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SCHOOL OF CHEMICAL AND ENVIRONMENTAL ENGINEERING

Laboratory of Toxic and Hazardous Waste Management

Ph.D. DISSERTATION

«Modelling solid waste management systems, using advanced life cycle assessment (LCA) tools»

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ΠΟΛΥΤΕΧΝΕΙΟ ΚΡΗΤΗΣ

ΣΧΟΛΗ ΧΗΜΙΚΩΝ ΜΗΧΑΝΙΚΩΝ ΚΑΙ ΜΗΧΑΝΙΚΩΝ ΠΕΡΙΒΑΛΛΟΝΤΟΣ

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«Μοντελοποίηση συστημάτων διαχείρισης αστικών στερεών απορριμμάτων, χρησιμοποιώντας εξειδικευμένα εργαλεία ανάλυσης κύκλου ζωής (ΑΚΖ)»

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Dedicated to my inspiration: Anna…

…to inspire Giannis and Marios

Desire is the key to motivation, but it's determination and commitment to an unrelenting pursuit of your goal - a commitment to excellence - that will enable you to attain the success you seek.

Mario Andretti

ΠΕΡΙΛΗΨΗ

Τα προβλήματα από τις περιβαλλοντικές επιπτώσεις που συνδέονται με τις ανθρωπογενείς δραστηριότητες, τις τελευταίες δεκαετίες έχουν ενταθεί. Οι κυβερνήσεις και ειδικότερα η Ευρωπαϊκή Ένωση έχουν στραφεί στη δημιουργία, καθιέρωση και εφαρμογή νομικών πλαισίων, οδηγιών και κατευθυντήριων γραμμών, στοχεύοντας βραχυπρόθεσμα στην ελαχιστοποίηση και μακροπρόθεσμα την αντιστροφή των αρνητικών περιβαλλοντικών αυτών επιπτώσεων. Όσον αφορά στη διαχείριση των αστικών στερεών απορριμμάτων, η εφαρμογή της Οδηγίας 851/2018(ΕΕ) θεσπίζει μέτρα για την προστασία του περιβάλλοντος και της ανθρώπινης υγείας, προλαμβάνοντας ή μειώνοντας την παραγωγή απορριμμάτων καθώς και τις αρνητικές συνέπειες της παραγωγής και διαχείρισης αυτών, περιορίζοντας έτσι τον συνολικό αντίκτυπο της χρήσης των πόρων στην διαχείριση των αστικών στερεών απορριμμάτων και βελτιώνοντας την αποδοτικότητά της. Ταυτόχρονα, ο νέος κλιματικός νόμος 4936/2022 που θεσπίζει μέτρα και πολιτικές για την προσαρμογή της χώρας στην κλιματική αλλαγή και τη διασφάλιση της πορείας ανθρακοποίησης έως το έτος 2050, παίζει καθοριστική σημασία για τη μετάβαση σε μια κυκλική οικονομία και την εξασφάλιση της μακροπρόθεσμης ανταγωνιστικότητας της Ευρωπαϊκής Ένωσης. Επιτάσσεται λοιπόν η διερεύνηση νέων σεναρίων διαχείρισης των παραγόμενων απορριμμάτων, αλλά και ο εκσυγχρονισμός των ήδη υπαρχόντων, ώστε να εναρμονίζονται με τις νέες κατευθυντήριες γραμμές. Ειδικότερα, οι στόχοι για τη μείωση των βιοαποβλήτων που καταλήγουν σε χώρους υγειονομικής ταφής και την ελάττωση των καταναλωμένων φυσικών πόρων, οδηγούν στην αναθεώρηση και τον επανασχεδιασμό των υφιστάμενων συστημάτων διαχείρισης απορριμμάτων. Τίθεται όμως το ερώτημα αν και κατά πόσο είναι περιβαλλοντικά ωφέλιμος ο εκσυγχρονισμός των συστημάτων διαχείρισης, καθώς και ποια θα είναι τα εργαλεία που θα μπορέσουν να αξιολογήσουν αυτή τη μετάβαση.

Η Ανάλυση Κύκλου Ζωής (ΑΚΖ) (Life Cycle Assessment - LCA) στη διαχείριση των στερεών απορριμμάτων αποτελεί πολύτιμο εργαλείο, τόσο για την κατανόηση των περιβαλλοντικών επιπτώσεων, όσο και για την εις βάθος εξέταση των διεργασιών που διέπουν ένα τέτοιο σύστημα. Με τη χρήση της παραπάνω μεθοδολογίας, μπορούν να εξεταστούν, να αναλυθούν και να μοντελοποιηθούν από

απλά έως εξαιρετικά πολύπλοκα διαχειριστικά σχέδια και διαδικασίες. Καθώς όμως τα μοντέλα που προκύπτουν βασίζονται στην επεξεργασία μεγάλου όγκου δεδομένων και την επίλυση πολύπλοκων υπολογισμών, είναι εμφανής η ανάγκη για υπολογιστική δύναμη, αρκετές παραδοχές και έξυπνα ή ευέλικτα υπολογιστικά και απεικονιστικά εργαλεία. Είναι επομένως εμφανής η ανάγκη και η χρησιμότητα εξειδικευμένων υπολογιστικών ΑΚΖ εργαλείων ώστε να επιτευχθούν οι στόχοι κάθε μελέτης, καθώς και δεδομένων με αντιπροσωπευτικά στοιχεία που να ανταποκρίνονται χωροχρονικά στα ιδιαίτερα χαρακτηριστικά του προς μελέτη συστήματος.

Το σύστημα διαχείρισης των στερεών αστικών απορριμμάτων της Περιφερειακής Ενότητας Χανίων στην Κρήτη, αποτελεί τυπικό παράδειγμα ολοκληρωμένης διαχείρισης αστικών στερεών απορριμμάτων, αντιπροσωπευτικό του συνόλου των συστημάτων που έχουν αρχίσει να εφαρμόζονται στην περιοχή της Μεσογείου. Το συγκεκριμένο σύστημα τέθηκε υπό συστηματική παρακολούθηση και μελέτη για τρία χρόνια με δειγματοληψίες και συλλογή δεδομένων. Οι πληροφορίες που συλλέχθηκαν επέτρεψαν, με τη χρήση εξειδικευμένων υπολογιστικών εργαλείων ΑΚΖ, την εις βάθος ανάλυση των κρίσιμων διαδικασιών που διέπουν τα στάδια της διαχείρισης των στερεών αστικών απορριμμάτων, από την συλλογή, τη μεταφορά, αλλά και όλα τα στάδια της επεξεργασίας τους όπως ο διαχωρισμός, η ανάκτηση και η κομποστοποίηση των υλικών αυτών. Τα δεδομένα που προέκυψαν, επέτρεψαν την κατανόηση των διεργασιών που λαμβάνουν χώρα, καθώς και να την σύνδεσή τους με τις περιβαλλοντικές επιπτώσεις του υπό μελέτη συστήματος σε τομείς, όπως η ανθρώπινη υγεία, η ποιότητα του αέρα και των υδατικών οικοσυστημάτων ή της κλιματικής αλλαγής. Με βάση τα δεδομένα αυτά η παρούσα διατριβή επικεντρώθηκε στη διερεύνηση της συλλογής - μεταφοράς των ΑΣΑ, καθώς και στην ολοκληρωμένη ανάκτηση και επεξεργασία των ανακυκλώσιμων υλικών και των βιοαποβλήτων.

Πιο συγκεκριμένα, η παρούσα διατριβή στοχεύει στην αξιολόγηση των περιβαλλοντικών επιπτώσεων που συνδέονται με τις πρακτικές συλλογής των αστικών στερεών απορριμμάτων στην Περιφερειακή Ενότητα Χανίων, συμβατικά αλλά και με την εισαγωγή σταθμών μεταφόρτωσης. Αξιοποιώντας πραγματικά, δεδομένα ευρέως διαθέσιμα στους διαχειριστές συστημάτων, όπως ζυγολόγια, λίστες δρομολογίων, αλλά και χιλιομετρικά δεδομένα ή στοιχεία κατανάλωσης καυσίμων από τον στόλο οχημάτων συλλογής, τα οποία είναι εύκολα διαθέσιμα, η παρούσα διατριβή, αποσκοπεί στη δημιουργία προσεγγιστικών αποθετηρίων ανάλυσης κύκλου ζωής σχετικά με τη διαδικασία συλλογής και μεταφοράς απορριμμάτων. Στη συνέχεια, τα δεδομένα αυτά χρησιμοποιούνται για τον σχεδιασμό και τη μελέτη σεναρίων συλλογής και μεταφοράς σύμμεικτων και ανακυκλώσιμων υλικών στη μελετώμενη περιφέρεια και συγκρίνονται ως προς την επίδραση της χωροθέτησης σταθμών μεταφόρτωσης απορριμμάτων στο περιβαλλοντικό αποτύπωμα της διαδικασίας. Τα αποτελέσματα έδειξαν ότι ο συνδυασμός απευθείας μεταφοράς και εισαγωγής σταθμών μεταφόρτωσης απορριμμάτων, υπό συγκεκριμένες προϋποθέσεις, μπορεί να έχει ευνοϊκά αποτελέσματα για το περιβάλλον, εάν συνδυάζεται με ένα ολοκληρωμένο σύστημα διαχείρισης απορριμμάτων με εκτεταμένη ανάκτηση ανακυκλώσιμων υλικών.

Στο κομμάτι της επεξεργασίας των συλλεγόμενων υλικών, η διατριβή επικεντρώθηκε στη δημιουργία απογραφών για όλες τις διεργασίες επεξεργασίας και ανάκτησης των δυο κυριότερων επεξεργαζόμενων ρευμάτων, δηλαδή σύμμεικτα και ανακυκλώσιμα, ενώ ιδιαίτερη βάση δόθηκε στη διεργασία της μηχανικής κομποστοποίησης. Η συγκεκριμένη διεργασία, αποτελεί μια διαδεδομένη μεθοδολογία επεξεργασίας του μηχανικά συλλεγόμενου οργανικού κλάσματος των στερεών αστικών απορριμμάτων, ώστε τα υλικά αυτά να σταθεροποιηθούν και να μειωθούν οι περιβαλλοντικές τους επιπτώσεις., Στην παρούσα διατριβή, η διαδικασία μοντελοποιήθηκε και δημιουργήθηκε ένας κατάλογος (inventory) των διεργασιών που συνθέτουν τη μελετώμενη επεξεργασία βασιζόμενη στη μονάδα μηχανικής κομποστοποίησης της περιφερειακής ενότητας Χανίων. Στόχος ήταν να καταγραφούν και να μοντελοποιηθούν όλες οι εισροές και εκροές του συστήματος, οι διεργασίες, αλλά και οι μεταβολές των υλικών. Για τον λόγο αυτό υπολογίστηκαν οι λόγοι ανάκτησης των υλικών που συνθέτουν το εισερχόμενο οργανικό κλάσμα, αλλά και οι λόγοι αποδόμησής τους κατά τη διεργασία της κομποστοποίησης.

Με εργαλείο τη λεπτομερή αποτύπωση της μηχανικής κομποστοποίησης και τα στοιχεία από τις απογραφές για κάθε επιμέρους διεργασία ανάκτησης στο μελετώμενο σύστημα, η διατριβή επικεντρώθηκε επίσης στη διερεύνηση της ανάκτησης και αξιοποίησης των βιοαποβλήτων, διεξάγοντας μια συγκριτική μελέτη ανάλυσης κύκλου ζωής ανάμεσα σε δυο σενάρια διαχείρισης βιοαποβλήτων. Τα αποτελέσματα της έρευνας δείχνουν ότι η μετάβαση από το υπάρχον σύστημα μηχανικής ανάκτησης και κομποστοποίησης βιοαποβλήτων από σύμμεικτα απόβλητα σε ένα σύστημα διαλογής στην πηγή και κομποστοποίησης, μπορεί να ελαχιστοποιήσει τις περιβαλλοντικές επιπτώσεις στην ανθρώπινη υγεία κατά 4,6 φορές, όσον αφορά την ποιότητα των υδατικών οικοσυστημάτων, κατά 6,3 φορές τη διατήρηση των φυσικών πόρων και κατά 2,5 φορές όταν η επεξεργασία των βιοαποβλήτων συνδυάζεται με την εκτεταμένη ανάκτηση ανακυκλώσιμων υλικών και τη χρήση του παραγόμενου κόμποστ για εδαφοβελτίωση και υποκατάσταση λιπασμάτων.

EXECUTIVE SUMMARY

The problems arising from environmental impacts associated with human activities have intensified in recent decades. Governments, particularly the European Union, have focused on creating, establishing, and implementing legal frameworks, directives, and guidelines. Their short-term goal is the minimization and, in the long term, the reversal of these negative environmental impacts. Directive 851/2018 (EU) introduces environmental protection and human health measures to manage urban solid waste. These measures aim to prevent or reduce waste production, mitigate the negative consequences of production and waste management, limit the overall impact of resource use, and enhance the waste management system efficiency. Simultaneously, the new climate law, Law No. 4936/2022, establishes measures and policies for the country's adaptation to climate change and ensures decarbonization by 2050. These initiatives play a crucial role in transitioning to a circular economy and ensuring the long-term competitiveness of the European Union. Therefore, exploring new waste management scenarios and modernizing existing systems must align with the new guidelines. Specifically, goals for reducing biowaste ending up in landfills and decreasing the consumption of natural resources necessitate revising and redesigning existing waste management systems. However, as the systems undergo modernization, the question arises about the environmental benefits of these upgrades and the tools that can evaluate this transition.

The Life Cycle Assessment (LCA) in managing solid waste represents a precious tool for understanding environmental impacts and the in-depth exploration of the processes that regulate such a system. Through this methodology, management plans ranging from simple to highly complex can be examined and analyzed. Indeed, as the boundaries of the studied system expand and the system becomes more complex, taking into account more information and data, the need for computational power increases correspondingly. Therefore, using specialized computational tools becomes essential for achieving the goals of each research study.

In this thesis, the integrated waste management system of the Chania region in Crete, which represents a typical Mediterranean integrated waste management system (IWMS), underwent a comprehensive study and modelling through extensive waste

sampling and data collection. Such as weighting, truck travel logs, and fuel consumption info from the waste collection fleet were easily accessible to waste managers. The resulting data enabled an in-depth analysis of the critical processes involved in collecting, transporting, and treating municipal solid waste. This valuable information was utilized to enhance the understanding of the system's dynamics and environmental impacts.

Additionally, the present thesis evaluated the environmental impact associated with existing waste collection practices in the Chania region of Greece. Herein, the introduction of waste transfer stations was considered in the context of resource consumption. The study leveraged actual, readily accessible data, such as weight records, total monthly fuel consumed, and total distance travelled by the collection vehicles, to create and evaluate a life cycle assessment inventory. Advanced LCA tools software was used to compare the implications of varying waste transfer station locations and quantities in a modern integrated solid waste management system. Using the produced data, five scenarios – one conventional direct haul and four scenarios including waste transfer stations - were explored, and their environmental impacts and efficiencies within the context of integrated waste management were assessed. The aim was to provide a comprehensive analysis that can inform better waste management practices, balancing operational efficiency, resource consumption, and environmental impact. The final results showed that a combination of direct and assigned to waste transfer stations (WTS) transport is the optimal scenario for the region but also revealed the benefits arising from proper and methodical transportation programming.

Moreover, mechanical composting is a popular treatment method for the mechanically separated organic fraction of municipal solid wastes (MSW) to stabilize the waste material and reduce its environmental impacts. The model and life cycle inventory database were created based on the existing centralized mechanical composting facility in Chania (Crete, Greece). All stages of the composting process, wherein input-output flows were comprehensively analyzed based on specific waste fragments. The transfer coefficients were calculated for each waste fragment throughout the processes. The degradation rate was measured as kg of C and N released per mg of the treated material. The results show that process degradation rates are independent of the initial fragmental composition. This is the first study that accurately models the fate of specific waste fragments in a composting plant. At the same time, the developed life cycle inventory (concerning mass and energy balances) can be applied to estimate the environmental impacts regarding mechanical composting of the organic fraction of municipal solid wastes.

Lastly, an LCA was performed to investigate the environmental impacts of two alternative approaches in a biowaste management system. The system inventory was based on actual data and on-site sampling for two consecutive years at the mechanical and biological treatment (MBT) facility in the prefecture of Chania (Greece). The facility pertains to MBT for household waste and material recycling facility (MRF) for the recyclable fractions in two different process lines. The mass balances and environmental performance were assessed from waste generation to end-use. The LCA and ReCiPe 2016 methodology allowed for estimating the endpoint environmental impacts on human health, ecosystem quality and resource scarcity. The results show that biowaste source segregation in an integrated waste management system significantly benefits its recoverability potential and environmental performance. Impacts on human health (HH) have been reduced by 4.6 times, on freshwater ecosystem quality (EQf) by 6.3 times, and resource scarcity (RS) usage by 2.5 times when biowaste is combined with compost production and use, material recovery and reprocessing for fertilizer and raw material substitution.

