2.502×10^{-7}	kg
Antimony, ground	5.424×10^{-15}	kg	Manganese, ocean	4.574×10^{-7}	kg	Vanadium, ion, ocean	2.439×10^{-4}	kg
Antimony-124, ground	7.327×10^{-9}	kBq	Manganese-54, ground	2.381×10^{-5}	kBq	Tin, ion, surface	1.664×10^{-11}	kg
Antimony-125, ground	4.993×10^{-9}	kBq	Mercury, ocean	2.657×10^{-9}	kg	AOX, surface	5.906×10^{-10}	kg
arsenic, ocean	2.502×10^{-7}	kg	Methanol, ground	7.109×10^{-8}	kg	Iron, ion, surface	1.750×10^{-8}	kg
Barium, ocean	1.405×10^{-5}	kg	Molybdenum, ocean	6.209×10^{-13}	kg	Cobalt, surface	3.327×10^{-11}	kg
Benzene, ocean	2.809×10^{-6}	kg	naphthalene, ocean	4.766×10^{-7}	kg	Vanadium, ion, surface	5.032×10^{-9}	kg
benzo[a]anthracene, ocean	3.262×10^{-9}	kg	Nitrate, ocean	1.146×10^{-6}	kg	DOC,surface	1.431×10^{-7}	kg
benzo[k]fluoranthene, ocean	3.626×10^{-9}	kg	Nitrogen, ground	1.687×10^{-5}	kg	Arsenic, ion, surface	1.664×10^{-11}	kg
Beryllium, ocean	2.033×10^{-8}	kg	particles (> PM10), ocean	5.367×10^{-4}	kg	Oils, surface	7.894×10^{-9}	kg
BOD5, ocean	6.744×10^{-7}	kg	PM10, ground	2.769×10^{-12}	kg	Fluoride, surface	2.828×10^{-8}	kg
Boron, ocean	1.170×10^{-10}	kg	Phenol, ocean	6.182×10^{-6}	kg	Manganese, surface	4.991×10^{-10}	kg
Bromine, ground	2.671×10^{-10}	kg	Phosphate, ground	3.376×10^{-6}	kg	SS, surface	2.118×10^{-7}	kg
cadmium, ocean	1.136×10^{-7}	kg	Plutonium-alpha, ground	2.805×10^{-6}	kBq	Nitrogen, surface	1.137×10^{-6}	kg
calcium, ocean	1.278×10^{-8}	kg	Hydrocarbons, aromatic, ground	1.162×10^{-8}	kg	Hypochlorite,surface	4.596×10^{-8}	kg
Carbon-14, ground	3.567×10^{-5}	kBq	potassium, ground	1.158×10^{-7}	kg	Water	6.850×10^{-4}	m3

Parameter	Value	Unit	Parameter	Value	Unit	Parameter	Value	Unit
Carbonate, ocean	8.838×10^{-4}	kg	R-40, ground	1.108×10^{-10}	kg	Sulfide, surface	7.486×10^{-9}	kg
Cesium-134, ground	3.582×10^{-5}	kBq	Radium-226, ground	1.161×10^{-2}	kBq	Cadmium, ion, surface	4.048×10^{-11}	kg
Cesium-137, ground	3.312×10^{-4}	kBq	Ruthenium-106, ground	7.046×10^{-7}	kBq	Nickel, ion, surface	2.461×10^{-9}	kg
COD	7.459×10^{-5}	kg	Selenium, ground	1.768×10^{-8}	kg	TOC, surface	1.431×10^{-7}	kg
Chloride, ocean	6.979×10^{-2}	kg	silver, ocean	1.842×10^{-12}	kg	Sulfite, surface	2.512×10^{-7}	kg
Chlorine, ground	5.646×10^{-7}	kg	Silver-110, ground	1.071×10^{-9}	kBq	Copper, ion, surface	3.533×10^{-10}	kg
chromium, ocean	3.985×10^{-7}	kg	sodium, ocean	1.347×10^{-5}	kg	Chloride, surface	3.404×10^{-6}	kg
Chromium, ion, ground	1.273×10^{-9}	kg	Strontium, ocean	1.242×10^{-7}	kg	COD, , surface	2.559×10^{-7}	kg
Chromium VI, ground	2.152×10^{-17}	kg	Strontium-90, ground	3.403×10^{-5}	kBq	Chromium, ion, surface	4.265×10^{-10}	kg
chrysene, ocean	1.843×10^{-8}	kg	Sulfate, ocean	3.723×10^{-4}	kg	BOD5, surface	2.340×10^{-7}	kg
Cobalt, ocean	3.558×10^{-7}	kg	Sulfide, ocean	1.609×10^{-4}	kg	Sulfate, surface	2.782×10^{-5}	kg
Cobalt-58, ground	2.739×10^{-7}	kBq	Sulfur, ocean	6.260×10^{-11}	kg	Thallium, surface	5.157×10^{-10}	kg
Cobalt-60, ground	1.536×10^{-4}	kBq	Thallium, ground	6.456×10^{-12}	kg	Lead, surface	6.949×10^{-10}	kg
cresol, ocean	1.622×10^{-12}	kg	tin, ocean	2.206×10^{-12}	kg	Hydrocarbons, surface	5.169×10^{-9}	kg
Curium alpha, ground	9.339×10^{-7}	kBq	titanium, ocean	2.248×10^{-13}	kg	Zinc, ion, surface	8.789×10^{-10}	kg
Cyanide, ground	1.553×10^{-8}	kg	Toluene, ocean	1.538×10^{-6}	kg	Mercury, surface	1.747×10^{-11}	kg
decane, ocean	2.653×10^{-5}	kg	TOC, ocean	6.744×10^{-7}	kg	Phosphorus, surface	2.579×10^{-9}	kg
Benzene, ethyl-, ocean	3.426×10^{-7}	kg	Uranium-238, ground	2.036×10^{-4}	kBq			
fluoranthene, ocean	$3. \times 10^{-9}$	kg	vanadium, ocean	2.440×10^{-7}	kg			

Table D.5. LCI of anaerobic digestion of 1 tonne of waste
(Data extracted from DTU 2017)

Parameter	Value	Unit	Parameter	Value	Unit	Parameter	Value	Unit
Emission to air								
1,3,5-trimethylbenzene	8.344 x 10^{-15}	kg	Mercury	4.394 x 10^{-8}	kg	Acetone	-1.25 x 10^{-4}	kg
Dioxins	1.167 x 10^{-10}	kg	Methanol	2.820 x 10^{-8}	kg	Acetic acid	-5.13 x 10^{-4}	kg
Acetaldehyde	4.342 x 10^{-8}	kg	Molybdenum	8.552 x 10^{-9}	kg	Hydrocarbons	-2.56 x 10^{-5}	kg
Acetic acid	9.739 x 10^{-8}	kg	naphthalene	3.079 x 10^{-9}	kg	Acetaldehyde	-1.25 x 10^{-4}	kg
Acetone	3.962 x 10^{-8}	kg	Butane	8.410 x 10^{-5}	kg	Pa < 2.5 um	-5.42 x 10^{-3}	kg
acid (as H+)	3.689 x 10^{-11}	kg	Nickel	7.735 x 10^{-8}	kg	Sulfur dioxide	-7.44 x 10^{-1}	kg
Acrolein	5.012 x 10^{-10}	kg	nitrogen monoxide	2.991 x 10^{-12}	kg	Ethanol	-2.56 x 10^{-4}	kg
Ammonia	2.010 x 10^{-2}	kg	Dinitrogen monoxide	2.262 x 10^{-2}	kg	Zinc	-1.44 x 10^{-5}	kg
ammonium	1.584 x 10^{-13}	kg	NMVOC	5.670 x 10^{-6}	kg	Formaldehyde	-3.84 x 10^{-4}	kg
anthracene	2.932 x 10^{-11}	kg	octane	5.212 x 10^{-7}	kg	Cadmium	-1.90 x 10^{-6}	kg
Antimony	5.631 x 10^{-10}	kg	oxygen	1.474 x 10^{-4}	kg	Hydrogen fluoride	-2.41 x 10^{-5}	kg
Antimony-124	8.712 x 10^{-9}	kBq	palladium	3.287 x 10^{-18}	kg	Vanadium	-6.02 x 10^{-4}	kg
Argon-41	7.874 x 10^{-4}	kBq	P > 10 um	3.237 x 10^{-6}	kg	Propane	-2.56 x 10^{-5}	kg
Arsenic	4.204 x 10^{-9}	kg	Particulates, > 2.5 um, and < 10um	7.601 x 10^-03	kg	Chromium VI	-2.50 x 10^E-07	kg
arsenic trioxide	5.525 x 10^-14	kg	P < 2.5 um	2.097 x 10^{-5}	kg	Carbon monoxide, fossil	-8.03 x 10^{-3}	kg
Barium	6.466 x 10^{-7}	kg	Pentane	2.875 x 10^{-5}	kg	Beryllium	-4.48 x 10^{-8}	kg
Benzene	2.064 x 10^{-6}	kg	phenanthrene	9.673 x 10^{-10}	kg	Carbon dioxide, fossil	-6.39 x 10	kg
benzo[a]anthracene	1.475 x 10^{-11}	kg	Phenol	4.964 x 10^{-12}	kg	Methanol	-2.56 x 10^{-4}	kg
Benzo(a)pyrene	2.951 x 10^{-11}	kg	Phosphine	7.460 x 10^{-14}	kg	Sodium	-2.41 x 10^{-4}	kg
benzo[g,h,i]perylene	1.316 x 10^{-11}	kg	Plutonium-alpha	1.378 x 10^{-11}	kBq	Lead	-2.05 x 10^{-5}	kg
benzo[k]fluoranthene	2.633 x 10^{-11}	kg	Polychlorinated biphenyls	3.242 x 10^{-5}	kg	P, > 2.5 um, and < 10um	-3.61 x 10^{-3}	kg
Beryllium	1.487 x 10^{-10}	kg	Hydrocarbons, aromatic	9.776 x 10^{-8}	kg	PAH	-1.44 x 10^{-6}	kg
Boron	1.006 x 10^{-7}	kg	Propane	4.035 x 10^{-4}	kg	Hydrogen chloride	-3.62 x 10^{-5}	kg
Bromine	2.607 x 10^{-8}	kg	Propene	8.587 x 10^{-9}	kg	Mercury	-1.20 x 10^{-7}	kg
Butadiene	1.356 x 10^{-13}	kg	Propionic acid	3.507 x 10^{-11}	kg	Nitrogen oxides	-1.92 x 10^{-1}	kg
Cadmium	5.011 x 10^{-9}	kg	Radon-222	3.623 x 10^{-1}	kBq	Ammonia	-8.03 x 10^{-5}	kg
Carbon disulfide	2.071 x 10^{-13}	kg	rhodium	3.173 x 10^{-18}	kg	Chromium	-6.80 x 10^{-6}	kg
Carbon monoxide, non-fossil	2.745 x 10^{-1}	kg	Scandium	1.926 x 10^{-14}	kg	Arsenic	-3.97 x 10^{-6}	kg
Carbon-14	6.180 x 10^{-4}	kBq	Selenium	1.538 x 10^{-8}	kg	Carbon dioxide, non-fossil	5.473 x 10	kg
Cesium-134	6.034 x 10^{-8}	kBq	Silver	1.511 x 10^{-11}	kg	PAH	3.213 x 10^{-5}	kg
Cesium-137	1.214 x 10^{-7}	kBq	Strontium	7.418 x 10^{-13}	kg	Methane, non-fossil	2.812 x 10	kg
CFC-11	1.308 x 10^{-9}	kg	Styrene	3.717 x 10^{-13}	kg	Ethene, trichloro-	1.487 x 10^{-12}	kg
CFC-114	2.519 x 10^{-9}	kg	Sulfate	1.010 x 10^{-9}	kg	Ethene, tetrachloro-	3.042 x10^{-18}	kg
CFC-12	2.812 x 10^{-10}	kg	Sulfur dioxide	-8.07 x 10^{-3}	kg	Cumene	7.389 x10^{-17}	kg
methane, chlorotrifluoro-, CFC-13	1.766 x 10^{-10}	kg	Sulfur hexafluoride	8.249 x 10^{-13}	kg	Aluminium	1.511 x10^{-11}	kg
Chlorine	7.856 x 10^{-7}	kg	tellurium	1.582 x 10^{-12}	kg	Radium-226	3.324 x 10^{-6}	kBq
Chromium	3.817 x 10^{-8}	kg	Thallium	3.557 x 10^{-11}	kg	Indeno(1,2,3-cd)pyrene	1.393 x10^{-11}	kg
chromium III	1.187 x10^{-11}	kg	Tin	7.669 x10^{-9}	kg	Polonium-210	7.609 x10^{-7}	kBq
chrysene	3.624 x10^{-11}	kg	tin oxide	8.565 x10^{-16}	kg	Methyl acrylate	1.696 x10^{-15}	kg
Cobalt	1.223 x10^{-8}	kg	Titanium	8.392 x10^{-10}	kg	2-Propanol	2.832 x10^{-11}	kg
Cobalt-58	4.646 x10^{-9}	kBq	Toluene	6.826 x10^{-8}	kg	Chrysene	5.154 x10^{-11}	kg
Cobalt-60	1.860 x10^{-8}	kBq	Uranium-234	7.343 x10^{-7}	kBq	Ethane, 1,2-dichloro-	2.908 x10^{-17}	kg
Copper	2.873 x10^{-8}	kg	Uranium-235	1.966 x10^{-6}	kBq	Silicon tetrafluoride	2.640 x10^{-14}	kg