Abbreviations

Nomenclature	
Al	Aluminium materials
CLO	Compost-like output
DALYs	Disability-adjusted life years
EOFP	Photochemical oxidant formation potential: ecosystem.
EQF	Ecosystem quality for freshwater systems
EQT	Ecosystem quality for terrestrial systems
EU	European Union
FEP	Freshwater eutrophication potential
FETP	Freshwater eco-toxicity potential
FFP	Fossil fuel potential
GWP	Global warming potential
HH	Human health
HHW	Household waste
HOFP	Photochemical oxidant formation potential: human health
HTPc	Human toxicity potential: cancer
HTPnc	Human toxicity potential: non-cancer
IRP	Ionizing radiation potential
IWMS	Integrated waste management system
LCA	Life cycle assessment
LCIA	Life cycle impact assessment
LOP	Agricultural land occupation potential
MBT	Mechanical and biological treatment
METP	Marine eco-toxicity potential
MFA	Material flow analysis
MPs	Micro plastics
MRF	Material recovery facility
MR	Material recycling
MSBW	Municipal solid biowaste
MS-OFMSW	Mechanically sorted organic fraction of municipal solid waste
MSW	Municipal solid waste
MW	Mix waste
NIR	Near-infrared
ODP	Ozone depletion potential
OFMSW	Organic fragment of municipal solid waste
PE	Polyethylene
PET	Polyethylene terephthalate
PMFP	Fine particulate matter formation potential
PP	Polypropylene
RF	Recyclable Fractions
RS	Resource scarcity
SOP	Surplus ore potential
TAP	Terrestrial acidification potential
TETP	Terrestrial eco-toxicity potential
WCP	Water consumption potential
WTS	Waste transfer station

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1. Chapter

Introduction

Municipal solid waste and biowaste generation have significantly increased over the last decades. In 2020, the EU generated 225 million tonnes of municipal solid waste (Eurostat, 2021), of which about 34% (76.5 million tonnes) is estimated to be biowaste (Carabassa et al., 2020; van der Linden and Reichel, 2020). A significant portion of biowaste is mixed with other household waste and disposed of in landfills. Various waste management schemes have been implemented throughout these years, focusing on waste collection, materials recovery, and treatment. The improper management of MSW, including open burning, open dumping, and unsanitary landfilling, contributes to numerous environmental issues such as global warming, ozone depletion, human health hazards, ecosystem damage, and abiotic resource depletion (Laurent et al., 2014). The general linear economy approach of producing, consuming, and disposing waste has affected our ecosystem and natural resources (Aryan et al., 2023). Effective decision-making in the MSWM (Municipal Solid Waste Management) industry necessitates a comprehensive assessment to minimise the hazards associated with these impacts. The collection and disposal of MSW pose significant challenges in many countries worldwide since the collection has a significant share of environmental impacts (Abdel-Shafy and Mansour, 2018). Consequently, there is immense pressure on our already strained waste management systems.

An integrated waste management system (IWMS) covers all the aspects of waste life from its generation till its final disposal, "Cradle-to-Grave", with collection and treatment to encompass two of the primary waste processes of their life cycle. Waste collection and road transport contribute significantly to global greenhouse gas emissions, accounting for approximately a quarter (24%) of the total emissions (Friedrich and Trois, 2013). The heavy diesel trucks and collection vehicles used in waste transportation worldwide form a large segment of this carbon footprint, releasing not only carbon emissions but also odours and particulate emissions into the air (Gioria et al., 2020; Pulles et al., 2012; Suarez-Bertoa et al., 2020). Following the treatment of municipal solid waste (MSW) entails a series of processes aimed at minimising the adverse effects of waste accumulation and disposal, however it is essential to acknowledge that these technologies and procedures also entail resource consumption, thereby imposing burdens across various impact categories within the environment. The final step in the waste management hierarchy is treatment; waste treatment necessitates utilising resources such as water, fuel, and electricity, resulting in associated environmental burdens, including greenhouse gas emissions, air pollution, and water usage. Processes such as incineration release pollutants into the atmosphere. At the same time, landfills generate biogas, which contains a mixture of gases consisting mainly of methane (CH_4) and carbon dioxide (CO_2), and a non-stabilised digestate, that are the final products of the anaerobic waste decomposition (Cerda et al., 2018). Biowastes and bio-decomposable materials that end up in landfills are considered the primary contributors in this process since they constitute more than 44% of total household waste globally (Ardolino et al., 2020; Bartocci et al., 2020). Although some waste treatment methods enable energy recovery, their overall environmental benefits hinge upon waste composition, process efficiency, and the availability of renewable energy alternatives.

The scientific management of solid waste poses significant challenges for cities in developing countries since each region has unique characteristics and circumstances that prevent implementing a standardised waste management system everywhere. Local data, individualities, and economic conditions complicate selecting an appropriate waste management system. On the other hand, managing selective waste collection and various treatments, mainly recycling, plays a vital role in achieving a circular economy. This entails more efficient resource utilisation and a greater emphasis on environmental preservation than the traditional linear economy (López-Portillo et al., 2021). This makes the integrated waste management system complex, as environmental implications must be considered and compared. In this context, LCA has become a valuable tool for waste management decision-makers.

LCA is a computer-based tool used to evaluate the environmental impacts of a product or service throughout its entire life cycle, including raw material acquisition, production, use, and disposal (Christensen et al., 2020; Klöpffer, 2014). In waste management, LCA examines the potential environmental impacts of a waste life cycle, from its generation to its disposal. Numerous published studies have demonstrated the popularity of LCA in analysing MSW management systems, and various organisations, such as the International Organization for Standardization (ISO), have contributed to developing LCA methodologies.

However, a comprehensive LCA study necessitates the careful examination of numerous data parameters and variables, requiring multiple calculations that rely on powerful computing resources provided by specialized software tools. The utilization of such tools, in conjunction with actual local data, has enabled waste managers to explore and address diverse inquiries pertaining to effective waste management, while considering the environmental consequences and taking into consideration the unique local, cultural, and geographical characteristics of each study area.

Research topics

This thesis centres its investigation on applying the LCA methodology and advanced LCA tools to analyze, assess, and propose methodologies to enhance waste management practices within the jurisdiction of Chania Prefecture. Given the intricate nature of the current waste management framework and the substantial volume of data, this research has been partitioned into two distinct segments. The initial segment is dedicated to scrutinizing the processes associated with waste collection and transportation, commencing from the collection containers and culminating at the waste treatment facility. The second segment investigates the fate of waste, spanning from its treatment facility disposition onwards. Each of these segments is meticulously designed to scrutinize pivotal facets of environmental impacts and to present prospective scenarios for alleviating their ecological burdens (**Figure 1**).

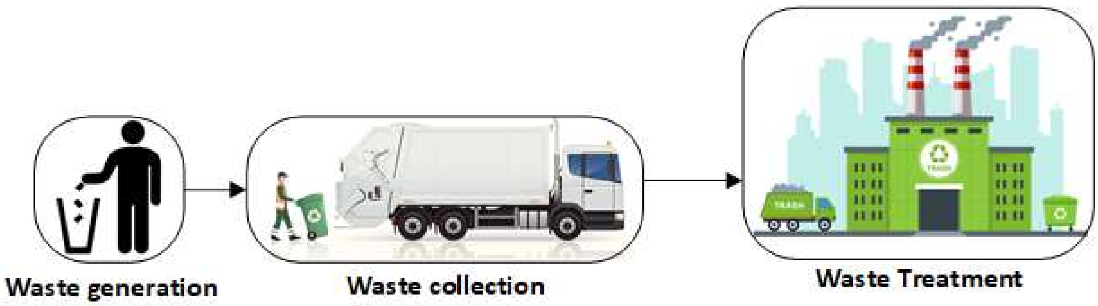

Figure 1 *shows the main processes of waste management that this study focuses on based on the case study of the Prefecture of Chania.*

Waste collection

In waste management systems, the term waste collection covers all the steps, including transporting waste materials from the collection containers to the treatment facility or disposal site (Ghiani et al., 2021; Yadav and Karmakar, 2020). Usually, in cities, the MSW is collected and transported by specialised collection vehicles following predetermined routes. Depending on the area and waste production, the vehicles travel periodically around the city. Modern waste management systems employ transfer stations (WTS) to increase efficiency and cost-effectiveness in transporting waste materials to treatment facilities (Höke and Yalcinkaya, 2021). The waste collection process is often divided into sub-processes. Some studies divide the process into collection and transport, while others further break it down into travel from parking, collection of waste, transport to the treatment facility, and return journey to the parking station (Brogaard and Christensen, 2012; Larsen et al., 2009).

The environmental impacts of waste collection and transport depend on variables like the weight of waste collected, energy consumption (e.g., fuel) and the distance the waste material is transported. Several suggestions have been proposed to enhance collection efficiency, reduce cost, and lessen the environmental impacts to achieve a more environmentally friendly procedure. Many systems have also incorporated WTS to make transport to treatment facilities more efficient and cost-effective (Jia et al., 2022b). The optimum location for installing a WTS depends on local and country economics (Chatzouridis and Komilis, 2012). Although the cost and benefits of WTS have been extensively studied, there is a lack of research to evaluate the environmental impacts of integrating WTS into waste management systems. However, the costs related to WTS, like loading and unloading tractors and energy and fuel consumption, are overlooked (Antonopoulos et al., 2013; Nhubu et al., 2019). The methodologies accounting for emissions from waste collection often rely on hard-to-find data or are case-specific, thus not necessarily applicable to different locations. Most of them are focused on the collection phase, proposing algorithms considering several specialised data like number of containers, distance between them, and less on the transport part (Alberdi et al., 2020; Friedrich and Trois, 2013b; Pérez et al., 2017). This thesis aims to fill this research gap by proposing a methodology that utilises readily accessible data to waste collection agencies, such as weight data logs, annual fuel consumption, and total distance travelled. The waste management in Chania prefecture represents Greece's typical collection and transport system. It was one of the first regions that introduced source separate waste collections for recyclable materials.

Biowaste composting modelling

Biowaste (BW) represents a significant fraction of municipal solid waste (MSW), which consists of food and kitchen waste from households, HORECA (hotels, restaurants, caterers), and green waste from gardens and parks (Malamis et al., 2017). It comprises the most significant fraction of household waste, reaching up to 44% globally (Ardolino et al., 2020; Bartocci et al., 2020).

Biowaste recovery and utilisation started in the 90s with Directive 1999/31/EC (Council of the European Communities, 1999). This directive obliges the member states of the EU to reduce the amount of biodegradable municipal waste and aims for 65% of all MSW produced to be recycled before 2030 (while only 10% should be disposed of in landfills). A feasible approach used for the past 20 years is recovering the organic fraction of municipal solid wastes (OFMSW) from unsorted waste and used in biological processes such as composting and anaerobic treatment. Anaerobic treatment has been studied in several cases since biogas can be produced for energy recovery (Fan et al., 2018; Wi et al., 2020). The research in mechanical closed composting systems has focused on source-segregated OFMSW processes. The mechanical sorting systems vary from simple installation, such as shredder, trommel, and magnet, to medium or high complexity OFMSW sorting systems to deliver various quality and purity materials (Graça et al., 2021).

Diverting OFMSW from landfills and using it as composting material has many environmental benefits (e.g., reducing greenhouse gas emissions (Wei et al., 2017)), while it can be easily integrated with material recovery facilities (MRFs). It involves the biological aerobic degradation of organic matter under controlled conditions (Díaz et al., 2021), resulting in a nutrient-rich product The resulting product when OFMSW originated from unsorted mixed waste is called compost-like output (CLO) (Carabassa et al., 2020), and its quality is related to the purity of the initial materials and the pretreatment method (Vasileiadou et al., 2021). Compost can significantly enhance the fertility of the soil environment by increasing the soil's organic carbon (SOC), total N (TN), and soil microbial biomass (SMB). At the same time, it positively affects the activity of enzymes involved in the C, N, and P cycles (Bhattacharyya et al., 2003). However, the impurities and contaminants usually released from CLO require increased attention (Wei et al., 2017). CLO is considered one of the primary sources of microplastics (MPs) in the agricultural environment, negatively influencing soil microbial processes or plant growth (Baiano et al., 2021). Therefore, the use of CLO in land applications is limited and regulated to restore quarries, dumping sites, or road slopes (Carabassa et al., 2020; Palansooriya et al., 2020; Wei et al., 2017).

Societies have started transitioning towards a model based on source segregation of biowaste that can produce higher-quality compost with significantly reduced environmental impacts. At the same time, the existing facilities are adapted to accept source-segregated biowaste. The evaluation of the composting systems is complicated, with many variables which must be considered. It involves numerous calculations and requires accurate data to model the variables of each system better. The use of LCA is based on the guidelines of ISO 14040 and 14044 (BSI, 2006), and can provide a much-improved viewpoint on waste management by connecting materials, resources, and waste flows with potential environmental impacts. Every LCA study incorporates several available local information and data sets called the life cycle inventory (LCI). In particular, the LCI is a compilation of all mass flows and emissions associated with the activities within the waste management system as well as upstream and downstream activities linked to the management of the waste. It relies on recent, representative, and accurate data such as waste types and their individual material fractions, detailed physicochemical composition, mass balances for all relevant material fractions, energy balances for all processes and technologies, records of the emissions, and inventories of all relevant upstream and downstream processes (Christensen et al., 2020). However, it is challenging to find case-specific data or LCI that include waste composition, energy and resource inputs, and material substitution in an LCA implementation study (Ripa et al., 2017). Establishing a relevant and high-accurate LCI is often demanding but crucial since it is the technical basis for assessing the waste management system. Existing models and software offer some assistance and databases in setting up the LCI, but it is always important to ensure relevance and consistency in the technical data of the specific study. LCA methodologies and advanced software such as EASETECH (developed by DTU) are based on fragmentation analysis to follow elemental balances throughout the processes.

Inventories on existing facilities managing the OFMSW from unsorted mixed waste are scarce (Laurent et al., 2014). In most cases, such materials are treated by private facilities, and the available data concerning full-size treatment are not published. Although several composting technologies have been studied in European countries, a few have developed LCI for composting systems treating segregated biowaste (Pini et al., 2018). The available information about the materials of OFMSW and nutrient flows are inconsistent, making it difficult to develop alternative scenarios during urban planning (Guo et al., 2019). Therefore, there is an increasing need for predictive models to support environmental policy and decision-making. Few studies have investigated the composting of mechanically sorted OFMSW obtained at MBT plants. Thus, this research in this field is urgently needed (Mironov et al., 2021)

Source-segregated biowaste composting

The adoption of the Circular Economy Package deals with a reduction of municipal solid waste being landfilled to 10% by 2035, the improvement of biowaste management goals to gradually increase recycling to 65% of the total solid waste produced by 2035, the promotion of biowaste separate collection by 2023, and exclusion of mechanical and biological treatment of municipal waste as recycling by 2027. At the same time, EU strategy toward a circular economy focuses on material flows and recycling to keep materials in the loop.

To comply with the new EU directives, the need for cleaner materials mandates transitioning to a source-segregated biowaste collection (by adopting the brown container). In this direction, the involved treatment facilities must adapt to this transition by altering or redesigning their infrastructures or processes to accept and treat segregated biowaste. Many researches have proven the environmental benefits of separate biowaste collection and composting against traditional waste management systems like landfilling (Ardolino et al., 2017; Colón et al., 2015; Martínez-Blanco et al., 2010; Seruga and Krzywonos, 2021). The environmental behaviour of transitioning from mechanical separation OFMSW composting to a separate biowaste collection is still unknown. At the same time, the impacts of using the recovered materials and producing compost have also not been extensively studied (Bourtsalas and Themelis, 2022). To the best of the authors' knowledge, the literature lacks studies that address this matter from the cradle-to-the-grave perspective, considering the fate of remaining stream behaviour and the recovery of recyclable materials.

Biowaste typically has a high water content (> 60%) (Laurent et al., 2014) that migrates among the waste components under mixing and compression during waste collection and transportation. Recyclable materials in the waste streams are affected and contaminated, and their recoverability is significantly compromised (Eriksen et al., 2018; Magrinho and Semiao, 2008; Pivnenko et al., 2014). The gradual diversion of biowaste from mixed waste could restore recyclable recoverability, which depends on their composition and the technology used in the MBTs (Pressley et al., 2015). These facilities are usually costume-designed, addressing the input streams and the desirable products (Bourtsalas and Themelis, 2022). The operating data of these facilities are proprietary, while the equipment and processing of different materials are unknown or theoretical (Ardolino et al., 2017). Only a few mechanical sorting facilities in Europe can process mixed waste and recyclables. It is challenging to obtain their inventories, making it valuable to study their environmental performance (JRC-IES, 2010).

Objectives of PhD thesis

The main scope of this thesis is to prove that using LCA's advanced tool, a waste management system can be studied, modelled and improved to comply with the goals of sustainable integrated waste management dictated by the new EU policy and directives. To accomplish this, it focuses on modelling the two primary integrated waste management processes, collection and treatment, studying them in environmental criteria and proposing alternative solutions. This is accomplished by the three studies presented below.

Inventory and LCA of Waste Collection and Transfer: The study aims to create a waste collection inventory for mixed waste and source-segregated recyclable fragment collection and transport and evaluates the environmental impacts of positioning WTS in the study area. The study is based on the life cycle inventory of Chania prefecture, utilising data available to most waste managers like collection truck monthly and annual fuel consumption, distance travelled, and waste weight collected.

Life Cycle Inventory (LCI) for Composting: The study aims to develop a LCI for the composting process, specifically focusing on the mechanical recycling and composting facility in Chania, Greece. This involves modelling the composting unit and mapping the fragmental mass balance between its sub-processes. The aim is to quantify the inputs (e.g., water, electricity, fuel) and outputs (e.g., emissions) associated with the composting process. The data collected over two years are used to create a comprehensive LCI.

LCA of Source-Segregated Biowaste Composting: This investigation aims to conduct a LCA comparing the environmental impacts of source-segregated biowaste composting versus mechanically segregated biowaste composting from mixed waste. This assessment utilises data from the Chania integrated waste management system, representing a typical Mediterranean system. The study employs advanced LCA tools like EASETECH and the ReCiPe 2016 Life cycle impact assessment methodology to evaluate the environmental consequences of different waste management practices.