Cyanide	4.991 x10^{-8}	kg	Uranium-238	3.124 x10^{-6}	kBq	HFC-134a	3.291 x10^{-12}	kg
Cyclohexane	4.748 x10^{-12}	kg	used air	7.267 x10^{-2}	kg	Naphtalene	4.379 x10^{-9}	kg
dibenz[a,h]anthracene	8.204 x10^{-12}	kg	Vanadium	6.927 x10^{-7}	kg	Fluorene	4.310 x10^{-10}	kg
HCC-30	1.961 x10^{-16}	kg	Ethene, chloro-	2.504 x10^{-9}	kg	Dimethylamine	7.538 x10^{-16}	kg
Ethane	2.253 x10^{-4}	kg	Heat, waste	1.342 x 10	MJ	HFC-125	5.271 x10^{-12}	kg
Ethanol	2.911 x10^{-8}	kg	Xenon-131m	6.453 x10^{-6}	kBq	Anthracene	4.170 x10^{-11}	kg
Benzene, ethyl-	9.918 x10^{-8}	kg	Xenon-133	4.586 x10^{-3}	kBq	Benz(a)anthracene	2.098 x10^{-11}	kg
Ethene	8.113 x10^{-9}	kg	Xenon-135	4.823 x10^{-3}	kBq	Phenanthrene	1.376 x10^{-9}	kg
Methane, tetrafluoro-, R-14	2.759 x10^{-11}	kg	Xenon-137	6.745 x10^{-4}	kBq	Fluoranthene	1.358 x10^{-10}	kg
fluoranthene	9.551 x10^{-11}	kg	Xenon-138	7.521 x10^{-4}	kBq	Methane, trifluoro-, HFC-23	3.611 x10^{-11}	kg
fluorene	3.030 x10^{-10}	kg	Xylene	4.155 x10^{-7}	kg	HCFC-140	2.764 x10^{-13}	kg
Fluorine	3.922 x10^{-7}	kg	Zinc	8.016 x10^{-8}	kg	HFC-32	7.907 x10^{-13}	kg
Formaldehyde	-2.15 x10^{-5}	kg	zinc oxide	1.713 x10^{-15}	kg	Butene	4.168 x10^{-9}	kg
HCFC-22	3.087 x10^{-10}	kg	Carbon dioxide, fossil	-1.61 x 10	kg	Butyl acetate	8.129 x10^{-13}	kg
Helium	9.255 x10^{-10}	kg	Methane, fossil	-3.86 x10^{-4}	kg	Sulfur trioxide	7.523 x10^{-15}	kg
Heptane	2.910 x10^{-6}	kg	Nitrogen oxides	2.742 x 10	kg	Benzo(k)fluoranthene	3.744 x10^{-11}	kg
hexamethylene diamine	7.972 x10^{-15}	kg	Diethylamine	6.637 x10^{-18}	kg	Benzo(ghi)perylene	1.872 x10^{-11}	kg
Hexane	2.371 x10^{-5}	kg	Water	-6.20 x10^{-3}	kg	Lead-210	5.073 x10^{-7}	kBq
hydrocyanic acid	6.891 x10^{-12}	kg	Nitrogen	4.262 x10^{-5}	kg	Nitrogen fluoride	1.505 x10^{-13}	kg
Hydrogen	5.404 x10^{-6}	kg	Carbon monoxide, fossil	7.451 x10^{-3}	kg	Thorium-234	2.596 x10^{-10}	kBq
Arsine	1.907 x10^{-11}	kg	Dioxins	-8.03 x10^{-12}	kg	Thorium-230	7.610 x10^{-7}	kBq
hydrogen bromide	3.896 x10^{-11}	kg	Cobalt	-2.11 x10^{-5}	kg	Protactinium-234	2.533 x10^{-10}	kBq
Hydrogen chloride	4.876 x10^{-3}	kg	Hydrocarbons,	-5.13 x10^{-4}	kg	Nitrate	1.691 x10^{-17}	kg
Hydrogen fluoride	1.297 x10^{-3}	kg	Methane, fossil	-2.56 x10^{-3}	kg	Chromium VI	3.606 x10^{-16}	kg
hydrogen iodide	3.465 x10^{-14}	kg	Benzene	-2.56 x10^{-5}	kg	Ethane thiol	1.020 x10^{-9}	kg
Hydrogen sulfide	8.167 x10^{-6}	kg	Dinitrogen monoxide	-1.60 x10^{-5}	kg	HFC-116	5.603 x10^{-13}	kg
Hydrogen-3, Tritium	2.152 x10^{-3}	kBq	Nickel	-1.66 x10^{-4}	kg	Dibenz(a,h)anthracene	1.167 x10^{-11}	kg
indeno(1,2,3-cd)pyrene	9.795 x10^{-12}	kg	Hydrocarbons, aromatic	-5.13 x10^{-6}	kg	carbon monoxide	4.330 x10^{-4}	kg
Iodine-129	4.668 x10^{-7}	kBq	Iron	-5.62 x10^{-5}	kg	Methane, fossil	-1.86 x10^{-4}	kg
Iodine-131	6.319 x10^{-6}	kBq	Calcium	-2.41 x10^{-5}	kg	Molybdenum	-4.58 x10^{-6}	kg
Iron	1.560 x10^{-7}	kg	Benzo(a)pyrene	-1.20 x10^{-8}	kg	lead dioxide	9.844 x10^{-15}	kg
Krypton-85	7.911 x 10	kBq	Selenium	-3.01 x10^{-6}	kg	Manganese	6.905 x10^{-8}	kg
Lead	9.129 x10^{-8}	kg	Copper	-2.83 x10^{-5}	kg	P> 10 um	-3.01 x10^{-3}	kg
Emission to water								
1,2-dibromoethane, ground	1.113 x10^{-15}	kg	fluoranthene, ground	1.096 x10^{-11}	kg	VOC, surface	2.149 x10^{-11}	kg
1,2-dichloropropane, ground	6.399 x10^{-18}	kg	Fluoride, ground	2.354 x10^{-6}	kg	Uranium-234, surface	3.456 x10^{-6}	kBq
2,3,7,8-tetrachlorodibenzo-p-dioxin, ground	1.101 x10^{-22}	kg	fluorine, ground	6.189 x10^{-10}	kg	Xylene, surface	1.039 x10^{-7}	kg
Acenaphthene, ground	7.335 x10^{-11}	kg	Hexane, ground	6.970 x10^{-14}	kg	Americium-surface	3.804 x 10^-09	kBq
Acenaphthylene, ground	3.101 x10^{-11}	kg	Hydrocarbons, ground	1.537 x10^{-9}	kg	Strontium, surface	4.614 x10^{-7}	kg
Acetic acid, ground	2.099 x 10^-07	kg	hydrogen chloride, ground	1.200 x10^{-11}	kg	Silver, ion, surface	2.679 x10^{-11}	kg
Acidity, unspecified, ground	1.867 x 10^-09	kg	hydrogen fluoride, ground	9.912 x10^{-11}	kg	Ruthenium-106, ocean	8.782 x10^{-5}	kBq
Acrylonitrile, ground	4.680 x 10^-13	kg	Hydrogen-3, Tritium, ground	3.122 x10^{-1}	kBq	Ruthenium-106, surface	9.067 x10^{-7}	kBq
AOX, ground	3.308 x10^{-7}	kg	Hydroxide, ground	3.118 x10^{-10}	kg	Protactinium-234, surface	1.184 x10^{-6}	kBq
Aluminium, ground	1.189 x10^{-7}	kg	Iodine-129, ground	3.057 x10^{-5}	kBq	Magnesium, surface	1.86 x10^{-6}	kg
Americium-241, ground	2.114 x10^{-7}	kBq	Iodine-131, ground	1.568 x10^{-9}	kBq	Ammonium, ion, surface	1.574 x10^{-14}	kg

ammonia, ocean	6.450 x10^{-11}	kg	Lead, ground	3.533 x10^{-8}	kg	Sodium, ion, ocean	6.77 x10^{-6}	kg
anthracene, ground	1.221 x10^{-10}	kg	potassium, ground	3.473 x10^{-8}	kg	Methanol, surface	1.44 x10^{-6}	kg
Antimony, ground	1.627 x10^{-15}	kg	R-40, ground	3.324 x10^{-11}	kg	Barium, surface	1.273 x10^{-6}	kg
Antimony-124, ground	2.198 x10^{-9}	kBq	Radium-226, ground	3.483 x10^{-3}	kBq	Iodine-131, surface	4.014 x10^{-8}	kBq
Antimony-125, ground	1.498 x10^{-9}	kBq	Ruthenium-106, ground	2.11 x10^{-7}	kBq	Iodine-129, surface	5.511 x10^{-7}	kBq
arsenic, ground	6.091 x10^{-8}	kg	Selenium, ground	5.303 x10^{-9}	kg	Cyanide, surface	8.839 x10^{-9}	kg
Barium, ocean	1. x10^{-5}	kg	silver, ground	5.071 x10^{-12}	kg	Chrysene, surface	8.726 x10^{-11}	kg
Barium, ground	4.159 x10^{-7}	kg	silver, ocean	5.526 x10^{-13}	kg	Cobalt-60, ocean	3.122 x10^{-6}	kBq
Benzene, ground	1.528 x10^{-7}	kg	Silver-110, ground	3.21 x10^{-10}	kBq	Cobalt-60, surface	9.980 x10^{-7}	kBq
Benzene, ocean	3.147 x10^{-6}	kg	sodium, ground	2.540 x10^{-5}	kg	Beryllium, surface	7.103 x10^{-12}	kg
benzo[a]anthracene, ground	9.406 x10^{-12}	kg	Strontium, ground	9.472 x10^{-7}	kg	Molybdenum, surface	1.162 x10^{-8}	kg
benzo[a]anthracene, ocean	9.787 x10^{-10}	kg	Strontium, ocean	4.427 x10^{-8}	kg	Ethane, 1,2-dichloro-, surface	4.522 x10^{-19}	kg
benzo[k]fluoranthene, ground	3.223 x10^{-12}	kg	Strontium-90, ground	1.02 x10^{-5}	kBq	Aluminium, surface	2.851 x10^{-7}	kg
benzo[k]fluoranthene, ocean	1.088 x10^{-9}	kg	Sulfate, ground	1.205 x10^{-4}	kg	Hexane, surface	1.295 x10^{-17}	kg
Beryllium, ocean	6.099 x10^{-9}	kg	Sulfide, ground	4.974 x10^{-6}	kg	Naphtalene, surface	1.501 x10^{-8}	kg
Beryllium, ground	4.618 x10^{-12}	kg	Sulfide, ocean	1.186 x10^{-4}	kg	Fluoranthene, surface	2.811 x10^{-11}	kg
BOD5, ground	1.542 x10^{-6}	kg	Sulfite, ground	1.440 x10^{-9}	kg	Ethene, chloro-, surface	3.805 x10^{-14}	kg
BOD6, ocean	5.415 x10^{-7}	kg	Sulfur, ground	2.443 x10^{-11}	kg	Anthracene, surface	3.918 x10^{-10}	kg
Boron, ground	4.78 x10^{-9}	kg	Thallium, ground	1.937 x10^{-12}	kg	Benzene, ethyl-, surface	2.602 x10^{-8}	kg
Boron, ocean	3.510 x10^{-11}	kg	tin, ground	9.863 x10^{-13}	kg	Benz(a)anthracene, surface	2.385 x10^{-11}	kg
Bromine, ground	8.014 x10^{-11}	kg	titanium, ground	4.211 x10^{-10}	kg	Calcium, ion, surface	2.810 x10^{-5}	kg
cadmium, ocean	3.408 x10^{-8}	kg	Toluene, ground	9.163 x10^{-8}	kg	Cadmium, ion, ocean	3.009 x10^{-7}	kg
cadmium, ground	3.861 x10^{-8}	kg	TOC, ground	7.948 x10^{-6}	kg	Strontium-90, ocean	4.415 x10^{-6}	kBq
calcium, ocean	3.833 x10^{-9}	kg	Uranium-238, ground	6.107 x10^{-5}	kBq	Formaldehyde, surface	3.420 x10^{-17}	kg
Carbon-14, ground	1.070 x10^{-5}	kBq	vanadium, ocean	7.319 x10^{-8}	kg	Manganese-54, surface	9.809 x10^{-8}	kBq
Carbonate, ground	2.615 x10^{-5}	kg	vanadium, ground	5.863 x10^{-9}	kg	Chlorine, surface	2.677 x10^{-7}	kg
Carbonate, ocean	6.512 x10^{-4}	kg	VOC, ocean	5.415 x10^{-9}	kg	Chlorate, surface	2.939 x10^{-14}	kg
Cesium-134, ground	1.07 x10^{-5}	kBq	VOC, ground	2.906 x10^{-8}	kg	Cesium-137, surface	4.637 x10^{-7}	kBq
Cesium-137, ground	9.936 x10^{-5}	kBq	Heat, waste, ground	2.208 x10^{-2}	MJ	Cesium-134, surface	8.869 x10^{-8}	kBq
COD, ocean	5.541 x10^{-5}	kg	Xylene, ground	4.077 x10^{-8}	kg	Antimony-124, surface	2.350 x10^{-7}	kBq
COD, ground	3.167 x10^{-5}	kg	zinc, ground	2.321 x10^{-8}	kg	Tungsten, surface	2.376 x10^{-12}	kg
Chloride, ground	2.305 x10^{-3}	kg	Iron, ground	3.496 x10^{-6}	kg	Benzo(k)fluoranthene, surface	2.904 x10^{-12}	kg
Chlorine, ground	1.694 x10^{-7}	kg	Nickel, ground	4.432 x10^{-8}	kg	Strontium-90, surface	2.137 x10^{-8}	kBq
chromium, ground	9.655 x10^{-8}	kg	Zinc, ion, ocean	2.140 x10^{-6}	kg	Chromium VI, surface	1.862 x10^{-10}	kg
Chromium, ion, ground	3.819 x10^{-10}	kg	Iron, ion, ocean	1.312 x10^{-6}	kg	Bromine, surface	1.352 x10^{-16}	kg
Chromium VI, ground	6.456 x10^{-18}	kg	Ammonium, ion, ground	1.098 x10^{-6}	kg	Uranium-238, surface	3.658 x10^{-6}	kBq