Structure of PhD thesis

The PhD thesis is comprised of six chapters. See below:

Chapter 1: Introduction and Objectives

This chapter presents the research topics and objectives of the PhD thesis. It provides a comprehensive overview of the research questions and the significance of the study. The chapter highlights the innovative aspects of the research. It outlines the contributions to make to integrated solid waste management.

Chapter 2: Background and Literature Review

This chapter provides a thorough background on waste management, including its historical context and the legislative frameworks in the EU and Greece. It explores the critical components of a waste management system. The concept of LCA and its relevance to waste management are introduced. The chapter also provides an overview of the methodologies used for environmental impact assessment. Ultimately, it presents the most popular advanced LCA tools developed in use.

Chapter 3: Positioning transfer stations for waste collection and transport using LCA modelling

This chapter proposes a methodology for creating a life cycle inventory related to waste collection and transport, using the case study of the Chania region. It explores the environmental implications of introducing waste transfer stations in a conventional waste collection system. and studies the implications of positioning waste transfer stations based on environmental criteria for mixed waste and recyclable material collection.

Chapter 4: Methodology for Modelling Waste Composting Systems

This chapter presents a detailed methodology for modelling and creating a life cycle inventory for municipal waste composting systems. The focus is on simulating and studying the composting of mechanically recovered organic fragments of municipal solid waste (MSW) and green waste or source-segregated biowaste, using the case study of the region of Chania. The chapter explains the steps involved in the modelling process and highlights the key considerations and data sources.

Chapter 5: Environmental Impacts of Source-Segregated Biowaste Treatment

Using actual data from the case study of the integrated waste management system in the Chania region, this chapter applies the LCA methodology to evaluate the environmental impacts of introducing a source-segregated biowaste collection system. The chapter examines the existing waste management system that utilises mechanically segregated biowaste and assesses the additional environmental benefits and trade-offs associated with the source-segregated collection approach.

Chapter 6: Conclusions and Future Research Recommendations

The final chapter of the PhD thesis summarises the main findings and conclusions drawn from the research. It highlights the contributions to solid waste management by applying advanced LCA tools. The chapter also identifies potential areas for future research. It provides recommendations for further studies to enhance the understanding and optimisation of solid waste management systems.

Contribution and novelty of Ph.D. thesis

This study makes a substantial contribution to the existing body of literature on waste management and Life Cycle Assessment (LCA). It explores and introduces innovative inventories encompassing various aspects of waste management processes, including collection and transport, mechanical composting, and mechanical sorting of waste. Noteworthy is the meticulous delineation of the fate of individual waste materials throughout each studied process, departing from the conventional treatment of waste as a singular stream—a departure not observed in analogous studies.

The thesis represents a significant advancement by constructing inventories grounded in precise real-world data and employing straightforward methodologies capturing emissions from waste collection, utilizing a fragmental approach for all waste streams in 19 materials throughout all processes. The methodology draws on readily available information applicable to diverse locations, incorporating data routinely maintained by waste collection agencies, such as weight data logs, annual fuel consumption records, and total distance traveled. This endeavor aims to formulate a valuable inventory for waste collection and transport, offering practical insights for waste managers and researchers engaged in Life Cycle Assessment (LCA) studies within the waste management domain.

Another substantial contribution lies in the development of a comprehensive Life Cycle Inventory (LCI) focused on the mechanical recycling and composting facility in Chania, Crete. The LCI model intricately maps the mass balance among sub-processes, monitors the release of carbon (C) and nitrogen (N) emissions to the environment, and records the consumption of water, electricity, and fuel for treating organic fraction municipal solid waste (OFMSW) introduced to the facility. The two-year waste sampling and data collection endeavors yield detailed insights beneficial for waste management practitioners in estimating outputs and costs associated with treating OFMSW and source-segregated biowaste.

Furthermore, the thesis employs LCA to evaluate the environmental impacts of biowaste segregation in mechanical-biological treatment (MBT) facilities. This study marks the first instance where source-segregated Biowaste treatment is compared to mechanical sorting biowaste treatment based on environmental criteria. The environmental repercussions of biowaste segregation are quantified using real regional and time-specific data for treatment technologies, waste generation, and flows. Computer-based LCA tools streamline assessment procedures, alleviating waste managers and scientists from intricate calculations.

Finally, a notable aspect of this research is installing a Waste transfer station based on environmental criteria for the first time. Existing methodologies accounting for emissions from waste collection often rely on hard-to-find data or are case-specific, not necessarily applicable to different locations. Most of these methodologies focus on the collection phase, proposing algorithms considering specialized data, such as the number of containers and the distance between them, with less emphasis on transport. This study addresses this research gap by proposing a methodology that utilizes readily accessible data from waste collection agencies.

Publications

Scientific publications in journals

1. Panagiotis Chazirakis, Apostolos Giannis, Evangelos Gidarakos, Modeling the Life Cycle Inventory of a Centralized Composting Facility in Greece, Applied Sciences 2022. (12), 2047.

2. Panagiotis Chazirakis, Apostolos Giannis, Evangelos Gidarakos, Material flow and environmental performance of the source segregated biowaste composting system. Waste Management, 2023 (160) 23–34,

3. Panagiotis Chazirakis, Apostolos Giannis, Evangelos Gidarakos, Positioning transfer stations for waste collection and transport using LCA modelling (under review)

Participation in conferences

1. Panagiotis Chazirakis, Apostolos Giannis, Evangelos Gidarakos, Creating a life cycle inventory of a centralised composting facility in the Mediterranean region, 7th international conference on Industrial & Hazardous Waste Management (CRETE 2021) 27-30 July 2021, Chania Crete.

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2. Chapter

Waste management

Waste management (or waste disposal) encompasses the activities and processes necessary for managing waste from its generation to its final disposal. This includes waste production, collection, transportation, treatment, reuse (as raw materials), or disposal, as well as the monitoring and regulation of the waste management process, waste-related laws, technologies, and economic mechanisms. Waste management deals with all types of waste, such as industrial, biological, household, municipal, organic, biomedical, and radioactive wastes. Waste can sometimes pose a threat to human health, with health issues arising from waste management processes, either directly or indirectly through the processing and handling of solid waste or through water, soil, and food consumption. Waste management aims to minimise the negative effects of waste on human health, the environment, planetary resources, and aesthetics.

Historically, health and safety have been the major concerns in waste management. These concerns still apply – waste must be managed to minimise risks to human health. Today, society demands more than just safety, waste management must also be sustainable. Sustainability or Sustainable Development has been defined as 'development which meets the needs of the present without compromising the ability of future generations to meet their own needs' (WCED, 1987). There must be synergy between economic development, social equity, and the environment. Therefore, sustainable waste management must be:

- Economically affordable
- Socially acceptable
- Environmentally effective

Proper waste management is crucial for building sustainable and livable cities, but it remains challenging for many developing countries and cities. Waste management practices vary among countries (developed and developing nations), regions (urban and rural areas), and residential and industrial sectors, with different approaches taken by each. Many waste management practices deal with municipal solid waste (MSW), the bulk of waste created by household, industrial, and commercial activities. This essential municipal service requires integrated systems that are efficient, sustainable, and socially supported. Measures of waste management include integrated techno-economic mechanisms of a circular economy, adequate disposal facilities, export and import control, and optimal sustainable design of products produced.

History of WM

The inception of waste management can be traced back to the shift in human lifestyle, wherein communities were established due to abandoning a nomadic existence over 12,000 years ago. This societal transformation led to the accumulation of solid waste, as humans began to generate refuse in concentrated areas (Rada, 2013). The issue of waste accumulation was further exacerbated as people continued to reside amidst the filth, revealing a distinct behaviour among the human species. Waste management thus emerged as an indispensable aspect of human settlements, reflecting the necessity to address the consequences of waste generation to maintain public health and environmental stability.

As the Industrial Revolution dawned, developing and utilising new materials and substances produced more durable physical corrosion waste. Concurrently, integrating these materials into the food chain precipitated many health issues that amplified and exacerbated the existing waste management challenges. Consequently, the need for systematic and comprehensive waste management strategies became increasingly pressing as the scale and complexity of waste generation expanded rapidly.

In the twentieth century, we have witnessed the depletion of fossil fuels and natural resources, further accentuating the environmental concerns arising from waste accumulation. This resource scarcity and emerging environmental issues compelled authorities to explore and implement innovative solutions for conserving raw materials and reducing and preventing waste generation. The concept of a circular economy was subsequently adopted to respond to these pressing challenges, promoting the efficient use of resources, minimising waste, and fostering a sustainable economic model.

In recent years, waste management has evolved to encompass a broader range of practices and objectives, including waste reduction, recycling, and developing waste-to-energy technologies. The transition towards a circular economy has necessitated reevaluating traditional waste management methods and integrating novel and sustainable practices. As we continue to confront the complexities of waste management, we must devise and implement effective strategies that prioritise environmental preservation and human well-being.

Legislation of Waste Management

European waste legislation

Early Initiatives

The history of European waste management legislation dates back to the early 1970s when the European Union (EU) first began to acknowledge the need for environmental protection and waste management. The 1975 Waste Framework Directive (75/442/EEC) was one of the first legislative initiatives to harmonise waste management practices among EU member states. It set basic definitions for waste and introduced the waste hierarchy concept, emphasising waste prevention, reduction, and recycling.

Evolving Frameworks and Regulations

Over the years, the EU has revised and updated its waste management legislation better to address the challenges of waste disposal and resource conservation. The 1991 Hazardous Waste Directive (91/689/EEC) strengthened waste classification and introduced stricter controls on hazardous waste disposal. In 1999, the Landfill Directive (1999/31/EC) established standards for landfill operations and waste acceptance criteria, aiming to reduce the negative environmental impacts of landfilling. In 2002, the Waste Electrical and Electronic Equipment (WEEE) Directive (2002/96/EC) and the Restriction of Hazardous Substances (RoHS) Directive (2002/95/EC) were introduced to tackle the growing issue of electronic waste and hazardous substances in electrical and electronic equipment. This directive was followed by a series of directives addressing waste issues, including the one passed in November 2008 (European Union Directive 2008/98/EC), which established a revised waste management framework. The goal continued to be the reduction in landfilling by applying the following waste hierarchy: Prevention, Preparing for reuse, Recycling, Another recovery (e.g., energy recovery), and then Disposal. **Figure 1**

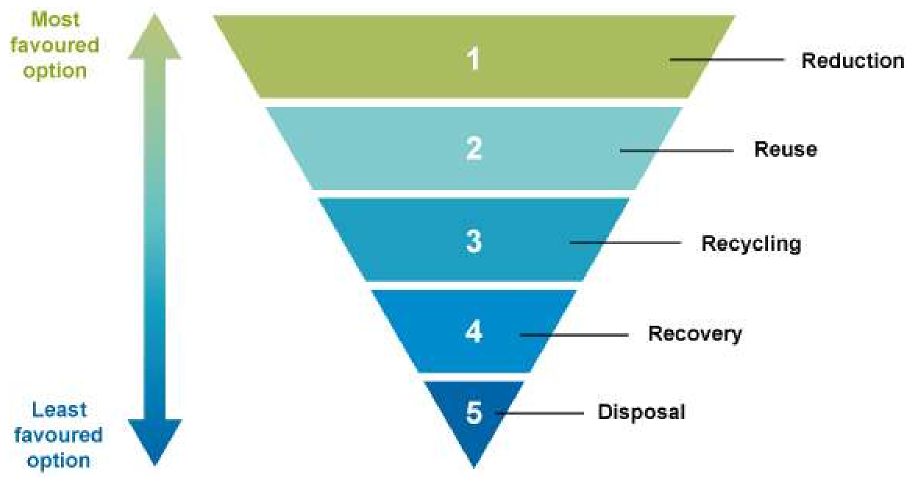

Figure 1*: The Pyramid of the waste hierarchy*

Emphasis on Circular Economy and Resource Efficiency

In 2008, the EU introduced the revised Waste Framework Directive (2008/98/EC), further developing the waste hierarchy concept, promoting waste prevention, reuse, and recycling. The goal continued to be reducing landfilling by applying the following waste hierarchy: Prevention, Preparing for reuse, Recycling, Another recovery (e.g., energy recovery), and then Disposal—**Figure 1**. The directive also introduced the "polluter pays" principle, holding waste producers responsible for the costs associated with waste management. The EU's focus on resource efficiency and the circular economy was reinforced with the adoption of the Circular Economy Package in 2015, which included amendments to key waste directives, such as the Waste Framework Directive, the Landfill Directive, and the Packaging and Packaging Waste Directive, as well as new targets for recycling and waste reduction.

Greek Waste Management Legislation History

As a member of the European Union, Greece adopted and implemented EU waste directives into its national legal framework. These directives establish the overall framework for waste management, set targets for recycling and waste reduction, and define the responsibilities of waste producers and operators.

National Waste Management Plan

A National Waste Management Plan (NWMP) was developed following the Waste Framework Directive's requirements. The NWMP provided a strategic roadmap for waste management, focusing on waste prevention, reuse, recycling, and recovery. It also set objectives for reducing waste disposal in landfills, improving waste management infrastructure, and promoting the circular economy. The plan has periodically been updated to align with evolving EU legislation and targets.

To promote waste reduction and recycling, Greece has implemented Extended Producer Responsibility (EPR) schemes for various waste streams, such as packaging, electrical and electronic equipment, batteries, and end-of-life vehicles. EPR schemes hold producers responsible for collecting, treating, and recycling their products once they become waste. These schemes aim to incentivise producers to design more easily recyclable products with a reduced environmental impact throughout their life cycle.

Waste management has primarily the responsibility of regional and local authorities. The country is divided into 13 administrative regions, each with its own Regional Waste Management Plan (RWMP) that aligns with the National Waste Management Plan. At the local level, municipalities are responsible for waste collection, transportation, and in some cases, treatment. They must also implement local waste prevention and recycling programs following national and regional plans.

Despite progress in waste management legislation and infrastructure, Greece still faces challenges in fully implementing EU waste management targets and transitioning to a circular economy. The main challenges are low recycling rates, insufficient waste sorting and separation at source, limited public awareness and participation, and inadequate waste management infrastructure in some regions. The government has to focus on several key areas to address these challenges and support the transition to a circular economy. These include improving waste management infrastructure, particularly developing more advanced recycling and waste treatment facilities. Additionally, efforts are being made to enhance the waste collection and sorting systems, such as expanding separate collection schemes for various waste streams.

Public education and awareness campaigns are also being prioritised to encourage citizens to actively participate in waste reduction, reuse, and recycling initiatives. Collaboration between governmental authorities, industries, and communities is essential for creating a shared understanding of the benefits of proper waste management and the circular economy.

Integrated Waste Management - Key components

Integrated Waste Management System (IWMS) is a comprehensive waste management approach encompassing various waste management, prevention, and reduction strategies. The overarching goal of IWMS is to provide environmental sustainability, economic affordability, and social acceptance for any specific region. This is achieved by combining treatment options, including waste reduction, reuse, recycling, composting, thermal treatment, and landfilling. The crucial aspect is not the number of waste management options employed or whether they are applied simultaneously, but instead that they are integrated optimally as part of a cohesive approach, implementing the most effective treatment methods to maximise environmental protection and social benefits while minimising economic costs. **Figure 2** (Quattrociocchi et al., 2014).

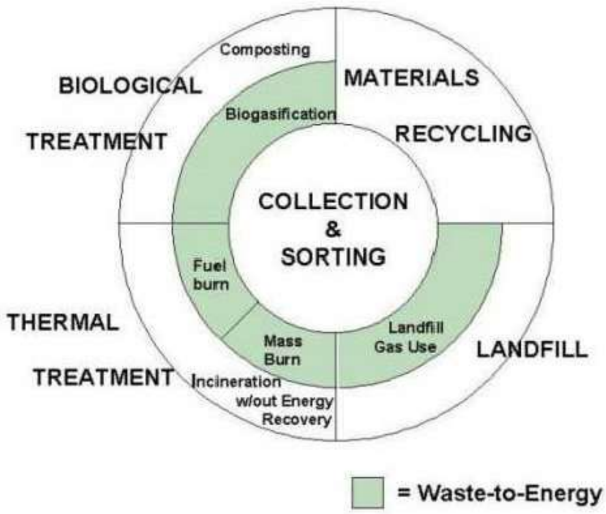

Figure 2: *illustrates the potential components of an IWM system in a developed country designed to manage municipal solid waste. The waste streams would comprise a complex array of materials necessitating a corresponding array of treatment options (Quattrociocchi et al., 2014)*

Fundamental elements of waste management systems are methodologies and technologies aimed at recovering materials, reusing, or processing waste to generate new materials or substitute raw materials. Integrated waste management comprises the following components: collection and transport, sorting and separation technologies, biological treatment technologies, thermal treatment technologies, and, ultimately, disposal.

Each of these technologies is designed to handle, collect, separate, or reduce the environmental impact of the generated solid waste. Although ISWMs are designed to manage waste and address environmental issues, solid waste management, as a process, is known to be a significant contributor to various environmental problems, such as climate change (e.g., from greenhouse gas emissions from landfills), stratospheric ozone depletion (e.g., from halocarbon emissions in discarded cooling systems or in-use foams), human health damages (e.g., from exposure to chemicals and particles during waste collection and treatment), ecosystem damages (e.g., from heavy metal emissions to air, soil, and surface water), and resource depletion (e.g., due to nonexistent or inefficient recycling systems for certain critical minerals or metals), among others. The alarming increase in solid waste generation thus necessitates management systems that comprehensively address these environmental challenges and ultimately contribute to the transition towards a more environmentally sustainable society (Bakas and Milios, 2013; Laurent et al., 2014b), as an industrial process consumes energy and resources to achieve its objectives and inevitably interacts with the environment.

In the subsequent sections, some of the primary processes of an IWMS studied in this thesis are briefly presented.

Collection and transport

The waste collection encompasses all the processes involved in waste generation, sorting, collection, transport, and delivery to the treatment facility or final deposition. It can be considered a multi-phase process, with at least five distinct phases, as illustrated in the **Figure 3**. Initially, the homeowner must transfer whatever is regarded as waste to the refuse can, either inside or outside the home. The second phase involves moving the trash can to the truck, typically carried out by the collection crew, referred to as backyard collection. If the can is transferred to the street by the waste generator or the home occupant, the system is called a curbside collection. The third part consists of waste compaction and transport from house to house throughout the scheduled area, where additional waste is collected. In most instances, a direct transport route is employed or, in other cases, a transfer phase to a larger vehicle, after which the material is transported to its final treatment or disposal site. Lastly, the final phase is the discharge to the destination.

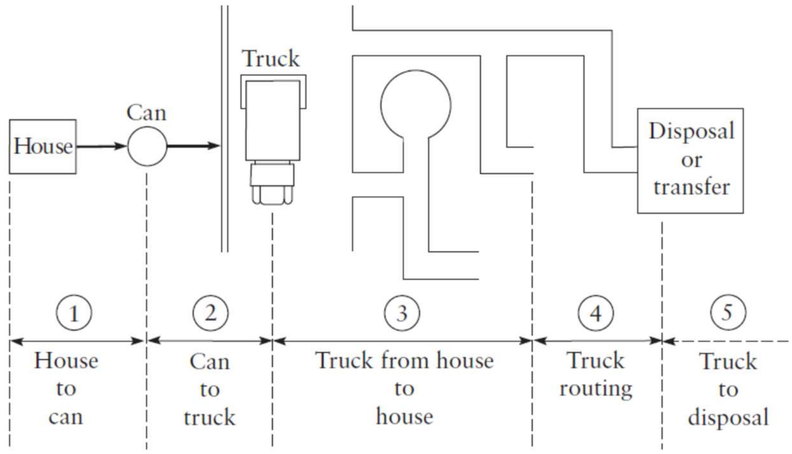

Figure 3*:Five phases of solid waste collection* (Worrell and Vesilind, 2011)

The collection is a sensitive and critical aspect of integrated waste management. It is labour-intensive, time-consuming, and resource-demanding, influencing subsequent waste treatment performance and recovered materials' quality. An efficient Municipal Solid Waste (MSW) collection system necessitates careful planning. Local authorities and municipalities primarily develop collection and sorting strategies, while different residential units may have distinct collection and sorting mechanisms. Solid waste collection systems are predominantly person/truck systems. MSW collection is typically performed by workers who traverse a town in trucks and then ride with the truck to a site where the truck is emptied.

Waste collection is a vital component of waste management, with numerous factors, flows, and materials to consider when modelling or designing such systems. Garbage bags, collection containers, collection trucks, and cleaning and protective equipment are employed for this purpose, and their environmental impact is often overlooked in many studies. Sorting, collection, and transport systems directly influence the environmental performance of recycling/disposal activities through emissions from the involved activities. They also have an indirect impact by affecting the reprocessing quality, facilitating acceptable input for subsequent treatment steps. Consequently, these systems warrant attention regarding the environmental performance of End-of-Life (EoL) systems (Erkisi-Arici et al., 2021).

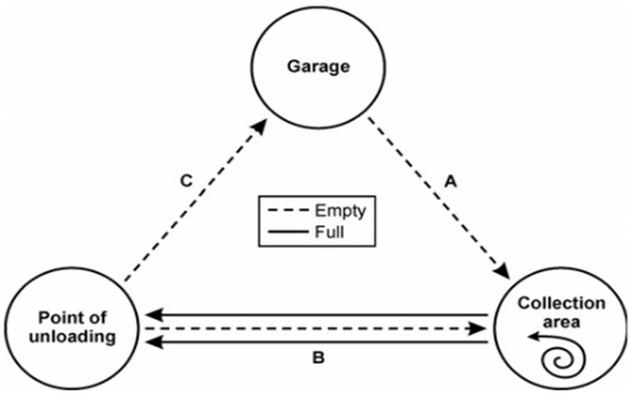

Figure 4: *Conceptual model for collection and transport of waste (Larsen et al., 2009)*

A well-established collection system enhances the performance of the sorting facility, achieving a suitable waste stream for recycling activities, a large waste flow, and minimal contamination. Modelling a collection system involves multiple parameters to consider, beginning with the infrastructure, equipment, and materials used. Diesel consumption per tonne of waste collected depends on various factors related to waste, the collection area, the truck, the distance to the unloading point, and the driver. Several models predict diesel consumption during waste collection based on detailed information on the number of stops, bins per stop, the distance between stops, etc. (Madden et al., 2022; Sonesson, 2000). However, larger collection areas' input parameters are highly variable and complex. One reason for the high degree of parameterisation is that the models also calculate operation time used in economic optimisation and assessment. The time aspect is not relevant for assessing the environmental burden of waste collection (Larsen et al., 2009).

Waste sorting Material Recovery Facilities (MRFs)

The recovery of reusable and recyclable materials typically occurs in dedicated facilities designed for sorting, separating, and collecting materials, known as Material Recovery Facilities (MRFs). Various technologies can be employed for material collection based on manual or mechanical separation processes, utilising several physical properties such as size, density, shape, colour, or other physicochemical properties. Facilities can process one or multiple waste streams simultaneously or separately, while MRFs can also be part of a complex that includes additional material treatment processes such as composting or incineration units.

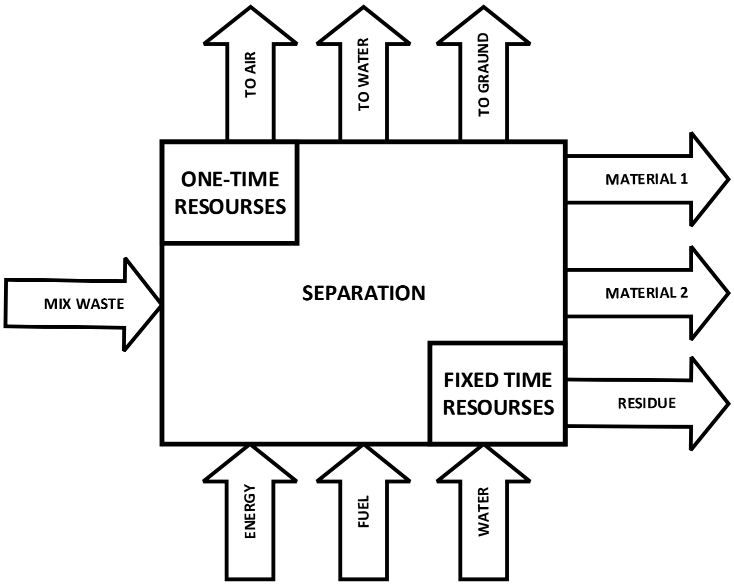

Figure 5: *Schematic flow of a general separation process*

These units are often characterised by their capacity for treated material, measured in kilograms or tonnes per day or year. The resources consumed or utilised in these facilities generally include:

- The local grid typically provides energy in the form of electricity. However, marginal technologies for electricity production are also commonly employed. In many cases, electricity generated from waste incineration or biogas utilisation from anaerobic digestion or landfill gas is prevalent.
- Fuel for vehicles handling and transporting waste and recovered materials.
- Water for material treatment, cleaning, or other purposes.
- Other resources include lubricants, maintenance materials, or other consumables.

Waste separation technologies

The primary processes occurring in these facilities are waste separation technologies, which are divided into gravity separation, electrostatic separation, magnetic density separation, flotation, and sensor-based sorting. Auxiliary technologies typically found in plastic recycling plants, such as magnetic and eddy current separators, are also described.

Some standard waste separation technologies used in MRFs include:

- Manual sorting, hand-picking, or robotic picking.
- Trommel screeners are large rotary drums shaped with a grate-like surface and large end openings used for separating coarse materials from bulk materials.
- Concentrating tables or density separators screen bulk materials based on density (specific gravity) and the size and shape of the particles.
- Air classifiers, cones, or cyclones utilise a spiral airflow action or acceleration within a chamber to separate or classify solid particles.
- Magnetic separators, which use powerful magnetic fields to separate steel, iron, and other ferromagnetic materials from non-magnetic bulk materials.
- Electrostatic separators, which employ preferential ionisation or charging of particles to separate conductors from dielectrics (non-conductors).
- Optical air-jet sorters that utilise cameras to detect predetermined plastics and accurately timed and positioned air-jets to propel selected items off the conveyor belt.
- Hydrocyclones, a type of static separator based on centrifugal separation, generating a vortex with a cono-cylindrical conuration.

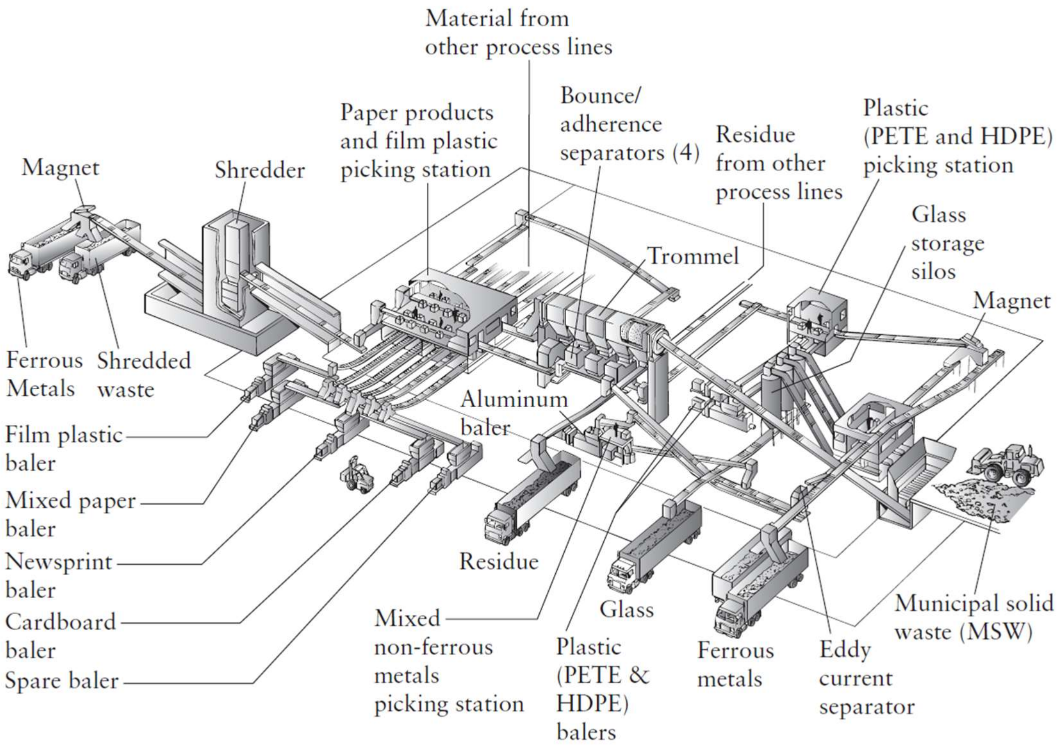

Figure 6*: The main processes occurring in a typical mix waste Facilities (Worrell and Vesilind, 2011)*

These systems are typically combined in MRFs to separate mixed materials that can be further reused, recycled, processed, or rejected without altering the physicochemical characteristics of the input materials.

Biological Treatment

Municipal waste comprises approximately 75% organic material, which can be converted into valuable energy through combustion or functional products via biochemical processes. Depending on the waste source, the organic components contain nitrogen, carbon, potassium, and other micronutrients suitable for soil use and substitution. Biological treatments are categorised into aerobic and anaerobic technologies, both of which rely on bacteria, nematodes, or other microorganisms to break down organic wastes, replicating natural processes and producing stable materials for land and agricultural use. The end-products of aerobic decomposition are stable and possess no additional energy for decomposing organisms (they are at their highest oxidation state). Conversely, the products of anaerobic decomposition still contain energy, as ammonia and hydrogen sulfide can be further oxidised, and methane contains significant energy that can be harnessed.

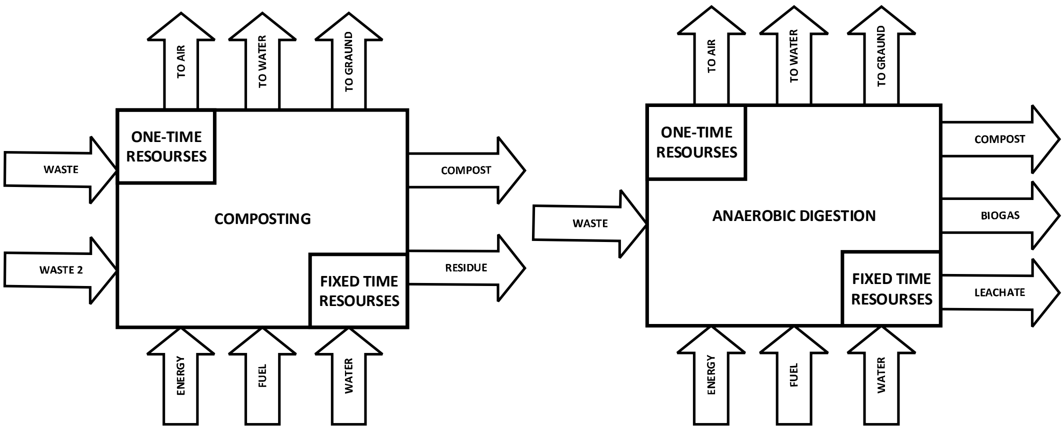

***Figure** 7: Schematic flow of a general aerobic process (composting) and an anaerobic process*

The three components of municipal solid waste (MSW) of most significant interest in bioconversion processes are garbage (food waste), paper products, and yard waste. The latter two are particularly valuable in biochemical processes as sources of cellulose, a potentially useful industrial raw material. The garbage fraction of refuse varies with geographical location and season and is influenced by factors such as dietary habits and living standards.

The two primary metabolic pathways for waste decomposition or degradation are aerobic (with oxygen) and anaerobic (without oxygen). An aerobic system can be generally represented as:

[complex organics] + O_2: CO_2 + H_2O + NO_3- + SO_4^2- + other stable

Anaerobic decomposition of organics can be described as:

[complex organics] + heat: CO_2 + CH_4 + H_2S + NH_4^++ products + heat

Facilities that process waste using biological processes are commonly referred to as mechanical biological treatment facilities (MBTs), which utilise composting, anaerobic digestion, or a combination of both to treat materials and simultaneously recover energy and soil improvers.

Composting is distinct from other processes as it is aerobic, and the end product is the partially decomposed organic fraction. Often promoted as a "natural" solid waste treatment process, composting is favoured due to its ease of implementation in backyard settings and its production of a valuable soil conditioner. Consequently, municipal engineers and city councils frequently face requests from citizen groups to adopt composting as an alternative to landfilling and combustion, which many perceive as wasteful of money and natural resources.

The primary resources accounting for biological processes include materials and one-time costs for infrastructure and equipment. Operational resources typically involve energy in the form of electricity and fuel for machinery, and water is a valuable resource needed in large quantities for aerobic and anaerobic processes. In some facilities, yeasts or minerals may be utilised to enhance biological processes or improve the final material.

Releases to the environment (air, water, and terrestrial) and exchanges with the environment are also critical factors to consider in the context of biological treatment of municipal waste, the processes aerobic vs anerobic, yield different final products, each with unique properties and applications. The final product of composting, known as compost or humus, is a nutrient-rich, stable material that can be used as a soil conditioner, enhancing soil fertility and structure, and characterised by the following properties:

- Nutrient content: Compost contains macro and micronutrients, such as nitrogen, phosphorus, potassium, calcium, magnesium, and trace elements, which are essential for plant growth.

- Organic matter: The organic matter in compost improves soil structure by enhancing aggregation, water retention, and aeration, promoting root growth and overall plant health.
- Microbial activity: Compost teems with beneficial microorganisms that contribute to nutrient cycling, suppress pathogens, and improve soil health.
- pH buffering: Compost can help buffer soil pH, making it more suitable for a wider range of plant species.
- Reduced environmental impact: Using compost in agriculture can reduce the need for synthetic fertilisers, decreasing the environmental impact of agricultural practices.

In contrast, the main final products of anaerobic digestion are biogas and digestate. Biogas is a mixture of gases, primarily methane (CH4) and carbon dioxide (CO_2), produced during anaerobic digestion. Methane, the primary component of biogas, is a potent greenhouse gas but can be harnessed as a renewable energy source for electricity generation, heating, or as a transportation fuel. Digestate, is the solid and liquid residues remaining after anaerobic digestion are collectively referred to as digestate.

Digestate is rich in nutrients, such as nitrogen, phosphorus, and potassium, and can be used as a soil conditioner or fertiliser. Its properties and applications include:

- Nutrient availability: Digestate releases nutrients slowly, providing a sustained supply to plants and reducing the risk of nutrient leaching into groundwater.
- Organic matter: Like compost, the organic matter in digestate can improve soil structure, water retention, and aeration.
- Microbial activity: Digestate contains beneficial microorganisms that enhance soil fertility and suppress pathogens.
- iv. Reduced environmental impact: The use of digestate in agriculture can decrease the reliance on synthetic fertilisers, minimising the environmental footprint of agricultural practices.