chrysene, ground	3.862 x10^{-11}	kg	Calcium, ion, ground	9.272 x10^{-7}	kg	Antimony-125, surface	2.648 x10^{-7}	kBq
Cobalt, ground	4.699 x 10^-11	kg	Copper, ion, ground	1.316 x10^{-7}	kg	Bromate, surface	1.459 x10^{-18}	kg
Cobalt-58, ground	8.217 x10^{-8}	kBq	Nickel, ion, ocean	8.560 x10^{-8}	kg	Boron, surface	4.026 x10^{-8}	kg
Cobalt-60, ground	4.607 x10^{-5}	kBq	Arsenic, ion, ocean	7.783 x10^{-7}	kg	Carbonate, surface	1.404 x10^{-4}	kg
cresol, ground	6.328 x10^{-13}	kg	Vanadium, ion, ocean	7.317 x10^{-5}	kg	Carbon-14, ocean	1.625 x10^{-4}	kBq
Curium alpha, ground	2.802 x10^{-7}	kBq	Arsenic, ion, ground	6.091x10^{-8}	kg	Uranium-235, surface	1.910 x10^{-7}	kBq
Cyanide, ground	4.658 x10^{-9}	kg	Tin, ion, surface	-9.60 x10^{-8}	kg	Thorium-234, surface	1.184 x10^{-6}	kBq
decane, ground	7.881 x10^{-7}	kg	AOX, surface	-3.02 x10^{-6}	kg	Titanium, ion, surface	1.418 x10^{-9}	kg
Benzene, ethyl-, ground	8.530 x10^{-9}	kg	Iron, ion, surface	-8.77 x10^{-5}	kg	Tin, ion, ocean	1.202 x10^{-16}	kg
fluoranthene, ground	1.096 x10^{-11}	kg	Cobalt, surface	-1.92 x10^{-7}	kg	Toluene, surface	2.906 x10^{-7}	kg
Fluoride, ground	2.354 x10^{-6}	kg	Vanadium, ion, surface	-2.90 x10^{-5}	kg	Thorium-230, surface	1.174 x10^{-4}	kBq
fluorine, ground	6.189 x10^{-10}	kg	DOC, surface	-8.25 x10^{4}	kg	Sulfur, surface	4.603 x10^{-15}	kg
Hexane, ground	6.970 x10^{-14}	kg	Arsenic, ion, surface	5.630 x10^{-8}	kg	Silver-110, surface	2.849 x10^{-7}	kBq
Hydrocarbons, unspecified, ground	1.537 x10^{-9}	kg	Oils, unspecified, surface	-4.55 x10^{-5}	kg	Selenium, surface	6.374 x10^{-9}	kg
hydrogen chloride, ground	1.200 x10^{-11}	kg	Fluoride, surface	-1.44 x10^{-4}	kg	Radium-226, surface	5.191 x10^{-3}	kBq
hydrogen fluoride, ground	9.912 x10^{-11}	kg	Manganese, surface	1.251 x10^{-5}	kg	PAH, surface	8.705 x10^{-10}	kg
Hydrogen-3, Tritium, ground	3.122 x10^{-1}	kBq	SS, surface	-1.22 x10^{-3}	kg	Potassium, ion, surface	7.991 x10^{-5}	kg
Hydroxide, ground	3.118 x10^{-10}	kg	Nitrogen, surface	-6.56 x10^{-3}	kg	Phosphate, ocean	4.86 x10^{-11}	kg
Iodine-129, ground	3.057 x10^{-5}	kBq	Hypochlorite, surface	-2.65 x10^{-4}	kg	Phenol, surface	5.213 x10^{-7}	kg
Iodine-131, ground	1.568 x10^{-9}	kBq	Water, unspecified	-3.95x 10	m3	Nitrite, ocean	8.852 x10^{-10}	kg
Lead, ground	3.533 x10^{-8}	kg	Sulfide, surface	-2.84x10^{-5}	kg	Nitrite, surface	1.451 x10^{-10}	kg
Magnesium, ground	2.280 x10^{-13}	kg	Cadmium, ion, surface	-1.67 x10^{-7}	kg	Cobalt-58, surface	1.397 x10^{-6}	kBq
Magnesium, ocean	9.545 x10^{-8}	kg	Nickel, ion, surface	-1.42 x10^{-5}	kg	Antimony, surface	2.006 x10^{-16}	kg
Manganese, ground	1.496 x10^{-8}	kg	TOC, surface	-3.85 x10^{-4}	kg	Iodine-131, ocean	2.854 x10^{-5}	kBq
Mercury, ground	5.388 x10^{-10}	kg	Sulfite, surface	-1.45 x10^{-3}	kg	Sodium, ion, surface	1.119 x10^{-4}	kg
Mercury, ocean	2.397 x10^{-9}	kg	Copper, ion, surface	-1.64 x10^{-6}	kg	Hydroxide, surface	1.912 x10^{-10}	kg
Methanol, ground	2.133 x10^{-8}	kg	Chloride, surface	-1.27 x10^{-2}	kg	Hydrogen-3, Tritium, surface	2.710 x10^{-2}	kBq
Molybdenum, ocean	1.863 x10^{-13}	kg	COD, surface	-1.42 x10^{-3}	kg	Hydrogen peroxide, surface	6.105 x10^{-9}	kg
Molybdenum, ground	8.172 x10^{-9}	kg	Chromium, ion, surface	-2.44 x10^{-6}	kg	Fluoride, ocean	6.988 x10^{-11}	kg
naphthalene, ground	4.761 x10^{-9}	kg	BOD5, surface	-1.34 x10^{-3}	kg	Dioxins, surface	9.902 x10^{-20}	kg
Nitrate, ground	1.107 x 10	kg	Sulfate, surface	-1.60 x10^{-1}	kg	Acetic acid, surface	2.643 x10^{-11}	kg
Nitrate, ocean	9.063 x10^{-7}	kg	Thallium, surface	-2.97 x10^{-6}	kg	Carbon-14, surface	2.926 x10^{-7}	kBq
Nitrogen, ground	5.061 x10^{-6}	kg	Lead, surface	-3.84 x10^{-6}	kg	Calcium, ion, ocean	6.968 x10^{-13}	kg
> PM10, ocean	1.610 x10^{-4}	kg	Hydrocarbons, surface	-2.98 x10^{-5}	kg	Benzene, surface	4.779 x10^{-7}	kg
PM10, ground	8.306 x10^{-13}	kg	Zinc, ion, surface	-4.95 x10^{-6}	kg	Acrylonitrile, surface	2.415 x10^{-16}	kg
Phenol, ground	2.215 x10^{-7}	kg	Mercury, surface	-9.93 x10^{-8}	kg	Acidity, surface	3.128 x10^{-10}	kg

Phosphate, ground	1.013 x10^{-6}	kg	Phosphorus, surface	-1.48 x10^{-5}	kg	Acenaphthylene, surface	8.970 x10^{-11}	kg
Plutonium-alpha, ground	8.414 x10^{-7}	kBq	Nitrate, surface	2.518 x 1	kg	Acenaphthene, surface	2.093 x10^{-10}	kg
Hydrocarbons, aromatic, ground	3.486 x10^{-9}	kg	Phosphate, surface	7.934 x10^{-6}	kg	Heat, waste, ocean	1.185 x10^{-3}	MJ
Arsenic, ion	-2.76 x10^{-6}	kg	Zinc, ion	-1.65 x10^{-5}	kg	Heat, waste, surface	3.711 x10^{-2}	MJ
Cadmium, ion	-2.76 x10^{-6}	kg	Chromium, ion	-1.38 x10^{-5}	kg	Lead	-1.18 x10^{-5}	kg
Calcium, ion	-8.29 x10^{-1}	kg	Copper, ion	-1.38 x10^{-5}	kg	Mercury	-2.62 x10^{-6}	kg
Carbonate	-1.74 x 10	kg	Fluoride	-4.61 x10^{-2}	kg	Nickel, ion	-1.10 x10^{-5}	kg
Phosphorus	-1.24 x10^{-2}	kg						

Table D.6. LCI of Incineration of 1 tonne of waste
(Data extracted from DTU 2017)

Parameter	Value	Unit	Parameter	Value	Unit	Parameter	Value	Unit
Emission to air								
Carbon monoxide, fossil	3.300 x10^{-2}	kg	Dinitrogen monoxide	4.191 x10^{-4}	kg	nitrogen monoxide	2.893 x10^{-11}	kg
1,3,5-trimethylbenzene	7.100 x10^{-13}	kg	Dioxins, measured as 2,3,7,8-tetrachlorodibenzo-p-dioxin	1.800 x10^{-11}	kg	Nitrogen oxides	8.489 x10^{-1}	kg
Acetaldehyde	4.281 x10^{-7}	kg	Ethane	2.696 x10^{-4}	kg	NMVOC	1.044 x10^{-3}	kg
Acetic acid	4.256 x10^{-6}	kg	Ethane, 1,2-dichloro-	9.230 x10^{-9}	kg	octane	6.623 x10^{-7}	kg
Acetone	4.129 x10^{-7}	kg	Ethane, 1,2-dichloro-1,1,2,2-tetrafluoro-, CFC-114	2.394 x10^{-7}	kg	oxygen	1.631 x10^{-3}	kg
acid (as H+)	1.044 x10^{-9}	kg	Ethanol	8.367 x10^{-7}	kg	PAH	4.345 x10^{-10}	kg
Acrolein	5.087 x10^{-10}	kg	Ethene	2.919 x10^{-8}	kg	palladium	4.199 x10^{-16}	kg
Aluminium	0.000 x 10	kg	Ethene, chloro-	1.205 x10^{-8}	kg	PM10	7.623 x10^{-7}	kg
Ammonia	1.418 x10^{-3}	kg	fluoranthene	2.348 x10^{-10}	kg	P < 2.5 um	1.886 x10^{-3}	kg
ammonium	6.886 x10^{-11}	kg	fluorene	7.450 x10^{-10}	kg	P > 10 um	3.000 x10^{-2}	kg
anthracene	7.209 x10^{-11}	kg	fluoride	9.679 x10^{-7}	kg	P > 2.5 um, and < 10um	8.633 x10^{-4}	kg
Antimony	3.811 x10^{-8}	kg	Fluorine	7.008 x10^{-7}	kg	Pentane	4.381 x10^{-5}	kg
Antimony-124	1.310 x10^{-8}	kBq	Formaldehyde	1.313 x10^{-5}	kg	phenanthrene	2.378 x10^{-9}	kg
Argon-41	8.260 x10^{-2}	kBq	Heat, waste	2.912 x 10	MJ	Phenol	4.985 x10^{-11}	kg
Arsenic	2.596 x10^{-7}	kg	Helium	6.040 x10^{-8}	kg	Phosphine	8.378 x10^{-13}	kg
arsenic trioxide	2.352 x10^{-13}	kg	Heptane	1.190 x10^{-6}	kg	Plutonium-alpha	5.303 x10^{-9}	kBq
Arsine	1.951 x10^{-11}	kg	hexamethylene diamine	3.262 x10^{-12}	kg	Polychlorinated biphenyls	4.250 x10^{-11}	kg
Barium	3.362 x10^{-6}	kg	Hexane	1.832 x10^{-6}	kg	Propane	2.921 x10^{-4}	kg
Benzene	3.025 x10^{-6}	kg	Hydrocarbons, aliphatic, alkanes, unspecified	-3.78 x10^{-3}	kg	Propene	5.975 x10^{-7}	kg
Benzene, ethyl-	6.569 x10^{-6}	kg	Hydrocarbons, aliphatic, unsaturated	-1.89 x10^{-4}	kg	Propionic acid	4.339 x10^{-10}	kg
Benzo(a)pyrene	1.379 x10^{-7}	kg	Hydrocarbons, aromatic	2.015 x10^{-6}	kg	Radon-222	2.034 x 10	kBq
benzo[a]anthracene	3.627 x10^{-11}	kg	hydrocyanic acid	3.157 x10^{-9}	kg	rhodium	4.044 x10^{-16}	kg
benzo[g,h,i]perylene	3.236 x10^{-11}	kg	Hydrogen	3.738 x10^{-4}	kg	Scandium	3.228 x10^{-12}	kg
benzo[k]fluoranthene	6.473 x10^{-11}	kg	hydrogen bromide	8.769 x10^{-9}	kg	Selenium	5.265 x10^{-7}	kg
Beryllium	3.715 x10^{-9}	kg	Hydrogen chloride	5.299 x10^{-3}	kg	Silver	3.430 x10^{-16}	kg
Boron	6.135 x10^{-6}	kg	Hydrogen fluoride	3.899 x10^{-4}	kg	Sodium	-1.78 x10^{-3}	kg
Bromine	2.496 x10^{-6}	kg	hydrogen iodide	9.630 x10^{-12}	kg	Strontium	1.302 x10^{-10}	kg
Butadiene	5.545 x10^{-11}	kg	Hydrogen sulfide	3.071 x10^{-5}	kg	Styrene	6.748 x10^{-12}	kg
Butane	8.213 x10^{-5}	kg	Hydrogen-3, Tritium	1.610 x10^{-1}	kBq	Sulfate	2.882 x10^{-10}	kg
Cadmium	4.027 x10^{-8}	kg	indeno(1,2,3-cd)pyrene	2.408 x10^{-11}	kg	Sulfur dioxide	2.910 x10^{-3}	kg
Calcium	-1.78 x10^{-4}	kg	Iodine-129	8.102 x10^{-5}	kBq	Sulfur hexafluoride	5.786 x10^{-11}	kg
Carbon dioxide, fossil	1.196 x10^{+1}	kg	Iodine-131	1.217 x10^{-5}	kBq	tellurium	1.366 x10^{-11}	kg
Carbon dioxide, non-fossil	5.437 x10^{2}	kg	Iron	6.852 x10^{-8}	kg	Thallium	7.191 x10^{-8}	kg
Carbon disulfide	1.666 x10^{-10}	kg	Krypton-85	1.396 x10^{+3}	kBq	Tin	1.852 x10^{-7}	kg
Carbon monoxide, non-fossil	2.484 x10^{-3}	kg	Lead	7.303 x10^{-7}	kg	tin oxide	7.29 x10^{-14}	kg
Carbon-14	3.790 x10^{-2}	kBq	lead dioxide	8.408 x10^{-13}	kg	Titanium	5.184 x10^{-10}	kg
Cesium-134	1.419 x10^{-5}	kBq	Magnesium	0.000 x 10	kg	Toluene	3.028 x10^{-6}	kg
Cesium-137	2.119 x10^{-5}	kBq	Manganese	2.602 x10^{-7}	kg	Uranium-234	8.847 x10^{-5}	kBq
chloride	3.138 x10^{-7}	kg	Mercury	3.278 x10^{-7}	kg	Uranium-235	3.409 x10^{-4}	kBq
Chlorine	4.930 x10^{-6}	kg	Methane, chlorodifluoro-, HCFC-22	5.492 x10^{-8}	kg	Uranium-238	4.562 x10^{-4}	kBq
Chromium	9.558 x10^{-8}	kg	Methane, chlorotrifluoro-, CFC-13	3.155 x10^{-8}	kg	used air	1.718 x 10	kg
chromium III	1.024 x10^{-10}	kg	Methane, dichloro-, HCC-30	5.565 x10^{-10}	kg	Vanadium	1.430 x10^{-6}	kg