Disposal- Landfilling

Despite efforts to reuse, recycle, and recover energy from municipal solid waste (MSW), a portion inevitably returns to the environment. Landfilling is an engineered method for disposing of solid or hazardous waste on land, designed to protect the environment. Biological, chemical, and physical processes facilitate waste degradation within a landfill, leading to leachate production (polluted water emanating from the landfill's base) and gas emissions. Landfilling is the most prevalent waste disposal method worldwide.

In the context of global warming, landfills are complex due to the need to account for various greenhouse gas (GHG) emissions. Methane, a significant emission from landfills resulting from organic matter degradation, can be converted or recovered for energy purposes, potentially offsetting fossil fuel-based energy. Additionally, not all biogenic carbon in a landfill will be released within a given timeframe (e.g., 100 years), and bound biogenic carbon may be considered a carbon sink. (Manfredi et al., 2009)

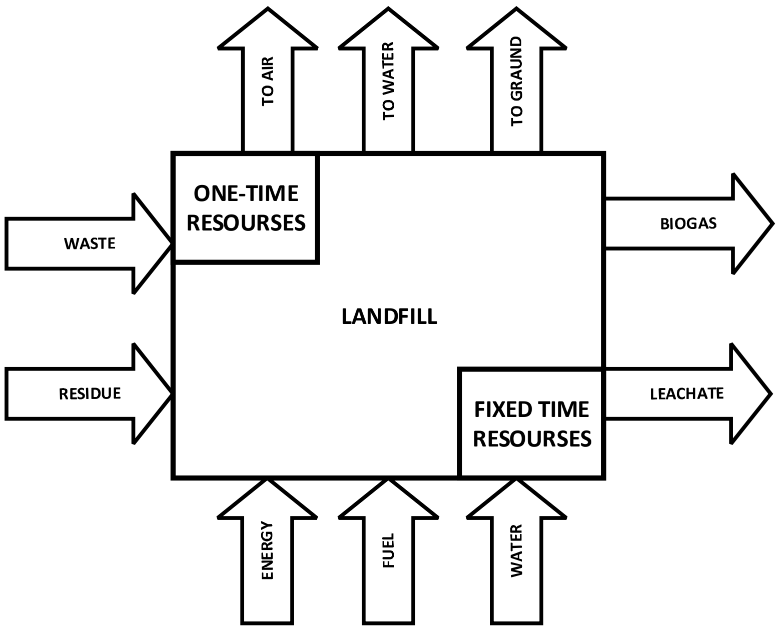

Figure 8: *the schematic flow of a general Landfill process*

Landfilling technologies have advanced significantly in recent decades, though these developments have not been universally implemented. Landfills range from simple dumps to highly engineered facilities, such as bioreactors, flushing bioreactor, and semi-aerobic landfills. Engineered landfills may employ landfill gas utilisation and control systems to reduce methane emissions and recover energy. Much of the current knowledge on landfills is based on mixed MSW landfills; however, as organic waste reduction becomes more prevalent in Europe, landfills containing less organic matter will become more common. These landfills will produce less gas, but their landfill gas recovery is likely to be less efficient.

A landfill is a dynamic system that functions long after the final materials have been deposited. Modelling a landfill requires consideration of several primary components and events in terms of space, mass, and time. Understanding and quantifying these processes involves analysing data from various studies and sources.

In conclusion, landfilling remains a widely used waste disposal method, encompassing a range of technologies from simple dumps to highly engineered facilities. While these systems continue to evolve to reduce greenhouse gas emissions and recover energy, the ongoing challenge is implementing advanced technologies more broadly and adapting to changing waste compositions.

Material substitution

Material substitution is a strategy that focuses on effectively and efficiently utilising raw materials to minimise waste generation throughout the processing system. It entails using alternative raw materials that do not produce waste during processing and incorporating reused or recycled materials. Material substitution is essential in waste management, as it calculates the avoided environmental impacts of replacing primary raw materials such as plastics, paper, metals, or fertilisers in various industries. Moreover, it considers substituting heat, energy, and fuel production and utilisation.

Material substitution encompasses three primary aspects:

Direct emissions and associated impacts: The environmental impact can be significantly reduced by substituting raw materials with alternatives that produce fewer emissions during processing and manufacturing. This approach helps mitigate the release of greenhouse gases and other pollutants, thereby contributing to a cleaner environment and improved public health.

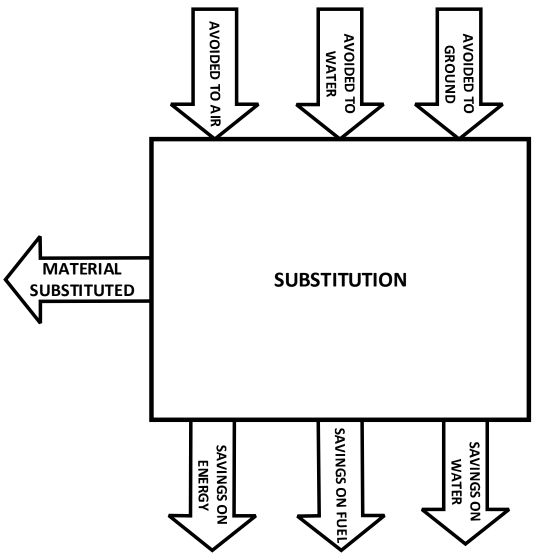

Figure 9: *schematic flow of a general material substitution process*

Material and energy needs and associated impacts: Material substitution addresses different industries' material and energy requirements and analyses the environmental consequences associated with their production chains. By identifying and using alternative materials and energy sources with lower environmental footprints, industries can effectively reduce their environmental impact.

Co-products and their substitution capacity: Material substitution also considers the potential of co-products, such as recycled materials and recovered energy, to replace the primary production of materials or energy. By utilising these co-products, industries can avoid the emissions and impacts associated with primary production processes, further enhancing their sustainability and reducing their environmental impact. A study by Viau et al. (2020) highlights the importance of substitution modelling in the life cycle assessment of municipal solid waste management. This research demonstrates that material substitution can provide tangible environmental benefits by replacing conventional materials and energy sources with more sustainable alternatives.

By carefully evaluating material substitution's potential impacts and benefits, industries and policymakers can make informed decisions to improve waste management practices and contribute to a more sustainable future.

Life cycle Assessment

In this part of the chapter, we look into the LCA concept, exploring its historical development, standardisation processes, integral components, and challenges associated with its implementation.

Introduction to Life Cycle Assessment (LCA)

LCA represents a standardised, scientifically grounded approach to assessing and quantifying the environmental impacts associated with a product or process system. LCA encompasses the examination of impacts stemming from potential harm to human health, ecosystems, and the capacity of future generations to flourish, including utilising natural resources. This method is extensively employed as a decision-support instrument, aiding in identifying crucial environmental factors and facilitating evaluating and comparing their environmental profiles (BSI, 2006).

The LCA methodology connects and quantifies the pathways of substances consumed or released into the environment for each process involved in a product or process system's life cycle. It facilitates the evaluation of environmental effects through midpoint and endpoint impacts, ensuring a comprehensive assessment of what is commonly referred to as an "endpoint." Quantifying endpoints can prove challenging, as a lengthy cause-and-effect chain exists between emissions and their impacts on organisms (JRC-IES, 2010). Midpoints offer a further assessment of the causality chain, taking into account potential impacts surrounding specific environmental mechanisms (Jolliet et al., 2003). Table 1 provides examples of midpoints and endpoints estimated utilising LCA, based on the ReCiPe 2016 LCA methodology.

History of Life Cycle Assessment (LCA)

Early approaches to life cycle thinking can be found in historical literature. For instance, Scottish economist and biologist Patrick Geddes developed a procedure in the 1880s that can be regarded as a precursor to Life Cycle Inventory (LCI), focusing primarily on energy supply, particularly coal. The concept of modern LCA emerged in the 1960s as concerns over environmental degradation and limited resource availability grew. Studies recognised as (partial) LCAs originated in the late 1960s and early 1970s, coinciding with heightened public awareness of resource and energy efficiency, pollution control, and solid waste management issues. (Guinée et al., 2011; Koppfle and Grahl, 2014)

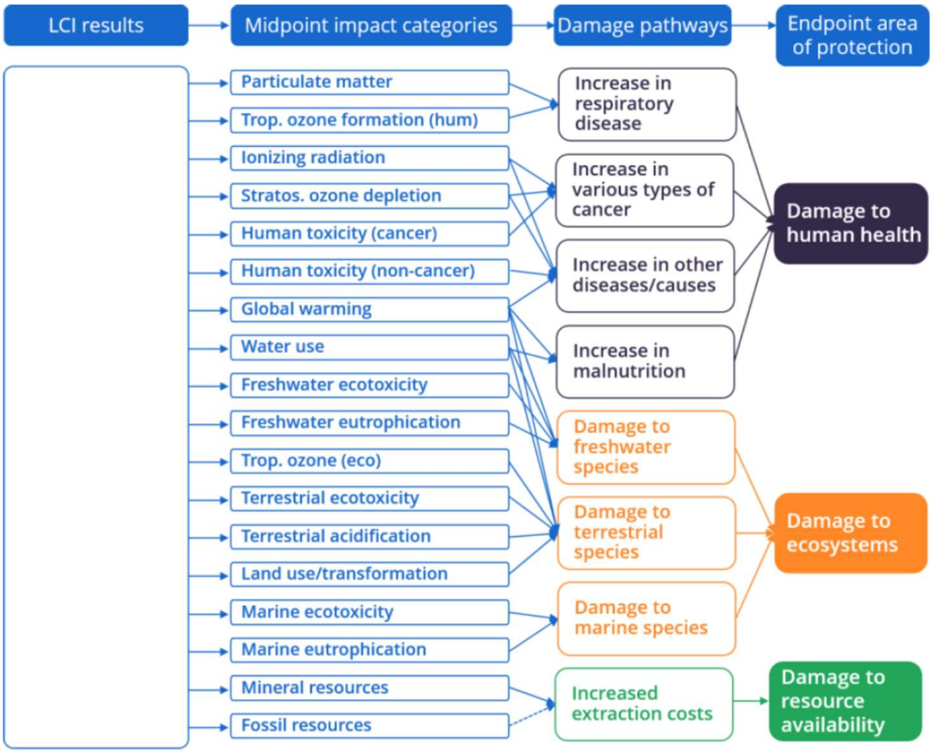

Figure 10: *Impact Categories and Protection Areas in ReCiPe 2016 Method*

Initial methodologies aimed to compare products, primarily concentrating on energy use, a few emissions, and later expanding to include waste generation. In the early stages, terms such as "Resource and Environmental Profile Analysis (REPA)" (Hunt et al., 1992) and "Eco-balances" were used to describe these approaches. Diverse methodologies were utilised and conducted during the 1970s and 1980s without a common theoretical framework or international scientific discourse.

The 1990s witnessed remarkable growth in global scientific and coordination activities, as evidenced by the increasing number of workshops, forums, handbooks, and journals. This period experienced a surge in methodological development, international collaboration, and coordination within the scientific community, with method development increasingly occurring in academic institutions. The Society of Environmental Toxicology and Chemistry (SETAC) played a pivotal role in fostering collaboration between LCA practitioners, users, and scientists to continually refine and harmonise LCA frameworks, terminology, and methodologies. This collaboration led to the SETAC "Code of Practice."

The International Organization for Standardization (ISO) became involved in 1994, formalising the standardisation of methods and procedures, and producing two international standards: ISO 14040 (2006E): 'Environmental management - Life cycle assessment - Principles and framework', and ISO 14044 (2006E): 'Environmental management - Life cycle assessment - Requirements and guidelines.' In 2000, a platform for debate and harmonisation of LCA methods was established, and LCA became part of policy documents and legislation. Several widely recognised life cycle impact assessment methods still in use today emerged during this period.

In 2002, the United Nations Environment Programme (UNEP) and the Society for Environmental Toxicology and Chemistry (SETAC) launched the International Life Cycle Partnership, known as the Life Cycle Initiative (39). The Life Cycle Initiative's primary objective was to incorporate life cycle thinking into practice and enhance supporting tools through improved data and indicators. The current period is characterised by divergent methods, with varying approaches developed regarding system boundaries and allocation methods. Today, LCA is defined as a tool to assess the potential environmental impact and resources used throughout a product's life cycle, from cradle to grave, or end-of-life to grave in relation to solid waste.

LCA in Waste Management

At first glance, employing LCA for waste management issues may appear contradictory to its original purpose and principles, which were designed to assess the environmental impacts of products "from the cradle to the grave." From this perspective, waste management would always be part of a specific product's life cycle, as products become waste at the end of their useful lifespan. However, waste management technologies can also be viewed as a service related to specific environmental impacts of interest. From this standpoint, two emission modelling approaches can be proposed: a process approach and a product approach (Koci and Trecakova, 2011)

Since the beginning of the 21st century, there has been a gradual growth in the application of LCA in the waste management research field (Laurent et al., 2014a). Initially, developed countries in Europe dominated LCA applications due to legislation requirements, but in the last five years, there has been a significant increase in the number of LCA studies performed in underdeveloped and developing countries (Paes et al., 2020). For instance, China has produced the majority of studies in this field, with Iran and Brazil also appearing in the top-ten (Paes et al., 2020). This trend reflects the rising concern for sustainable waste management in these countries as they face the challenge of increasing population, accelerated urbanisation, and rising material consumption (Zhang et al., 2021).

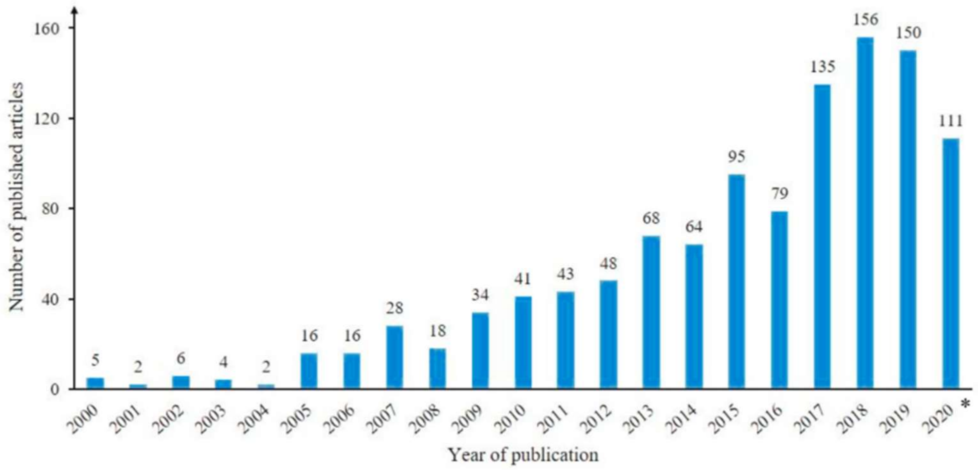

Figure 11: *Number of published articles applying LCA to solid waste management in the 21st century (2020 is an incomplete year which is indicated by "*"(Zhang et al., 2021)).*

According to the reviewed cases, Brazil, China, and India are the top three developing countries in terms of LCA studies, having published over half of the literature reviewed (Paes et al., 2020). These countries are committed to the climate agreement outlined at the Conference of the Parties of the United Nations Framework Convention in Climate Change, and investigations into the environmental impacts of improved MSWM systems have been promoted at the municipal level to meet GHG mitigation targets at the national scale (Paes et al., 2020). Therefore, the evolution of LCA studies in the 21st century has seen a significant shift towards an inclusive approach that considers the life-cycle perspective of waste management systems in both developed and developing countries (Laurent et al., 2014).

LCA Standardisation

Throughout the 1990s, the Society of Environmental Toxicology and Chemistry (SETAC) played a pivotal role in the development of LCA and LCA standards. Under SETAC's guidance, LCA evolved to consider numerous environmental impacts, particularly those related to toxicology. Over the decade, practitioners and researchers from both sides of the Atlantic developed a series of LCA standards to guide best practices. The first standard, ISO 14040, was released in 1997, followed by several others (ISO, 14040, 2006). These were eventually superseded by the combination of ISO 14040 and 14044 (ISO, 14044, 2006) in 2006. The standards maintained the requirement for a consensus document, and several areas remain controversial to this day. These areas include whether it is better to model the world as it is (attributional LCA or as it will change with the increase or decrease of product demand (consequential LCA), and how to allocate the impacts when one process or product flow generates multiple outputs, such as hydrogen and oxygen from electrolysis or virgin and recycled material from a plastic manufacturing process.

In 2002, the United Nations Environmental Program (UNEP) collaborated with SETAC to form the Life Cycle Initiative. The initiative continues to work on issues such as impact assessment method development and facilitating LCA usage by developing countries and Small and Medium-sized Enterprises. At this point in time, LCA studies were expensive and resource-intensive. Companies tended to study one representative product and then create rules of thumb to reduce the impacts on their products (Laurin, 2017)

Building upon the work conducted by the Society for Environmental Toxicology and Chemistry (SETAC), the ISO has further developed and reached an agreement among its global membership on a series of standards: the ISO 14040 series on Life Cycle Assessment

- ISO 14040 Environmental Management – Life Cycle Assessment – Principles and Framework (ISO, 1997).
- ISO 14041 Environmental Management – Life Cycle Assessment – Goal and Scope Definition and Life Cycle Inventory Analysis (ISO, 1998).
- ISO 14042 Environmental Management – Life Cycle Assessment – Life Cycle Impact Assessment (ISO/FDIS, 1999).
- ISO 14043 Environmental Management – Life Cycle Assessment – Life Cycle Interpretation (ISO/FDIS, 1999).