Parameter	Value	Unit	Parameter	Value	Unit	Parameter	Value	Unit
Chromium VI	1.392 x10^{-8}	kg	Methane, dichlorodifluoro-, CFC-12	5.03 x10^{-8}	kg	volatile organic compound	2.782 x10^{-9}	kg
chrysene	8.909 x10^{-11}	kg	Methane, fossil	5.546 x10^{-2}	kg	Water	7.207 x 10	kg
Cobalt	2.997 x10^{-8}	kg	Methane, non-fossil	1.740 x10^{-4}	kg	Xenon-131m	1.140 x10^{-3}	kBq
Cobalt-58	6.499 x10^{-8}	kBq	Methane, tetrafluoro-, R-14	9.960 x10^{-10}	kg	Xenon-133	1.866 x10^{-1}	kBq
Cobalt-60	1.648 x10^{-6}	kBq	Methane, trichlorofluoro-, CFC-11	2.338 x10^{-7}	kg	Xenon-135	6.169 x10^{-2}	kBq
Copper	1.397 x10^{-7}	kg	Methanol	6.866 x10^{-7}	kg	Xenon-137	1.617 x10^{-5}	kBq
Cyanide	1.958 x10^{-8}	kg	Molybdenum	4.410 x10^{-9}	kg	Xenon-138	2.084 x10^{-3}	kBq
Cyclohexane	3.818 x10^{-9}	kg	naphthalene	7.570 x10^{-9}	kg	Xylene	2.747 x10^{-5}	kg
dibenz[a,h]anthracene	2.017 x10^{-11}	kg	Nickel	2.006 x10^{-6}	kg	Zinc	9.297 x10^{-7}	kg
diethylamine	1.683 x10^{-15}	kg	Nitrogen	2.006 x10^{-3}	kg	zinc oxide	1.458 x10^{-13}	kg
Emission to water								
1,2-dibromoethane, ground	8.967 x10^{-13}	kg	chromium, ground	3.634 x10^{-7}	kg	Nitrate, ground	6.465 x10^{-4}	kg
1,2-dichloropropane, ground	2.617 x10^{-15}	kg	Chromium VI, ground	4.726 x10^{-7}	kg	Nitrogen, ground	7.536 x10^{-5}	kg
2,3,7,8-tetrachlorodibenzo-p-dioxin, ground	5.892 x10^{-13}	kg	Chromium, ion, ground	6. x10^{-8}	kg	Oils, unspecified, surface	-3.36 x10^{-4}	kg
Acenaphthene, ocean	5.217 x10^{-9}	kg	chrysene, ground	5.875 x10^{-11}	kg	P (> PM10), ground	1.221 x10^{-2}	kg
Acenaphthylene, ground	5.083 x10^{-11}	kg	Cobalt, ground	1.336 x10^{-10}	kg	PM10, ground	2.370 x10^{-7}	kg
Acetic acid, ocean	1.533 x10^{-8}	kg	COD, ground	1.836 x10^{-2}	kg	Phenol, ground	3.133 x10^{-7}	kg
Acidity, unspecified, ground	2.94 x10^{-7}	kg	copper, ocean	7.241 x10^{-8}	kg	Phosphate, ground	5.719 x10^{-6}	kg
Acrylonitrile, ground	1.914 x10^{-10}	kg	Copper, ion, ground	2.361 x10^{-7}	kg	Phosphorus, surface	-1.05 x10^{-4}	kg
Aluminium, ground	2.35 x10^{-5}	kg	cresol, ocean	6.912 x10^{-12}	kg	Plutonium-alpha, ground	1.489 x10^{-4}	kBq
Americium-241, ground	3.735 x10^{-5}	kBq	Curium alpha, ground	4.941 x10^{-5}	kBq	potassium, ground	2.947 x10^{-7}	kg
ammonia, ocean	9.165 x10^{-10}	kg	Cyanide, ground	5.786 x10^{-19}	kg	Potassium, ion, surface	4.411 x10^{-3}	kg
Ammonium, ion, ground	1.281 x10^{-4}	kg	decane, ground	1.332 x10^{-6}	kg	R-40, ground	3.388 x10^{-9}	kg
anthracene, ground	2.193 x10^{-10}	kg	DOC, surface	-6.09 x10^{-3}	kg	Radium-226, ground	6. x10^{-1}	kBq
Antimony, ground	1.385 x10^{-13}	kg	Ethane, 1,2-dichloro-, ground	1.128 x10^{-12}	kg	Ruthenium-106, ground	3.735 x10^{-5}	kBq
Antimony-124, ground	3.88 x10^{-7}	kBq	Ethene, chloro-, ground	8.353 x10^{-13}	kg	Selenium, ground	1.161 x10^{-7}	kg
Antimony-125, ground	2.645 x10^{-7}	kBq	fluoranthene, ocean	1.373 x10^{-9}	kg	silver, ground	6.798 x10^{-10}	kg
AOX, , ground	8.670 x10^{-7}	kg	Fluoride, ground	1.744 x10^{-3}	kg	Silver-110, ground	5.676 x10^{-8}	kBq
arsenic, ground	1.386 x10^{-7}	kg	fluorine, ground	2.490 x10^{-8}	kg	Sodium, ground	2.503 x10^{-1}	kg
Arsenic, ion, ocean	5.82 x10^{-8}	kg	Heat, waste, ground	3.647	MJ	Sodium, ion, ocean	8.890 x10^{-6}	kg
Barium, ocean	4.753 x10^{-6}	kg	Hexane, ground	1.034 x10^{-12}	kg	Strontium, ground	1.547 x10^{-5}	kg
Benzene, ground	2.479 x10^{-7}	kg	Hydrocarbons, aromatic, ground	2.442 x10^{-7}	kg	Strontium-90, ground	1.807 x10^{-3}	kBq
Benzene, ethyl-, ocean	1.394 x10^{-7}	kg	Hydrocarbons, unspecified, ground	3.082 x10^{-7}	kg	Sulfate, ground	6.399 x10^{-3}	kg
benzo[a]anthracene, ground	1.434 x10^{-11}	kg	hydrogen chloride, ground	3.436 x10^{-10}	kg	Sulfide, ground	7.142 x10^{-6}	kg
benzo[k]fluoranthene, ocean	1.297 x10^{-9}	kg	hydrogen fluoride, ground	2.852 x10^{-10}	kg	Sulfite, ground	1.156 x10^{-6}	kg
Beryllium, ocean	6.785 x10^{-9}	kg	Hydrogen-3, Tritium, ground	5.518 x 10	kBq	Sulfur, ground	4.668 x10^{-8}	kg
BOD5, Biological Oxygen Demand, ground	1.927 x10^{-5}	kg	Hydroxide, ground	4.941 x10^{-9}	kg	SS, surface	-9.02 x10^{-3}	kg
Boron, ocean	4.987 x10^{-10}	kg	Hypochlorite, ocean	-2.19 x10^{-3}	kg	Thallium, ground	6.089 x10^{-11}	kg
Bromate, ground	7.799 x10^{-10}	kg	Iodine-129, ground	5.402 x10^{-3}	kBq	tin, ground	3.427 x10^{-11}	kg
Bromine, ground	2.231 x10^{-10}	kg	Iodine-131, ground	2.771 x10^{-7}	kBq	Tin, ion, surface	-6.66 x10^{-7}	kg
cadmium, ground	2.965 x10^{-7}	kg	Iron, ground	1.290 x10^{-3}	kg	titanium, ocean	9.579 x10^{-13}	kg