Parts of LCA

The Steps of an LCA

The ISO never aimed to standardise LCA methods in detail. Due to the lack of standard agreement **on interpreting** some ISO requirements, several approaches have been developed throughout the years concerning system boundaries and allocation methods. The LCA framework operates in four phases: goal and scope definition, inventory analysis, impact assessment, and interpretation. The application of an LCA methodology must never be conceived as a straightforward procedure, as in every step, a reevaluation of the previous phases must be made, as shown in **Figure 12**

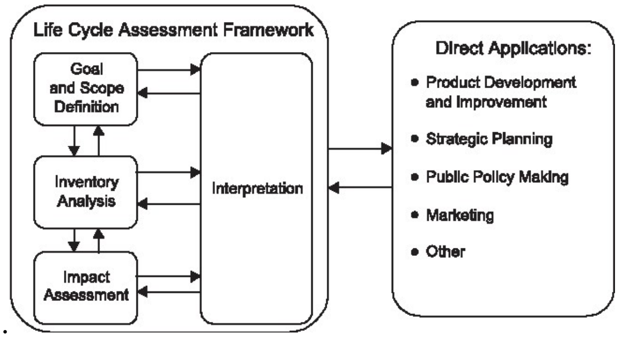

Figure 12: *Framework of LCA modified from the ISO 14040 standard.*

The process begins by identifying the goal or question the practitioner is attempting to answer. The Goal Definition component states the reason for performing a specific study, defines the options to be compared, and outlines the intended use of the results. In the subsequent steps, the study's scope will define the context and methodological framework used and outlined following the goal definition. The following terms will be set:

Selecting the functional unit is the quantitative description of the function or service for which the assessment is performed. It also determines the reference flow of materials that scale data collection. The functional unit is the basis for comparing products or services. The importance of defining the most appropriate Functional Unit cannot be over-emphasised. The functional unit is the cornerstone of an LCA study, providing the reference point to which both inputs and outputs are related and allowing a clear comparison of LCA results.

Setting the boundaries of the monitored system involves deciding which activities and processes belong to the studied system. The System Boundary defines the unit processes included in the system to be modeled. Ideally, the Product System should be modeled so that inputs and outputs at its boundary are elementary flows. Assumptions, simplifications, and cut-off criteria are set with the system at this stage.

There are two approaches to LCA: Attributional and Consequential. Attributional LCA assesses how a product has been produced, evaluating the current situation using historical mass and energy flows and current market trends. This is useful for identifying hotspots and establishing the impacts of today's products. Another LCA concept revolves around the consequence of change. In this method, the practitioner works to understand the consequences of choosing one alternative over another. This methodology is especially important when LCA is used for policymaking.

Life Cycle Inventory Analysis (LCI)

The subsequent step is the inventory analysis. A Life Cycle Inventory Analysis is concerned with data collection and calculation procedures necessary to complete the inventory. This stage consists of accounting for all material and energy inputs and outputs over the product or service's entire life cycle. The operational steps are presented in **Figure 13**

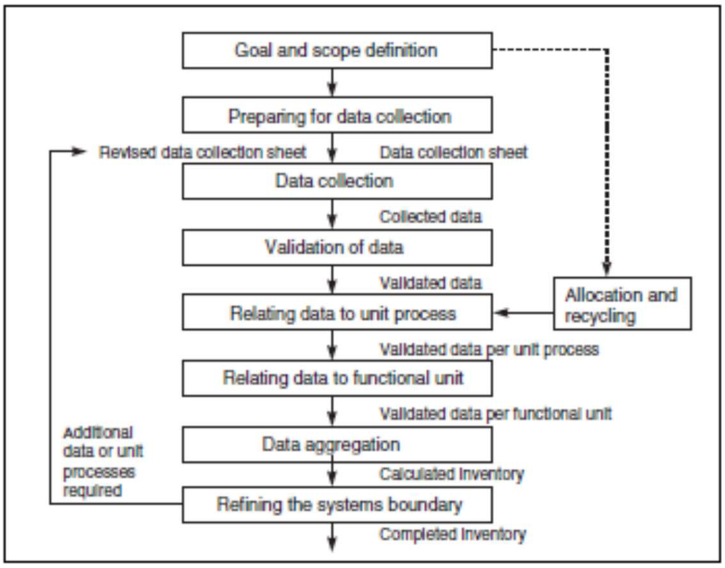

Figure 13 *Procedures for life cycle analysis (ISO 14041)*

The procedure entails describing the Life Cycle as a series of steps and then calculating the inputs and outputs for each of these steps (see **Figure 13**). This amounts to constructing a materials and energy balance for each step in the Life Cycle. The analysis of all inputs and outputs for each stage in the Life Cycle can then be combined to provide the overall **Life Cycle Inventory.**

Data quality requirements

There are two main categories of data used:

1. Specific data for production, distribution and waste management.
2. Generic data for energy production, raw material extraction and transportation

Data quality requirements should address time-related, geographical, and technology issues; the precision, completeness, and representativeness of the data; the consistency and reproducibility of the methods used throughout the LCA; the sources of the data and their representativeness, and the uncertainty of the information..

Sensitivity and uncertainty analysis

In accordance with ISO 14043, Life Cycle Inventories (LCIs) must undergo sensitivity and uncertainty analyses. The data and results should not be utilised without comprehending their quality and limitations. These processes also acknowledge that LCI encompasses data uncertainties and value judgments. Sensitivity analysis provides insights into the robustness of the LCI results and identifies areas where more or more precise data is needed to enhance the inventory. It assists in determining whether any assumptions made, such as those concerning missing data, significantly influence the LCI's final outcome and, if so, which assumptions have the most substantial impact. Uncertainty analysis is also crucial. Generic emission data might involve a broad range of emissions levels from one or more unit operations or may have changed since the emissions were measured. This introduces uncertainty into virtually every number within the inventory, necessitating consideration.

Life Cycle Impact Assessment (LCIA)

The LCIA phase of an LCA study offers a holistic perspective on environmental and resource issues for product or service systems. The LCIA phase aims to evaluate the product system from an environmental standpoint, employing category indicators derived from LCI results. To accomplish this, LCIA assigns LCI outcomes to specific, chosen impact categories (an impact category is utilised to group particular LCI results related to a specific environmental issue). Suitable indicators are chosen for each impact category, and a characterisation model calculates indicator results. The LCIA profile, comprising the collection of indicator results, delivers an environmental context for the emissions and resource usage associated with the product or service system. The LCIA phase also supplies information for the interpretation phase.

The classification stage necessitates identifying inventory data pertinent to each impact category and allocating the appropriate LCI results to each category. Data may belong to multiple categories; for example, NOx has both global warming and acidifying effects.

Selection of Impact Categories

Characterisation aims to establish a foundation for aggregating inventory results into an indicator for each category. Each impact category requires a specific model to transform inventory results into indicators. During the characterisation or modeling stage, calculations are made to evaluate the relative significance of each contributor to the overall impact of the system or operation under study by converting these to a common indicator. For instance, in the case of global warming, the most prevalent indicator used is Global Warming Potential (GWP) in CO2 equivalents. Essentially, there are two steps in the calculation. Each greenhouse gas is first converted into carbon dioxide equivalents based on a specific characterisation factor. The individual carbon dioxide equivalents are then combined to form a total indicator.

Optional steps

LCIA encompasses several mandatory elements that convert LCI results to indicator results. Additionally, there are optional elements for normalising, grouping, or weighting indicator results and data quality analysis techniques.

Normalisation

Normalisation, if conducted, involves relating the characterised data to a broader dataset or context, such as comparing SOx emissions to a country's total SOx emissions. Although normalisation can offer insights, it should be approached with caution, as results may vary significantly depending on the datasets utilised. Frequently, normalisation is excluded from LCA studies.

Weighting

Weighting entails converting indicator results from different impact categories into scores using numerical factors based on values. Weighting may also involve aggregating the weighted results into an overall score. This stage of an LCA is the most subjective, as it relies on value judgments rather than scientific principles..

Life Cycle Interpretation

Life Cycle Interpretation is a systematic approach to identify, qualify, verify, and evaluate information derived from the results of the Life Cycle Inventory (LCI) analysis and/or Life Cycle Impact Assessment (LCIA) of a product system. The interpretation aims to satisfy the application requirements as outlined in the study's goal and scope. The Life Cycle Interpretation phase of an LCA encompasses three elements

Identifying significant issues based on the LCI and LCIA phases of the LCA.

1. Evaluating significant issues through completeness, sensitivity, and consistency checks.
2. Drawing conclusions, making recommendations, and reporting significant issues.

Identification of Significant Issues

The purpose of this element is to structure the results from the LCI or LCIA phases in a manner that facilitates the identification of significant issues. This process should include any implications arising from the specific method employed and any assumptions made. Allocation rules, cut-off decisions, choice of indicators, and characterisation methods must all be addressed.

The evaluation element's objectives are to establish and enhance confidence and reliability in the study results. The evaluation results should be presented in a manner that provides the reader with a clear and understandable view of the study's outcome. To achieve this, completeness checks (ensuring all relevant information for interpretation is available and complete), sensitivity checks (assessing the reliability of the results by examining the uncertainty of the significant issues affecting the conclusion), and consistency checks (determining if the assumptions, methods, and data are consistent with the goal and scope) should be conducted. Conclusions should be drawn interactively with the other Life Cycle Interpretation phase elements.

In conclusion, over the past 15 years, Life Cycle Assessments (LCAs) have emerged as a powerful and versatile tool in waste management, contributing significantly to addressing the complex challenges associated with sustainability, climate change, and the transition towards a circular economy. These assessments have facilitated a deeper understanding and optimisation of waste management systems across a wide range of sectors, as evidenced by their successful application in six key areas: 1) comprehending the intricacies of existing waste management systems; 2) enhancing the performance and efficiency of current waste management systems; 3) conducting comprehensive comparisons of alternative technologies and their performance; 4) fostering innovation and the development of prospective technologies that hold promise for the future; 5) informing policy development and strategic planning at various levels of governance; and 6) refining and standardising reporting processes in waste management (Christensen et al., 2020)

The ongoing advancements in LCA methodology, coupled with the increasing availability of high-quality data and the integration of emerging technologies, will further enable waste management practitioners, policymakers, and stakeholders to make more informed decisions, thereby promoting sustainable practices and mitigating the adverse environmental impacts of waste generation and disposal. Additionally, as interdisciplinary collaboration continues to grow, LCAs will play a crucial role in bridging the gaps between scientific research, technological innovation, and policy development, facilitating a more cohesive and effective approach to addressing the pressing environmental challenges faced by today's global society.

LCA Characterization Methodologies

The ever-increasing global waste generation and the need to address environmental, social, and economic aspects of waste management have led to developing and utilising various LCA methodologies. This chapter aims to review some of the most widely used LCA methodologies in waste management, including ReCiPe, International Reference Life Cycle Data System (ILCD), and the CML method as their history, procedures, and differences.(Laurent et al., 2014b)

LCA methodologies have emerged as valuable tools to assess the environmental performance of waste management systems, providing a quantitative and systematic approach to identify areas for improvement and inform policy decisions. Depending on the philosophy and the goals of each research, the accounting and the steps to translate the collected data into environmental impacts led to the creation and establishment of several methodologies. In simple terms, the characterisation methodology converts the inventory data into impact categories (Mulya et al., 2022).

In the 1960s and 1970s, the early methodologies were based on a simple calculation input-output model that quantified the environmental impacts of products and processes by tracking the flow of materials and energy (Guinée et al., 2011) (Wernet et al., 2016). As waste management became a growing concern, LCA methodologies were adapted to evaluate the ecological consequences of different waste management strategies (Klöpffer, 2014).

During the 1980s, LCA methodologies began incorporating more comprehensive life cycle inventories, including more detailed data on material and energy flows, waste emissions, and environmental releases, allowing for a more thorough understanding of the environmental impacts of products and processes. One example of an LCA methodology that emerged in the 1980s is the CML method, developed by the Institute of Environmental Sciences at Leiden University in the Netherlands. The CML method was one of the first LCA methods to incorporate a comprehensive set of impact categories, including climate change, acidification, eutrophication, and ozone depletion.

In the 1990s and 2000s, there was a shift towards more standardised LCA methodologies, which led to the development of international standards such as ISO 14040 and ISO 14044. These standards provide guidelines for conducting LCA studies and ensure that studies are consistent and transparent.(JRC-IES, 2010)

Newer LCA methodologies have also incorporated more advanced modelling techniques, such as input-output analysis, hybrid LCA, and social LCA, to account for the environmental, social, and economic impacts of products and processes throughout their life cycles. (Klöpffer, 2014)

Methods

Various LCA methodologies are available for waste management applications, each with unique features, strengths, and limitations. The most widely used LCA methodologies in waste management include (Chevalier et al., 2011; Owsianiak et al., 2014):

The **ReCiPe** methodology is a commonly used LCA tool that comprehensively evaluates a product's or process's environmental impacts across multiple impact categories, such as climate change, acidification, and eutrophication. It provides a detailed understanding of ecological impacts by quantifying environmental impacts in two distinct groups: midpoints and endpoints. The midpoint group includes 17 ecological impacts, such as global warming, acidification, and ozone depletion, represented by relevant indicators. The endpoint group translates the environmental impacts into issues of concern, typically reflecting damage to human health, ecosystem quality, and resources. The European Commission proposed ReCiPe, frequently employed in Europe to inform waste management policies. The methodology considers a wide range of impacts, including resource depletion, greenhouse gas emissions, and other environmental factors, mainly using mass-based units. Although the method offers a comprehensive and holistic approach to waste management, it necessitates detailed data and substantial resources to implement effectively (Huijbregts et al., 2016; Oliveira et al., 2017; Ripa et al., 2017)

CLM (Cumulative Energy Demand-based Life Cycle Management) is a LCA methodology used to evaluate the energy consumption and environmental impact of products or services. It accounts for the total energy consumption and greenhouse gas emissions associated with a product throughout its life cycle, from raw material extraction to disposal. The methodology was proposed by the Japan Environmental Management Association for Industry and is widely used in Japan. The impacts accounted for in CLM include greenhouse gas emissions, air and water pollution, and resource depletion. The units used in CLM are typically energy-based, and the methodology is advantageous for its ability to consider the entire life cycle of a product.

However, its disadvantages include the need for detailed data and significant resources to implement effectively.

ILCD (International Reference Life Cycle Data System) is a standardised LCA methodology developed by the European Commission to ensure consistency and comparability of LCAs across different sectors and regions. It provides a common framework for data collection, modelling, and reporting in LCA studies. The methodology accounts for various environmental impacts, including climate change, ozone depletion, and ecosystem quality. The units used in ILCD vary depending on the impact category assessed, and the methodology is advantageous for its ability to provide a standardised approach to LCA. However, its disadvantages include the need for expert knowledge and the potential for oversimplification of complex environmental systems. (JRCh Centre -- Institute for Environment and Sustainability, 2010; Tobergte and Curtis, 2013)

Eco indicator is a methodology developed in the Netherlands that accounts for various environmental impacts, including human health, ecosystem quality, and resource depletion. It uses a set of impact categories and characterisation factors to quantify the environmental impacts associated with a product or service. The units used in Eco indicator are typically damage-based, and the methodology is advantageous for considering a wide range of environmental impacts. However, its disadvantages include the potential for oversimplification of complex environmental systems and the need for expert knowledge to implement them effectively.

IMPACT is a methodology developed by the US Environmental Protection Agency to quantify the environmental impacts associated with products or services. It accounts for many environmental impacts, including climate change, ozone depletion, and acidification. The units used in IMPACT vary depending on the impact category assessed, and the methodology is advantageous for its ability to consider a wide range of environmental impacts. However, its disadvantages include the need for expert knowledge and the potential for oversimplification of complex environmental systems.

EPS (Eco-profiles and Sustainability) is a standardised LCA methodology developed by the International Organization for Standardization to provide a consistent framework for reporting and comparing the environmental impacts of products or services. It accounts for many environmental impacts, including climate change, ozone depletion, and acidification. The units used in EPS vary depending on the impact category assessed, and the methodology is advantageous for its ability to provide a standardised approach to LCA. However, its disadvantages include the potential for oversimplification of complex environmental systems and the need for expert knowledge to implement them effectively.

TRACI (Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts) is a methodology developed by the US Environmental Protection Agency to assess the potential human health and ecological impacts associated with chemicals and other environmental stressors. It accounts for various impact categories, including human health effects, ecotoxicity, and ecosystem quality. The units used in TRACI vary depending on the impact category assessed, and the methodology is advantageous for its ability to consider the potential impacts of specific chemicals or environmental stressors. However, its disadvantages include the need for detailed data and significant resources to implement effectively.

Each LCA methodology has its strengths and limitations, depending on the context of its application. For instance, ReCiPe is useful in identifying hotspots in the waste management system and can be customised to reflect regional or national environmental concerns. On the other hand, the ILCD offers guidance on data quality and consistency, ensuring the reliability of the LCA results (Klöpffer, 2014). The CML method is widely used and offers a user-friendly interface, making it accessible to a broader range of stakeholders. However, these methodologies may face limitations regarding data availability, methodological consistency, transparency, and applicability to different waste management contexts (Huijbregts et al., 2016).

Comparative analysis of these LCA methodologies in waste management reveals differences in their impact assessment approaches, which can have implications for waste management decisions. For example, ReCiPe emphasises the impact categories related to human health, whereas the ILCD emphasises those related to natural resources. These differences can influence the prioritisation of waste management strategies and affect the overall environmental performance of the system.

One significant difference in how different LCA methodologies account for biogenic carbon is whether or not to consider it as a net carbon source or sink. Some methodologies, such as the Carbon Trust's PAS 2050, treat biogenic carbon as a net carbon source, meaning that they consider the emissions associated with the release of carbon from biogenic materials as equivalent to emissions from fossil fuels.

Other methodologies, such as the Intergovernmental Panel on Climate Change (IPCC) Guidelines for National Greenhouse Gas Inventories, treat biogenic carbon as a net carbon sink, meaning that they account for the carbon sequestration associated with the growth of biogenic materials, such as crops and forests.

Another difference in how different LCA methodologies account for biogenic carbon is the choice of carbon accounting method. Some methodologies use a stock-based approach, which accounts for the net change in carbon stocks over time, while others use a flux-based approach, which accounts for the flow of carbon in and out of a system at a given time.

Overall, the accounting of biogenic carbon in LCA is a complex and evolving area, and the choice of methodology can have significant implications for the results of an LCA study. As a result, it is important for LCA practitioners to carefully consider the assumptions and choices underlying their chosen methodology when accounting for biogenic carbon.

As waste management systems continue to evolve and new challenges emerge, LCA methodologies must adapt and improve to remain effective tools for environmental assessment. Future developments in LCA methodologies for waste management could include improvements in data availability, methodological harmonisation, and the incorporation of emerging waste management technologies and strategies. For example, the Water Footprint is an LCA methodology that evaluates the amount of water used and polluted throughout the entire life cycle of a product or process (ISO, 2006). This methodology can be used to assess the water footprint of different waste management strategies, such as recycling and incineration, and identify opportunities for water conservation.

In conclusion, LCA methodologies are valuable tools for waste management practitioners and policymakers in assessing the environmental performance of waste management systems. ReCiPe, ILCD, and CML are the most widely used LCA methodologies in waste management, each with unique features, strengths, and limitations. By understanding the history, methods, and differences among these methodologies, stakeholders can make informed decisions about their application and contribute to developing more sustainable waste management systems. Ongoing improvements and innovations in LCA methodologies will be critical to addressing waste management's environmental, social, and economic aspects and supporting the transition towards a circular economy.

Aspect	CLM	IMPACT	RECIPE	ILCD	EcoIndicator
Origin/ Development	Developed by McDonough and Braungart	Developed by a consortium of European researchers	Developed by the Swiss Federal Laboratories for Materials Science and Technology (Empa)	Developed by the European Commission	Developed by Pre Consultants
Scope	Focused on product design and optimisation	Provides a broad set of impact categories	Primarily used for industrial processes	Offers a comprehensive LCA framework for various applications	Designed for assessing product sustainability
Goal	Promotes sustainable product design and production	Assess environmental impacts in various categories	Quantify environmental impacts of processes	Facilitate consistent and comparable LCA studies in Europe	Assess ecological sustainability of products
Life Stages Analysed	Emphasises product life cycle stages	Typically analyses the entire life cycle of products	Mainly focuses on the manufacturing phase	Provides flexibility to select specific life cycle stages	Covers entire life cycle
Impact Categories	Emphasises material health and reusability	Covers a wide range of environmental impact categories	Limited set of impact categories	Provides a wide range of impact categories and subcategories	Considers a range of impact categories
Data Requirements	Requires detailed product and material data	Requires comprehensive life cycle inventory data	Requires detailed process data	Provides guidance on data quality and collection	Requires life cycle inventory data
Application	Primarily used for product design and certification	Commonly used for environmental impact assessment	Often applied to assess industrial processes	Used for policy support, research, and comparative LCAs	Used for product assessment and design
Transparency	Encourages transparency in material choices	Transparent and well-documented methodology	Provides transparency in process modeling	Emphasises transparency and harmonisation of LCA studies	Emphasises transparency in methodology
Geographic Focus	Widely applicable but often used in the US	Widely used in Europe and adaptable to other regions	Mainly used in Switzerland and Europe	Developed for use within the European context	Globally applicable
Complexity	Can be complex due to focus on product design	Moderate complexity due to comprehensive impact assessment	Moderate complexity, especially in data collection	Comprehensive and adaptable, potentially complex	Moderate complexity
Regulatory Alignment	Not specifically aligned with regulations	Used to support regulatory compliance and policy decisions	Used in regulatory context in Switzerland	Developed to align with EU policies and regulations	Used in some regulatory contexts

Inventories

Several LCA databases are currently available to researchers and industry practitioners. One widely used database is the ecoinvent database, which includes comprehensive and up-to-date data on global supply chains and processes across a range of industries. The ecoinvent database has been used in numerous studies to evaluate the environmental impacts of various products and processes, including biofuels (Frischknecht et al., 2015), food production (Basset-Mens and van der Werf, 2005), and transportation (Lützkendorf and Lorenz, 2013).

Another LCA database frequently used is the Global Feed LCA Institute database, which provides detailed data on the environmental impacts of feed production and use in livestock systems (Hagemann et al., 2018). The OpenLCA database is another freely available database that includes data on a wide range of products and processes and can be used with a variety of LCA software tools (Wernet et al., 2016).

Overall, the use of LCA databases is essential for advancing our understanding of the environmental impacts of products and processes, and for guiding decision-making towards more sustainable production and consumption practices.

Impacts categories

In LCA, results are often presented as impacts. These impacts represent the environmental effects of a product or process, and are quantified using midpoint or endpoint categories. Midpoint categories are used to measure the potential effects of a product or process on specific environmental mechanisms, such as global warming potential (GWP) or acidification potential (AP). Endpoint categories, on the other hand, represent the ultimate impact on human health or the environment, such as the number of cases of respiratory disease or the loss of biodiversity. By presenting results in terms of environmental impacts, LCA provides a comprehensive understanding of the potential consequences of a product or process, and can inform decisions about how to mitigate negative effects (Mulya et al., 2022)..

Midpoint impacts are a key element of LCA, used to quantify the environmental impacts of products, services or processes. These impacts are intermediate results that measure the potential harm caused to the environment and human health through a chain of cause-effect relationships. In the LCA literature, global warming potential (GWP) is the most frequently studied midpoint impact, followed by acidification potential (AP), eutrophication potential (EP), and human toxicity potential (HTP).

According to (Bare et al., 2000), these four midpoint impacts are consistently studied in at least half of the 240 articles reviewed. Other midpoint impacts, such as ecotoxicity potential (ETP) and ozone depletion potential (ODP), have seen minor increases in inclusion, while abiotic depletion-fuels (ADPF) and particulate matter formation (PMF) have experienced a tremendous increase in usage. Overall, the choice of midpoint impacts is dependent on the goal of the LCA, however, GWP, AP, EP, and HTP are the most widely used due to their applicability to most scenarios (Mulya et al., 2022).

Figure 14 presents the appearance of the most poular midpoint impacts studied in from 2009 till 20220 base on the reviws of Laurent et al. (2014) and Mulya et al. (2020). The Midpoint categories of 240 studies selected for review (GWP = Global Warming Potential; AP = Acidification of Soil and Water; EP = Eutrophication; HTP = Human Toxicity; POP = Photochemical Ozone Creation; ETP = Ecotoxicity; ADP = Depletion of Abiotic Resources; ODP = Ozone Layer Depletion; CED = Cumulative Energy Demand; ADPF = Depletion of Abiotic Resources – Fossil Fuels; PMF = Particulate Matter Formation)

Endpoint impacts represent the ultimate consequences of a product or process on human health, ecosystems, and resources. These impacts are categorised into three independent categories: "damage to human health", "damage to ecosystem", and "damage to resources". The first category measures impacts on human health, such as carcinogenic effects and respiratory organics. In contrast, the second category observes species loss due to environmental impacts like global warming and acidification.

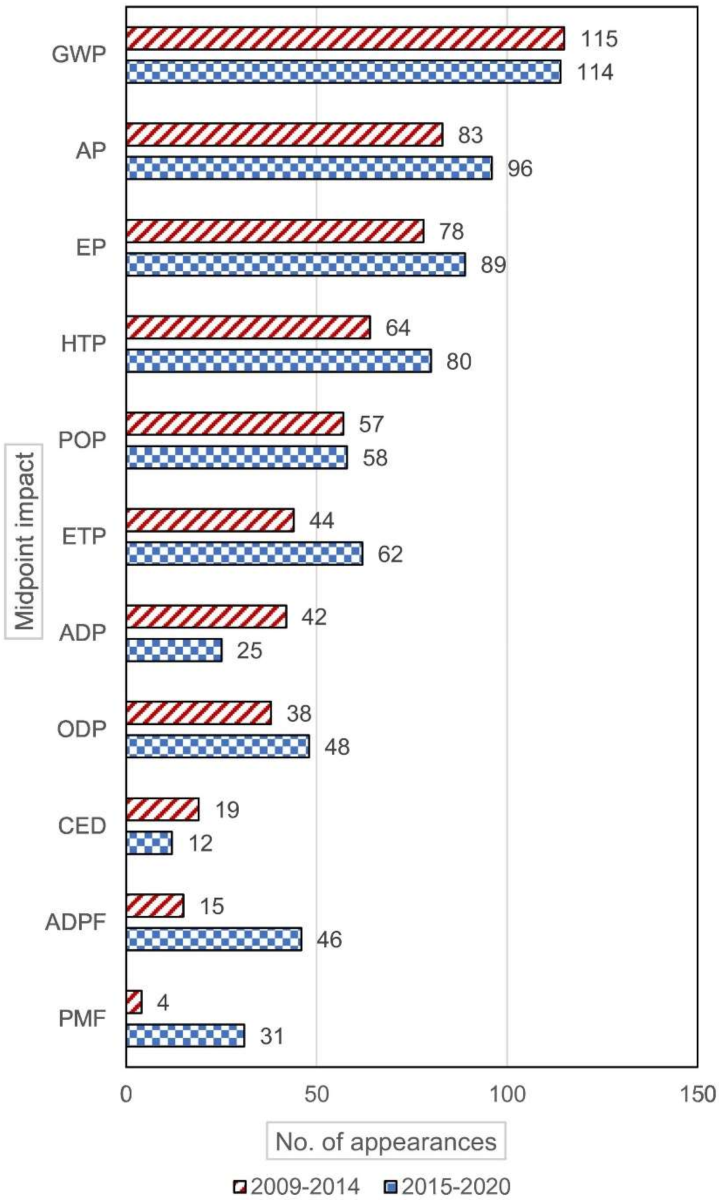

Figure 14 *Number of most popular mid point impacts appearance in LCA studies for waste management during two search periods: 2009- 2014(Laurent et al., 2014) and 2015-2020 (Mulya et al., 2022)*

The third category, "damage to resources", focuses on resource-related impacts centred around depletion or scarcity, such as abiotic depletion and fossil resource scarcity. While endpoints provide a general overview of the final assessment outcomes, they may overlook specific details and lead to miscommunication between researchers. Additionally, there may be missing pathways in endpoint modelling that can cause errors in calculations and final endpoint results. These limitations have led some researchers to focus on midpoint analysis instead, which involves fewer uncertainties and gaps that need to be addressed (Mulya et al., 2022).

LCA Tools

The history of LCA tools has evolved significantly over the years, starting from simple calculations and gradually incorporating more advanced technologies, including modern software and artificial intelligence (AI).

LCA was inception during the 1960s and 1970s in response to escalating concerns about resource scarcity and environmental degradation (Hauschild et al., 2013). Initial LCA practitioners manually performed elementary calculations and analyses, concentrating on energy consumption and waste generation of particular products or processes (BSI, 2006).

Spreadsheets and basic software (the 1980s-1990s): As personal computers became more widespread, LCA practitioners started using spreadsheets to store, organise, and analyse life cycle inventory (LCI) data (ISO, 2006b). Basic LCA software tools were also developed during this period, allowing for more efficient and standardised calculations (Weidema and Wesnæs, 1996).

Specialised LCA software (the late 1990s-2000s): As the LCA methodology matured and gained popularity, specialised LCA software tools such as SimaPro, GaBi, and Umberto were developed (Klöpffer & Grahl, 2014). These tools offered more advanced features, comprehensive databases, and support for various LCA methodologies, enabling more complex and accurate analyses (Hauschild et al., 2013).

Waste management-focused tools (the 2000s): As the need for sustainable waste management practices grew, LCA tools specifically designed for waste management assessments emerged. EASETECH is one example of a tool developed to evaluate the environmental performance of waste management systems and technologies.

Integration of AI and machine learning (2010s-present): The ongoing advancements in AI and machine learning have started influencing the LCA domain. These technologies can help improve the accuracy and efficiency of LCA tools by automating data collection, identifying patterns and trends, and optimising waste management strategies (Huijbregts et al., 2017).

The most popular LCA tools include (Laurent et al., 2014b; Vea et al., 2018) (Gentil et al., 2010) (EPLCA [i]):

- **SimaPro:** Developed by PRé Consultants, SimaPro is a generic LCA software developed in the early 1990s (www.simapro.com). It is based on the ISO 14040 and ISO 14044 standards. It allows users to assess the environmental impacts of products, processes, and services throughout their entire life cycle. SimaPro uses a comprehensive database to calculate impact categories such as climate change, acidification, and eutrophication. The software is widely used in academic research, industry, and government and has been cited in numerous studies. Advantages of SimaPro include its extensive database, user-friendly interface, and ability to generate reports and graphs. Disadvantages include its high cost and the need for specialised training to use it effectively (Rosenbaum et al., 2018).

- **UBERTO** is a LCA software developed by Quantis International. It is designed to assist companies and organisations in assessing the environmental impacts of their products and processes. UBERTO employs a comprehensive database that covers a wide range of impact categories, including climate change, acidification, and eutrophication. The software allows users to model and analyse the life cycle of a product, from the extraction of raw materials to the end of life. UBERTO has been used in several industries, including food, consumer goods, and packaging. Advantages of UBERTO include its user-friendly interface, ability to handle complex data, and ability to generate customisable reports. Disadvantages include its high cost and the need for specialised training to use it effectively.
- **EASETECH** (Environmental Assessment System for Environmental Technologies) is an LCA software tool developed by the Technical University of Denmark (DTU). It is specifically designed for conducting environmental assessments of waste management systems and other environmental technologies. EASETECH allows users to model and evaluate waste management systems' environmental impacts using life cycle inventory (LCI) data and different LCA methodologies. SimaPro is a leading LCA software developed in the early 1990s. It is based on the ISO 14040 and ISO 14044 standards and allows users to assess the environmental impacts of products, processes, and services throughout their entire life cycle. SimaPro uses a comprehensive database to calculate impact categories such as climate change, acidification, and eutrophication. The software is widely used in academic research, industry, and government, and has been cited in numerous studies. Advantages of SimaPro include its extensive database, user-friendly interface,

and ability to generate reports and graphs. Disadvantages include its high cost and the need for specialised training to use it effectively.

- **GaBi:** is a life cycle assessment software developed by the German Federal Environment Agency in 1995. It is based on the ISO 14040 and ISO 14044 standards and is widely used in academic research and industry. Gabi allows users to assess the environmental impacts of products, processes, and services throughout their entire life cycle. The software employs a comprehensive database that covers a wide range of impact categories, including climate change, acidification, and eutrophication. Gabi has been used to assess the environmental impacts of a wide range of products, including vehicles, buildings, and consumer goods. Advantages of Gabi include its extensive database, its ability to integrate with other software, and its ability to generate reports and graphs. Disadvantages include its high cost and the need for specialised training to use it effectively.

These popular LCA tools are widely used in various sectors to evaluate the environmental impacts of products, processes, and systems, including waste management. The choice of LCA tool depends on factors such as user requirements, available resources, and the specific context of the study (Weidema and Wesnæs, 1996).

The future of LCA tools is anticipated to involve greater integration of AI, machine learning, and other advanced technologies, allowing for more sophisticated analyses, improved data quality, and enhanced decision-making capabilities (Huijbregts et al., 2016). Additionally, developing cloud-based platforms and mobile applications could make LCA tools more accessible and user-friendly, promoting widespread adoption and facilitating collaboration between stakeholders in waste management and other industries.

Determining a tool shootable for waste management assessment depends on specific requirements, objectives, and preferences. Each tool has its strengths and weaknesses, and the ideal choice will vary depending on factors such as ease of use, availability of data, level of detail, and adaptability to specific waste management scenarios. EASETECH: Explicitly developed for waste management systems, this tool focuses on environmental assessments of waste management strategies and technologies, considering the following factors (Mulya et al., 2022):

A wide range of (non) commercial LCA models is available for environmental assessment SimaPro 2019; Thinkstep Gabi 2019; TEAM 2019; Umberto NXT LCA

2019; for a more complete list, see EPLCA 2019 (Lodato et al., 2021)). When selecting an LCA tool for waste management, it is crucial to consider factors such as:

- Compatibility with preferred LCA methodology or methodologies (BSI, 2006)
- Availability of waste management-specific LCI data
- Ease of use and adaptability to specific waste management scenarios
- Availability of support and documentation to assist with tool usage
- Budget constraints, as some LCA tools may have licensing fees

It is advisable to review each tool's features, compare them against specific needs, and possibly test the tools using trial versions or case studies before making a decision (Huijbregts et al., 2016). Consulting with colleagues or experts in the field who have experience using different LCA tools for waste management assessments may also be helpful (Weidema and Wesnæs, 1996).

EASETECH (Environmental Assessment System for Environmental Technologies) is an LCA software tool developed by the Technical University of Denmark (DTU). It is specifically designed for conducting environmental assessments of waste management systems and other environmental technologies. EASETECH allows users to model and evaluate the environmental impacts of waste management systems using life cycle inventory (LCI) data and different LCA methodologies. The software is widely used for assessing waste management strategies, including waste prevention, recycling, composting, incineration, and landfilling. It enables users to analyse various waste types, such as municipal solid waste, hazardous waste, and specific waste streams like packaging materials or electronic waste.