Parameter	Value	Unit	Parameter	Value	Unit	Parameter	Value	Unit
Cadmium, ion, surface	-1.72 x10^{-6}	kg	Iron, ion, ocean	1.038 x10^{-6}	kg	Titanium, ion, surface	6.923 x10^{-10}	kg
calcium, ocean	5.445 x10^{-8}	kg	Lead, ground	7.469 x10^{-7}	kg	TOC, ground	2.300 x10^{-5}	kg
Calcium, ion, ground	2.336 x10^{-2}	kg	Magnesium, ocean	1.224 x10^{-7}	kg	Toluene, ground	1.485 x10^{-7}	kg
Carbon-14, ground	1.891 x10^{-3}	kBq	Manganese, ground	5.000 x10^{-8}	kg	Uranium-238, ground	1.100 x10^{-2}	kBq
Carbonate, ground	4.281 x10^{-5}	kg	Mercury, ground	1.580 x10^{-8}	kg	vanadium, ocean	4.643 x10^{-8}	kg
Cesium-134, ground	1.939 x10^{-3}	kBq	Methanol, ground	1.222 x10^{-5}	kg	Vanadium, ion, ground	3.404 x10^{-8}	kg
Cesium-137, ground	1.754 x10^{-2}	kBq	Molybdenum, ground	6.498 x10^{-7}	kg	VOC, ground	5.624 x10^{-7}	kg
Chlorate, ground	7.317 x10^{-6}	kg	naphthalene, ocean	1.683 x10^{-7}	kg	Water, unspecified	-2.92 x 10	m3
Chloride, ground	4.596 x10^{-1}	kg	Nickel, ground	3.324 x10^{-7}	kg	Xylene, ground	1.559 x10^{-7}	kg
Chlorine, ground	2.950 x10^{-5}	kg	Nickel, ion, ocean	6.638 x10^{-8}	kg	zinc, ground	2.480 x10^{-7}	kg

Table D.7. LCI of landfilling of 1 tonne of waste
(Data extracted from DTU 2017)

Parameter	Value	Unit	Parameter	Value	Unit	Parameter	Value	Unit
Emission to air								
1,3,5-trimethylbenzene	9.358 x10^{-16}	kg	Ethane, 1,2-dichloro-	4.818 x10^{-7}	kg	Nitrogen	3.35 x10^{-5}	kg
1,4-Butanediol	3.992 x10^{-18}	kg	CFC-114	8.372 x10^{-9}	kg	Nitrogen monoxide	4.51 x10^{-14}	kg
1-Pentanol	8.338 x10^{-20}	kg	HFC-116	2. x10^{-19}	kg	Nitrogen oxides	1.905 x10^{-2}	kg
1-Pentene	6.30 x10^{-20}	kg	Ethanol	6.473 x10^{-9}	kg	NMVOC	4.822 x10^{-3}	kg
2-Aminopropanol	1.700 x10^{-20}	kg	Ethene	5.218 x10^{-7}	kg	Noble gases	4.649 x10^{-4}	kBq
2-Methyl-1-propanol	4.757 x10^{-19}	kg	Ethene, chloro-	3.799 x10^{-4}	kg	octane	3.509 x10^{-9}	kg
2-Methyl-2-butene	1.398 x10^{-23}	kg	Ethene, tetrachloro	1.232 x10^{-18}	kg	o-Nitrotoluene	2.619 x10^{-20}	kg
2-Nitrobenzoic acid	3.03 x10^{-20}	kg	Ethene, trichloro-	4.044 x10^{-4}	kg	oxygen	1.56 x10^{-5}	kg
2-Propanol	2.511 x10^{-14}	kg	Ethyl acetate	1.212 x10^{-13}	kg	Ozone	3.879 x10^{-15}	kg
Acenaphthene	1.105 x10^{-19}	kg	Ethyl cellulose	2.364 x10^{-16}	kg	PAH	1.927 x10^{-7}	kg
Acetaldehyde	1.787 x10^{-8}	kg	Ethylamine	1.489 x10^{-19}	kg	Palladium	4.641 x10^{-19}	kg
Acetic acid	2.361 x10^{-8}	kg	Ethylene diamine	2.004 x10^{-18}	kg	PM10	1.209 x10^{-4}	kg
Acetone	1.602 x10^{-8}	kg	Ethylene oxide	2.088 x10^{-14}	kg	P < 2.5 um	5.335 x10^{-6}	kg
Acetonitrile	1.414 x10^{-14}	kg	Ethyne	6.111 x10^{-14}	kg	P> 10 um	4.101 x10^{-11}	kg
Acid (as H+)	7.835 x10^{-13}	kg	fluoranthene	6.875 x10^{-13}	kg	P > 2.5 um, and < 10um	1.168 x10^{-3}	kg
Acrolein	1.069 x10^{-10}	kg	Fluorene	2.181 x10^{-12}	kg	Pentane	4.828 x10^{-6}	kg
Acrylic acid	6.515 x10^{-17}	kg	Fluoride	1.763 x10^{-9}	kg	Phenanthrene	6.963 x10^{-12}	kg
Actinides, radioactive	4.491 x10^{-13}	kBq	Fluorine	1.568 x10^{-9}	kg	Phenol	1.052 x10^{-4}	kg
Aerosols, radioactive	1.173 x10^{-11}	kBq	Fluosilicic acid	1.078 x10^{-14}	kg	Phenol, 2,4-dichloro	1.388 x10^{-18}	kg
Aldehydes	1.437 x10^{-18}	kg	Formaldehyde	7.360 x10^{-8}	kg	Phenol, pentachloro-	1.327 x10^{-14}	kg
Aluminium	1.311 x10^{-12}	kg	Formamide	1.525 x10^{-19}	kg	Phosphine	6.609 x10^{-15}	kg
Ammonia	1.134 x10^{-5}	kg	Formic acid	9.454 x10^{-14}	kg	Phosphorus	3.717 x10^{-14}	kg
ammonium	1.983 x10^{-14}	kg	Furan	3.012 x10^{-25}	kg	Platinum	1.829 x10^{-20}	kg
Ammonium carbonate	1.340 x10^{-15}	kg	Heat, waste	3.343 x10^{-1}	MJ	Plutonium-238	6.599 x10^{-18}	kBq
Aniline	9.049 x10^{-18}	kg	Helium	7.249 x10^{-10}	kg	Plutonium-alpha	9.681 x10^{-12}	kBq
anthracene	2.111 x10^{-13}	kg	Heptane	4.890 x10^{-7}	kg	Polonium-210	1.014 x10^{-11}	kBq
Anthranilic acid	2.220 x10^{-20}	kg	hexamethylene diamine	9.924 x10^{-16}	kg	Polychlorinated biphenyls	1.345 x10^{-5}	kg
Antimony	2. x10^{-10}	kg	Hexane	7.257 x10^{-7}	kg	Potassium	1.00 x10^{-11}	kg
Argon-41	2.907 x10^{-4}	kBq	Hydrocarbons,	2.749 x10^{-14}	kg	Potassium-40	1.610 x10^{-12}	kBq
Arsenic	7.752 x10^{-9}	kg	Hydrocarbons, aliphatic, alkanes	8.879 x10^{-12}	kg	Propanal	9.061 x10^{-16}	kg
Arsenic trioxide	7.94 x10^{-16}	kg	Hydrocarbons, aliphatic, unsaturated	5.498 x10^{-22}	kg	Propane	6.769 x10^{-5}	kg
Arsine	2.40 x10^{-12}	kg	Hydrocarbons, aromatic	5.054 x10^{-8}	kg	Propanol	5.228 x10^{-17}	kg
Barium	3.341 x10^{-7}	kg	Hydrocarbons, chlorinated	1.459 x10^{-15}	kg	Propene	2.888 x10^{-7}	kg
Benzal chloride	2.479 x10^{-23}	kg	hydrocyanic acid	8.628 x10^{-13}	kg	Propionic acid	2.861 x10^{-12}	kg
Benzaldehyde	8.761 x10^{-16}	kg	Hydrogen	5.42 x10^{-5}	kg	Propylamine	4.830 x10^{-20}	kg
Benzene	8.88 x10^{-4}	kg	hydrogen bromide	6.605 x10^{-12}	kg	Propylene oxide	3.801 x10^{-14}	kg
Benzene, chloro-	2.094 x10^{-4}	kg	Hydrogen chloride	2.318 x10^{-3}	kg	Protactinium-234	6.556 x10^{-12}	kBq
Benzene, dichloro	6.534 x10^{-19}	kg	Hydrogen fluoride	5.390 x10^{-4}	kg	Radioactive species, other beta emitters	1.245 x10^{-10}	kBq
Benzene, ethyl-	2.155 x10^{-3}	kg	Hydrogen iodide	7.149 x10^{-15}	kg	Radium-226	1.431 x10^{-12}	kBq
Benzene, hexachloro-	4.197 x10^{-18}	kg	Hydrogen peroxide	1.780 x10^{-16}	kg	Radium-228	7.751 x10^{-12}	kBq
Benzene, pentachloro-	1.053 x10^{-17}	kg	Hydrogen sulfide	4.551 x10^{-3}	kg	Radon-220	1.198 x10^{-13}	kBq
Benzo(a)pyrene	3.099 x10^{-10}	kg	Hydrogen-3, Tritium	5.665 x10^{-4}	kBq	Radon-222	7.158 x10^{-2}	kBq
benzo[a]anthracene	1.062 x10^{-13}	kg	indeno(1,2,3-cd)pyrene	7.051 x10^{-14}	kg	rhodium	4.475 x10^{-19}	kg
benzo[g,h,i]perylene	9.475 x10^{-14}	kg	Iodine	1.360 x10^{-15}	kg	Ruthenium-103	7.675 x10^{-18}	kBq
benzo[k]fluoranthene	1.895 x10^{-13}	kg	Iodine-135	1.199 x10^{-13}	kBq	Scandium	2.232 x10^{-14}	kg
Beryllium	4.966 x10^{-11}	kg	Iron	2.091 x10^{-8}	kg	Selenium	7.893 x10^{-9}	kg

Parameter	Value	Unit	Parameter	Value	Unit	Parameter	Value	Unit
Boron	4.323 x10^{-8}	kg	Isocyanic acid	2.275 x10^{-14}	kg	Silicon	2.534 x10^{-12}	kg
Boron trifluoride	1.039 x10^{-23}	kg	Isoprene	4.023 x10^{-24}	kg	Silicon tetrafluoride	2.090 x10^{-15}	kg
Bromine	2.414 x10^{-8}	kg	Isopropylamine	3.440 x10^{-20}	kg	Silver	5.273 x10^{-12}	kg
Butadiene	8.590 x10^{-14}	kg	Krypton-85	4.911 x 10	kBq	Silver-110	7.607 x10^{-17}	kBq
Butane	1.408 x10^{-5}	kg	Lactic acid	2.006 x 10^-18	kg	Sodium	3.211 x10^{-5}	kg
Butanol	3.884 x10^{-19}	kg	Lanthanum-140	3.162 x 10^-15	kBq	Strontium	8.787 x10^{-13}	kg
Butene	3.374 x10^{-11}	kg	Lead	2.918 x10^{-8}	kg	Styrene	1.481 x10^{-13}	kg
Butyrolactone	4.229 x10^{-19}	kg	lead dioxide	1.106 x10^{-15}	kg	Sulfate	3.586 x10^{-11}	kg
Cadmium	3.831x10^{-9}	kg	Lead-210	3.090 x10^{-19}	kBq	Sulfur dioxide	5.895 x10^{-3}	kg
Calcium	3.21x10^{-6}	kg	Magnesium	6.152 x10^{-13}	kg	Sulfur hexafluoride	6.721 x10^{-13}	kg
Carbon dioxide, fossil	-2.59 x10^{+2}	kg	Manganese	1.550 x10^{-9}	kg	Sulfuric acid	1.396 x10^{-16}	kg
Carbon dioxide, from soil or biomass stock	3.328x10^{-10}	kg	Manganese-54	2.943 x10^{-16}	kBq	Sulphur trioxide	9.747 x10^{-17}	kg
Carbon dioxide, non-fossil	5.334 x10	kg	Mercury	1.59 x10^{-7}	kg	tellurium	3.327 x10^{-14}	kg
Carbon disulfide	3.382 x10^{-11}	kg	Methane	3.024 x10^{-5}	kg	Terpenes	1.178 x10^{-14}	kg
Carbon monoxide, fossil	1.556 x10^{-3}	kg	Methane, bromo-, Halon 1001	5.671 x10^{-24}	kg	Thallium	6.233 x10^{-12}	kg
Carbon monoxide, non-fossil	1.293 x10^{-2}	kg	Methane, bromochlorodifluoro-, Halon 1211	1.371 x10^{-14}	kg	Thorium	2.265 x10^{-16}	kg
Carbon-14	1.334 x10^{-4}	kBq	Methane, bromotrifluoro-, Halon 1301	8.687 x10^{-13}	kg	Tin	2.766 x10^{-9}	kg
Chloramine	3.276 x10^{-19}	kg	Methane, chlorodifluoro-, HCFC-22	3.184 x10^{-4}	kg	tin oxide	9.607 x10^{-17}	kg
chloride	2.704 x10^{-9}	kg	Methane, chlorotrifluoro-, CFC-13	1.094 x10^{-10}	kg	Titanium	2.453 x10^{-12}	kg
Chlorine	1.246 x10^{-6}	kg	Methane, dichloro-, HCC-30	1.889 x10^{-3}	kg	Toluene	1.157x10^{-12}	kg
Chloroacetic acid	5.937 x10^{-17}	kg	Methane, dichlorodifluoro-, CFC-12	5.499 x10^{-4}	kg	Trimethylamine	1.25 x10^{-20}	kg
Chloroform	2.125 x10^{-17}	kg	Methane, dichlorofluoro-, HCFC-21	4.02 x10^{-20}	kg	Tungsten	1.95 x10^{-17}	kg
Chlorosilane, trimethyl-	5.404 x10^{-17}	kg	Methane, fossil	5.592 x10^{-3}	kg	Uranium	3.017 x10^{-16}	kg
Chlorosulfonic acid	1.970 x10^{-19}	kg	Methane, monochloro-, R-40	1.648 x10^{-18}	kg	used air	2.501 x10^{-2}	kg
Chromium	1.840 x10^{-8}	kg	Methane, non-fossil	4.712 x10^{-13}	kg	Vanadium	2.763 x10^{-7}	kg
Chrysene	2.609 x10^{-13}	kg	Methane, tetrachloro-, R-10	1.167 x10^{-14}	kg	VOC	1.435 x10^{-10}	kg
Cobalt	3.392 x10^{-9}	kg	Methane, tetrafluoro-, R-14	5.362 x10^{-12}	kg	Water	1.662 x10^{-2}	kg
Copper	3.479 x10^{-7}	kg	Methane, trichlorofluoro-, CFC-11	8.926 x10^{-5}	kg	Xenon-131m	4.012 x10^{-6}	kBq
Cumene	1.180 x10^{-18}	kg	Methane, trifluoro-, HFC-23	1.281 x10^{-17}	kg	Xylene	4.310 x10^{-3}	kg
Cyanide	8.132 x10^{-9}	kg	Methanesulfonic acid	1.630 x10^{-19}	kg	Zinc	2.180 x10^{-7}	kg
Cyanoacetic acid	1.613 x10^{-19}	kg	Methanol	5.953 x10^{-9}	kg	Zirconium	3.082 x10^{-15}	kg
Cyclohexane	1.336 x10^{-10}	kg	Methyl acetate	7.022 x10^{-21}	kg	Ethane	3.766 x10^{-5}	kg
Dibenz[a,h]anthracene	5.905 x10^{-14}	kg	Molybdenum	1.628 x10^{-9}	kg	Ethane, 1,1,1,2-tetrafluoro-, HFC-134a	1.415 x10^{-17}	kg
Diethylamine	4.303 x10^{-19}	kg	Monochloroethane	1.004 x10^{-3}	kg	Ethane, 1,1,1-trichloro-, HCFC-140	3.870 x10^{-24}	kg
Dimethyl malonate	2.023 x10^{-19}	kg	Monoethanolamine	8.389 x10^{-15}	kg	Nitrate	3.469 x10^{-15}	kg
Dinitrogen monoxide	2.761 x10^{-5}	kg	m-Xylene	5.072 x10^{-14}	kg	Nitrobenzene	1.21 x10^{-17}	kg
Dioxins, measured as 2,3,7,8-tetrachlorodibenzo-p-dioxin	1.194 x10^{-11}	kg	Naphthalene	2.21 x10^{-11}	kg			
Dipropylamine	2.560 x10^{-18}	kg	Nickel	1.361 x10^{-7}	kg			

Parameter	Value	Unit	Parameter	Value	Unit	Parameter	Value	Unit
Emission to water								
1,2-dichloropropane, ground	3.082 x10^{-14}	kg	copper, ocean	7.111 x10^{-10}	kg	Oils, unspecified, ocean	1.573 x10^{-8}	kg
1,4-Butanediol, surface	7.959 x10^{-19}	kg	Copper, ion, ground	5.232 x10^{-8}	kg	o-Xylene, unspecified	7.744 x10^{-19}	kg
1-Pentanol, surface	1.597 x10^{-18}	kg	cresol, ground	2.596 x10^{-13}	kg	PAH, ocean	4.249 x10^{-12}	kg
1-Pentene, surface	2.001 x10^{-19}	kg	Cumene, surface	3.826 x10^{-12}	kg	P > PM10, ocean	1.797 x10^{-6}	kg
2,3,7,8-tetrachlorodibenzo-p-dioxin, ground	1.512 x10^{-19}	kg	Curium alpha, ground	1.741 x10^{-7}	kBq	PM10, ground	2.037 x10^{-6}	kg
2-Aminopropanol, surface	2.913 x10^{-10}	kg	Cyanide, ground	2.449 x10^{-9}	kg	Phenol, ocean	9.572 x10^{-7}	kg
2-Methyl-1-propanol, surface	4.262 x10^{-20}	kg	decane, ocean	5.374 x10^{-8}	kg	Phosphate, ground	2.800 x10^{-6}	kg
2-Methyl-2-butene, surface	1.142 x10^{-7}	kg	Dichromate, surface	2.372 x10^{-14}	kg	Phosphorus, ground	2.275 x10^{-15}	kg
2-Propanol, surface	3.354 x10^{-23}	kg	Diethylamine, surface	9.654 x10^{-18}	kg	Plutonium-alpha, ground	5.233 x10^{-7}	kBq
4-Methyl-2-pentanone, unspecified	1.904 x10^{-19}	kg	Dimethylamine, surface	5.217 x10^{-18}	kg	Polonium-210, ground	1.016 x10^{-12}	kBq
Acenaphthene, ocean	1.471 x10^{-19}	kg	Dipropylamine, surface	6.145 x10^{-18}	kg	potassium, ground	3.256 x10^{-7}	kg
Acenaphthylene, ocean	2.248 x10^{-9}	kg	Dissolved solids, ground	7.850 x10^{-11}	kg	Potassium, ion, ground	1.296 x10^{-10}	kg
Acetaldehyde, surface	8.564 x10^{-10}	kg	DOC, Dissolved Organic Carbon, ground, long-term	1.729 x10^{-9}	kg	Potassium-40, ground	8.069 x10^{-14}	kBq
Acetic acid, ocean	9.735 x10^{-16}	kg	Ethane, 1,2-dichloro-, ground	2.019 x10^{-8}	kg	Propanal, surface	2.897 x10^{-19}	kg
Acetone, surface	7.077 x10^{-9}	kg	Ethanol, surface	1.108 x10^{-15}	kg	Propanol, surface	4.613 x10^{-19}	kg
Acetonitrile, surface	1.346 x10^{-17}	kg	Ethene, surface	1.601 x10^{-12}	kg	Propene, surface	1.505 x10^{-12}	kg
Acetyl chloride, surface	1.351 x10^{-19}	kg	Ethene, chloro-, ground	9.526 x10^{-8}	kg	Propionic acid, surface	3.412 x10^{-18}	kg
Acidity, unspecified, ground	1.572 x10^{-19}	kg	Ethyl acetate, surface	1.035 x10^{-17}	kg	Propylamine, surface	1.159 x10^{-19}	kg
Acidity, unspecified, ground	5.882 x10^{-7}	kg	Ethylamine, surface	3.575 x10^{-19}	kg	Propylene oxide, surface	9.146 x10^{-14}	kg
Acrylate, ion, surface	1.542 x10^{-16}	kg	Ethylene diamine, surface	4.824 x10^{-18}	kg	Protactinium-234, surface	1.209 x10^{-10}	kBq
Acrylonitrile, ground	2.965 x10^{-13}	kg	Ethylene oxide, surface	8.947 x10^{-17}	kg	R-40, ground	7.278 x10^{-12}	kg
Actinides, radioactive, unspecified, ocean	7.858 x10^{-11}	kBq	fluoranthene, ocean	7.265 x10^{-12}	kg	Radioactive species, alpha emitters, surface	2.358 x10^{-12}	kBq
Aluminium, ocean	1.015 x10^{-10}	kg	Fluoride, ground	2.146 x10^{-6}	kg	Radioactive species, Nuclides, ocean	4.698 x10^{-8}	kBq
Americium-241, ground	1.314 x10^{-7}	kBq	fluorine, ground	1.214 x10^{-11}	kg	Radium-224, ocean	2.713 x10^{-8}	kBq
ammonia, ocean	2.650 x10^{-11}	kg	Fluosilicic acid, surface	1.940 x10^{-14}	kg	Radium-226, ground	2.166 x10^{-3}	kBq
Ammonium, ion, ground	1.171 x10^{-6}	kg	Formaldehyde, surface	4.728 x10^{-16}	kg	Radium-228, ocean	5.426 x10^{-8}	kBq
Aniline, surface	2.173 x10^{-17}	kg	Formamide, surface	3.660 x10^{-19}	kg	Rubidium, ocean	5.426 x10^{-12}	kg
anthracene, ocean	7.713 x10^{-12}	kg	Formate, surface	4.287 x10^{-17}	kg	Ruthenium-103, surface	7.457 x10^{-15}	kBq
Antimony, ground	2.133 x10^{-13}	kg	Formic acid, surface	1.062 x10^{-19}	kg	Ruthenium-106, ground	1.314 x10^{-7}	kBq
AOX, ocean	2.490 x10^{-13}	kg	Glutaraldehyde, ocean	7.273 x10^{-13}	kg	Scandium, ground	7.655 x10^{-14}	kg
arsenic, ocean	4.350 x10^{-10}	kg	Heat, waste, ground	1.481 x10^{-2}	MJ	Selenium, ground	2.802 x10^{-9}	kg
Arsenic, ion, ocean	2.354 x10^{-10}	kg	Hexane, ocean	4.887 x10^{-14}	kg	Silicon, ground	1.039 x10^{-10}	kg
Barite, ocean	5.891 x10^{-9}	kg	Hydrocarbons, aliphatic, alkanes, ocean	7.053 x10^{-11}	kg	silver, ground	9.463 x10^{-13}	kg
Barium, ocean	2.176 x10^{-6}	kg	Hydrocarbons, aliphatic, unsaturated, ocean	6.511 x10^{-12}	kg	Silver, ion, ground	4.266 x10^{-15}	kg

Parameter	Value	Unit	Parameter	Value	Unit	Parameter	Value	Unit
Barium-140, surface	9.624 x10^{-14}	kBq	Hydrocarbons, aromatic, ground	2.130 x10^{-9}	kg	Sodium, ground	5.250 x10^{-4}	kg
Benzene, ground	7.924 x10^{-8}	kg	Hydrocarbons, unspecified, ground	1.032 x10^{-6}	kg	Sodium formate, surface	1.156 x10^{-16}	kg
Benzene, chloro-, surface	4.103 x10^{-15}	kg	hydrogen chloride, ground	2.642 x10^{-13}	kg	Sodium, ion, ocean	2.112 x10^{-7}	kg
Benzene, ethyl-, ocean	5.309 x10^{-8}	kg	hydrogen fluoride, ground	1.006 x10^{-12}	kg	Sodium-24, surface	2.674 x10^{-13}	kBq
benzo[a]anthracene, ground	7.932 x10^{-14}	kg	Hydrogen peroxide, surface	3.349 x10^{-15}	kg	Solids, inorganic, ground	2.357 x10^{-9}	kg
benzo[k]fluoranthene, ground	2.339 x10^{-14}	kg	Hydrogen sulfide, ground, long-term	4.943 x10^{-12}	kg	Strontium, ground	1.016 x10^{-6}	kg
Beryllium, ocean	3.145 x10^{-9}	kg	Hydrogen-3, Tritium, ground	1.941 x10^{-1}	kBq	Sulfate, ground	3.442 x10^{-4}	kg
BOD5, ground	6.873 x10^{-6}	kg	Hydroxide, ground	1.624 x10^{-10}	kg	Sulfide, ground	2.577 x10^{-6}	kg
Borate, surface	1.690 x10^{-17}	kg	Hypochlorite, ocean	3.961 x10^{-5}	kg	Sulfite, ground	1.157 x10^{-9}	kg
Boron, ground	3.839 x10^{-9}	kg	Iodide, ground	6.010 x10^{-14}	kg	Sulfur, ocean	2.490 x10^{-11}	kg
Bromate, ground	4.265 x10^{-9}	kg	iron, ocean	4.240 x10^{-9}	kg	Suspended solids, unspecified, ocean	2.133 x10^{-8}	kg
Bromide, surface	6.033 x10^{-15}	kg	Iron, ion, ocean	4.132 x10^{-9}	kg	t-Butyl methyl ether, ocean	3.547 x10^{-12}	kg
Bromine, ground	4.212 x10^{-11}	kg	Iron-59, surface	1.661 x10^{-14}	kBq	t-Butylamine, surface	3.338 x10^{-19}	kg
Butanol, surface	4.407 x10^{-16}	kg	Isopropylamine, surface	8.255 x10^{-20}	kg	Technetium-99m, surface	8.197 x10^{-13}	kBq
Butene, surface	4.219 x10^{-16}	kg	Lactic acid, surface	4.814 x10^{-18}	kg	Tellurium-123m, surface	1.387 x10^{-12}	kBq
Butyl acetate, surface	5.717 x10^{-16}	kg	Lanthanum-140, surface	1.025 x10^{-13}	kBq	Tellurium-132, surface	2.046 x10^{-15}	kBq
Butyrolactone, surface	1.015 x10^{-18}	kg	Lead, ground	1.905 x10^{-8}	kg	Thallium, ground	1.015 x10^{-12}	kg
cadmium, ocean	8.899 x10^{-10}	kg	Lithium, ion, surface	6.982 x10^{-18}	kg	Thorium-228, ground	8.185 x10^{-15}	kBq
Cadmium, ion, ground	4.968 x10^{-12}	kg	Magnesium, ground	1.854 x10^{-10}	kg	Thorium-230, surface	1.650 x10^{-8}	kBq
calcium, ocean	1.575 x10^{-9}	kg	Manganese, ocean	7.080 x10^{-8}	kg	Tin, ocean	2.719 x10^{-13}	kg
Calcium, ion, ground	4.375 x10^{-6}	kg	Manganese-54, ground	4.452 x10^{-6}	kBq	Tin, ion, ground	1.330 x10^{-15}	kg
Carbon disulfide, surface	8.674 x10^{-18}	kg	Mercury, ground	5.503 x10^{-10}	kg	titanium, ocean	2.770 x10^{-14}	kg
Carbon-14, ground	6.654 x10^{-6}	kBq	Methane, dichloro-, HCC-30, surface	1.898 x10^{-11}	kg	Titanium, ion, ground	5.980 x10^{-14}	kg
Carbonate, ground	2.790 x10^{-5}	kg	Methanol, ground	8.032 x10^{-8}	kg	TOC, ground	6.784 x10^{-6}	kg
Carboxylic acids, unspecified, ocean	3.055 x10^{-9}	kg	Methyl acetate, surface	1.685 x10^{-20}	kg	Toluene, ocean	2.385 x10^{-7}	kg
Cerium-141, surface	3.848 x10^{-14}	kBq	Methyl acrylate, surface	1.444 x10^{-15}	kg	Toluene, 2-chloro, surface	7.512 x10^{-18}	kg
Cerium-144, surface	1.171 x10^{-14}	kBq	Methyl amine, surface	3.877 x10^{-18}	kg	Tributyltin compounds, ocean	1.410 x10^{-12}	kg
Cesium, ocean	5.426 x10^{-13}	kg	Methyl formate, surface	1.302 x10^{-19}	kg	Triethylene glycol, ocean	1.071 x10^{-13}	kg
Chloramine, surface	2.940 x10^{-18}	kg	Molybdenum, ocean	2.833 x10^{-13}	kg	Trimethylamine, surface	3.020 x10^{-20}	kg
Chlorate, ground	2.424 x10^{-6}	kg	Molybdenum-99, surface	3.534 x10^{-14}	kBq	Tungsten, ground	1.281 x10^{-13}	kg
Chloride, ground	3.115 x10^{-3}	kg	m-Xylene, surface	3.875 x10^{-19}	kg	Uranium alpha, surface	6.965 x10^{-9}	kBq
Chlorinated solvents, ocean	7.590 x10^{-22}	kg	naphthalene, ground	4.189 x10^{-11}	kg	Uranium-234, surface	1.451 x10^{-10}	kBq
Chlorine, ground	1.368 x10^{-7}	kg	nickel, ocean	3.936 x10^{-10}	kg	Uranium-235, surface	2.394 x10^{-10}	kBq
Chloroacetic acid, surface	2.844 x10^{-15}	kg	Nickel, ion, ocean	2.684 x10^{-10}	kg	Uranium-238, ground	3.804 x10^{-5}	kBq
Chloroacetyl chloride, surface	5.685 x10^{-20}	kg	Niobium-95, surface	9.585 x10^{-13}	kBq	Urea, surface	3.489 x10^{-19}	kg
Chloroform, surface	8.747 x10^{-18}	kg	Nitrate, ground	7.694 x10^{-7}	kg	vanadium, ocean	2.357 x10^{-10}	kg
Chlorosulfonic acid, surface	4.913 x10^{-19}	kg	Nitrite, ground, long-term	1.734 x10^{-13}	kg	Vanadium, ion, ground	2.230 x10^{-10}	kg
Chromium, ocean	1.426 x10^{-9}	kg	Nitrobenzene, surface	4.858 x10^{-17}	kg	VOC, ocean	1.243 x10^{-9}	kg
chrysene, ground	3.156 x10^{-13}	kg	Nitrogen, ground	3.007 x10^{-6}	kg	Water, surface	4.130 x10^{-3}	kg

Parameter	Value	Unit	Parameter	Value	Unit	Parameter	Value	Unit
Cobalt, ocean	5.504 x10^{-8}	kg	Nitrogen, organic bound, ground, long-term	5.217 x10^{-12}	kg	zinc, ground	1.660 x10^{-10}	kg
COD, ocean	1.161 x10^{-5}	kg	o-Dichlorobenzene, surface	2.001 x10^{-13}	kg	Zinc, ion, ocean	1.387 x10^{-8}	kg

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