- it has been applied in various academic and industrial contexts to assess and compare waste management systems' environmental performance, inform policy development, and support decision-making (Chazirakis et al., 2022; Clavreul et al., 2014; Delre et al., 2019; Jensen et al., 2016; Zhao et al., 2015). The software's focus on waste management systems makes it a valuable tool for researchers and practitioners in the waste management sector.
- The tool can provide valuable insights for our research on waste management for several reasons listed below:
- It is designed specifically for waste management applications, ensuring that the tool is tailored to the unique requirements and challenges of the sector. This

focus allows for a more accurate and relevant analysis of waste management systems.

- It incorporates a vast database of waste management technologies, processes, and emissions, enabling you to model various waste management scenarios accurately. This extensive database will allow you to assess the environmental impacts of different waste management strategies and identify the most sustainable solutions.
- It offers a flexible and customisable framework that can be adapted to specific waste management contexts. It can input local data and customise parameters to reflect the region's unique characteristics, leading to more accurate and context-specific results.
- It enables evaluating waste management systems using multiple environmental impacts categories, such as climate change, human toxicity, and resource depletion. This comprehensive assessment helps identify potential trade-offs and synergies between different waste management strategies, facilitating more informed decision-making.
- It allows the practitioner to model and compare various waste management scenarios, providing insights into the potential environmental implications of different strategies. This feature is valuable for exploring the effects of varying waste management policies, technologies, and infrastructure changes in the Chania region.
- It is a transparent and well-documented methodology that facilitates a more robust and reliable LCA, ensuring your research findings are credible and defensible.

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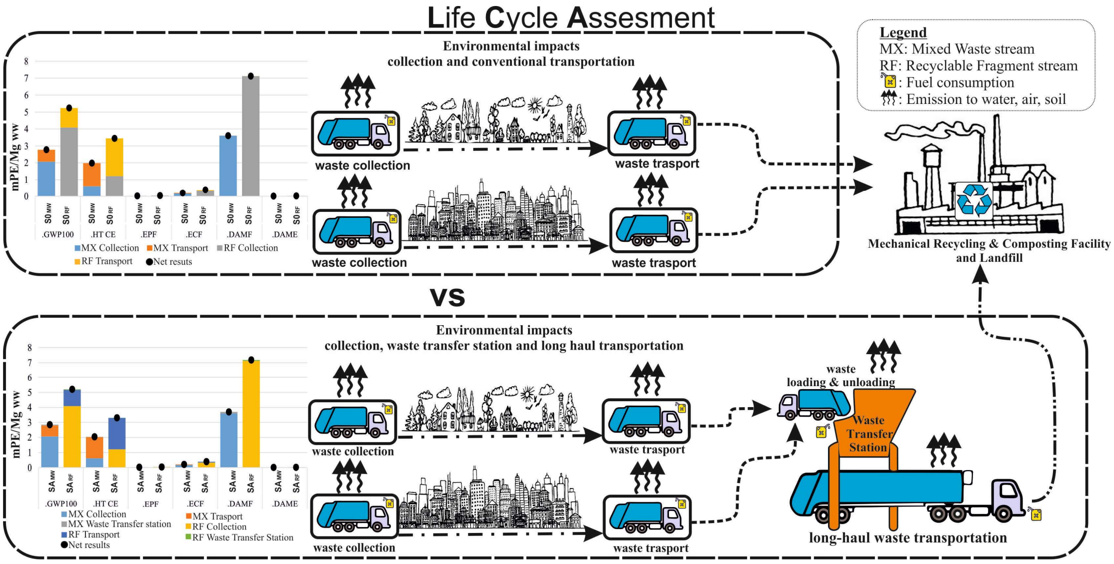
3. Chapter

Positioning transfer stations for waste collection and transport using LCA modelling.

Abstract

The current study aims to evaluate the environmental impacts associated with waste collection and transport practices while assessing the positioning of waste transfer stations (WTS) in the context of fuel consumption and environmental footprints for two main waste streams, Mixed Waste (MW) and Recyclable Fragment (RF). The life cycle assessment (LCA) study uses real data from waste managers, such as weight records, total monthly fuel consumption, and total distance travelled by the collection vehicles. Five scenarios (one for conventional direct waste collection and transport and four with positioning of WTS in various numbers and locations in the area) are explored assessing the environmental impacts and process efficiency of a typical waste collection system in Greece (Chania prefecture). The results show that a significant portion (24-30%) of waste collection and transport emissions is linked to waste transport to the disposal locations. In contrast, introducing WTS can improve the environmental profile of the total process for all impact categories and cumulative fuel consumption. Careful planning based on geographical and population data is critical which can lead to environmental savings, in this study up to 29% for recycling fragment. The advantages become more pronounced when the distance between the Waste Transfer Stations (WTS) and the final disposal or treatment waste facilities surpasses the breakeven point. Additionally, the standard deviation of net results can serve as a reliable estimator of the efficiency of the collection and transport processes.

Graphical Abstract



1. Methodology

The LCA is carried out by applying the standards of ISO 14040 and 14044. The system boundary includes the waste collection from the kerbside containers and transport to the MRCF-landfill of the prefecture of Chania for the two main municipal solid waste streams, mixed waste (MW) and recyclable fragment (RF). The functional unit is 1 Mg of ww mixed waste or recyclable materials collected and transported

1.1. Study area and waste management

The Chania prefecture covers an area of 2,376 km^2. Predominantly mountainous to the south, the area is bordered by the sea on three sides and neighbored by the prefecture of Rethymno to the east. As of 2021, Chania has a population of 156,706 inhabitants (El.Stat, 2021). The prefecture is divided into six municipalities. Each municipality consists of several local districts except the island of Gavdos and the municipality of Sfakia, which, due to their individual characteristics, have one local district each. In Table 1, each municipality and local district are presented. There are also presented the corresponding permanent population, waste production for mixed and recyclables and distances from the geographical center of each local district to the treatment facility – landfill. Most of the population resides in the northern plains of the prefecture, with population density fluctuating based on the season and tourist activity, which can double the population during the summer months. Due to the mountainous morphology of the southern terrain of the prefecture, the road network is more extended in the northern part of the prefecture, detailed maps of the municipalities, local districts, morphology and road network are presented in Figure S1 in the supplementary information. The waste production follows seasonal trend, peaking in the high season and diminishing in winter. The municipalities use a kerbside collection system with 1,100 L color-coded collection containers for the two primary waste streams: Mixed Waste (MW) (green containers) and Recyclables Fragment (RF), like packaging plastic, paper, and metals (blue containers). Over 65% of the permanent population resides in the municipality of Chania. According to the territorial typology published by Eurostat (2018), the area covered by the city of Chania is categorised as urban, while the remaining prefecture is considered rural. This definition is also applied to the waste collection routes for both MW and RF. All waste collected is transferred to the Mechanical Recycling and Composting Facility and landfill (MRCF-landfill) in the Mechanical Recycling and Composting Facility and landfill (MRCF-landfill) in the Akrotiri area in the northeast of the region.

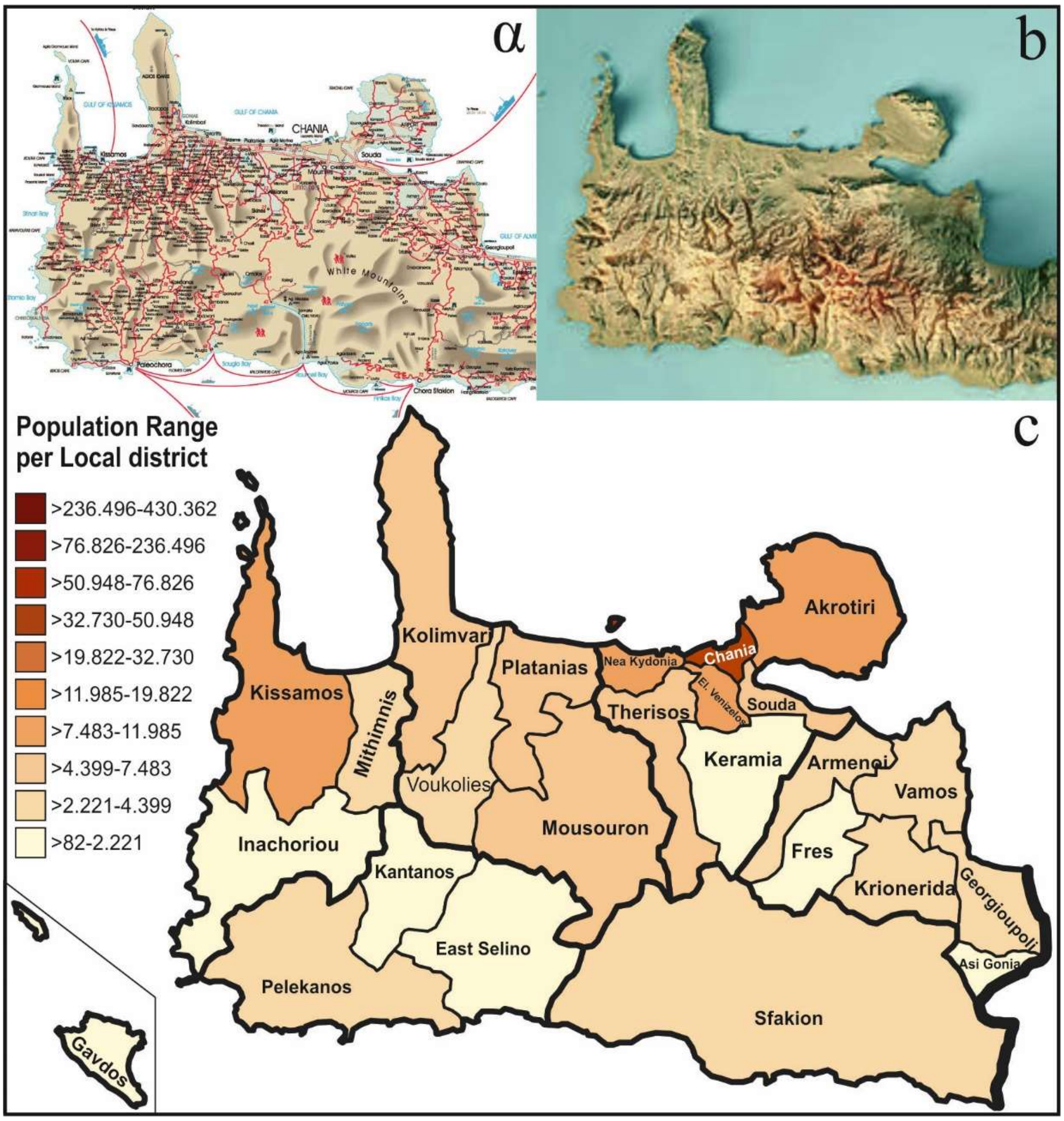


Figure 1 Prefecture of Chania: a) main road network map b) terrain map; and c) population density map in the main local communal areas.

Table 1 *shows municipalities and their local districts of the region of Chania, the distance of the regional centre from the MBT –landfill in km, the permanent population for 2021 and the corresponding waste production for that year in Mg.*

Prefecture Municipality Local district	Distance to MBT-Landfill (km)	Population	Recyclable fragment (Mg)	Mixed waste (Mg)	Total waste (Mg)
Chania Prefecture		**156.706**	**10,516.590**	**77,550.630**	**88,067.220**
Apokoronas Municipality		**12.247**	**914.610**	**8,342.890**	**9,257.500**
Armenoi	36.6	3.159	235.915	2,151.971	2,387.886
Asi gonia	74.5	456	34.054	310.636	344.690
Vamos	42.3	3.069	229.194	2,090.661	2,319.855
Georgioupolis	52.5	2.708	202.234	1,844.741	2,046.976
Krionerida	53.6	2.068	154.439	1,408.761	1,563.200
Fres	44.0	787	58.773	536.119	594.893
Kandanos-selinos Municipality		**5.009**	**192.150**	**2,534.950**	**2,727.100**
EastSelino	93.3	760	29.154	384.620	413.774
Kandanos	62.2	893	34.256	451.929	486.185
Pelekanos	95.7	3.356	128.739	1,698.401	1,827.141
Kissamos Municipality		**10.632**	**573.020**	**5,846.570**	**6,419.590**
Innahori	82.8	908	48.937	499.312	548.249
Kissamos	62.2	7.608	410.039	4,183.663	4,593.702
Mythimna	67.3	2.116	114.043	1,163.595	1,277.638
Platanias Municipality		**15.299**	**847.841**	**8,672.445**	**9,520.286**
Voukolies	51.5	2.877	159.438	1,630.866	1,790.304
Mousouron	38.0	3.618	200.503	2,050.912	2,251.415
Platanias	36.9	4.665	258.525	2,644.418	2,902.944
Kolymvari	49.0	4.139	229.375	2,346.248	2,575.623
Sfakia Municipality	83.3	**2.002**	**114.920**	**1,342.890**	**1,457.810**
Gavdos Local community [a]	83.3	142			
Chania Municipality		**111.375**	**7,874.049**	**50,810.885**	**58,684.934**
Akrotiri	7.9	14.111	997.627	6,437.642	7,435.269
Eleftherios Venizelos	23.0	13.018	920.353	5,939.000	6,859.353
Therisos	39.5	8.914	630.207	4,066.696	4,696.902
Keramia	37.3	738	52.176	336.686	388.862
Nea Kidonia	33.0	11.597	819.891	5,290.719	6,110.610
Souda	21.3	8.438	596.554	3,849.538	4,446.092
Chania	21.0	54.559	3,857.241	24,890.604	28,747.846

a) Gavdos local community is an Island that, for the collection of waste and recyclables, is served by collection vehicles from the local district of Sfakia and transferred to the island and back by Ferry boat. The ferry boat impacts are not accounted for, while the quantities of waste and population are added to the corresponding values of Sfakia Municipality.

The term Waste Production Node (WPN) (Chatzouridis and Komilis, 2012; Komilis, 2008) is used in this research to describe the geographical centre of each local district based on the drivable road network. It is assumed to represent the average distance of any district village. All distances used in this research are counted from this point and represent the shortest drivable distance between them and the waste facility.

Collection Vehicles: Various vehicles of different capacities, compaction types and emission standard technologies are used to collect MSW. This study is focused only on the EURO VI emission standard, backhoe kerbside collection, 16 m^3 volume, and press-type trucks commonly used in all the municipalities (Kaousis - CRV 2000 Standard®). The above collection vehicle is widely used in most municipalities of Greece.

Waste Transfer Stations (WTSs): Typical mobile waste transfer stations comprise an elevated stationary platform featuring a hopper at the edge. A 56 m^3 semi-trailer, parked beneath the platform, receives the waste materials. The collection vehicles unload the waste through the hopper into the semi-trailer, where the materials are compressed using the trailer's hydraulic system. After loading, the semi-trailer is towed to the waste facility by a tractor compliant with the EURO VI emission standards. The proposed WTS consists of a "KAOUSIS HAS 60® semi-trailer" that uses a silent technology air-cooled 4-stroke diesel engine with three cylinders to compress and discharge the waste. The engine is equipped with a Diesel Particulate Filter (DPF) and adheres to the emission standards outlined in EU Regulation 2016/1628 (EC, 2016).

Truck route analysis: Truck routes are divided into collection and transport. The collection commences at the location of the first collection bin and concludes at the last collected bin. The collection phase does not include the distance driven, as diesel usage is predominantly determined by waste type, housing type, and truck type and less by local or geographical differences in waste management. Parameters such as distance, number of stops, and collection frequency are all inherently linked to the waste type and housing type, and thus, in the diesel consumption. The distance is already included in the diesel consumption value for the collection. The transport accounts for all other distances traversed by the truck - from parking to the first container, from the last container to the treatment facility, and back to the parking station, as well as short trips for refuelling or maintenance. The transport phase does factor in the driven distance; hence, the unit of measurement is the consumed liters (L) of fuel per transported metric ton (tn) divided by kilometers (km) travelled.

1.1 Designing of scenarios

Five scenarios are designed to evaluate the environmental impacts of WTSs (Figure 2) for MW and RF streams. The first scenario S0 portrays the baseline collection and transport scheme in which collection vehicles collect the waste in each local district and then directly haul them to the treatment facility. In this scenario, the distances driven for the transport phase are accounted from the WPN of each local district to the treatment facility.

The scenario SA locates the WTSs as close as possible to the municipality centre and proximate to an existing major road network. In this approach, the WTSs are placed at distances varying from 1 to 5 km from the centre of the municipality. In most instances, the WTSs are located at distances between 1.5 and 2 km from a WPN, the approach features 6 WTS added, one in every municipality (the island of Gavdos is served by the Sfakia WTS). The scenario SB suggests the WTSs should be located within a distance of 16 km (10 miles) from the WPN. This aligns with USEPA (2002) guidelines stating that "transfer stations should be located no more than 16 km from the end of all collection routes in urban and suburban areas". To implement this, 16 km radius buffer zones are drawn around the centres of all WPNs. The WTSs are then sited at the centre of these buffer zone intersections. When the intersection involves more than two buffer zones, a WTS is positioned in the centre of the intersection, resulting in the highest number of overlapping buffer zones. The above approach introduces 5 WTS in the six municipalities, as shown in the corresponding map in Figure 2(SB). The scenarios SAi and SBi follow the same approach for positioning the WTSs like scenarios SA and SB, respectively, assuming that WPNs are in the premicies of the municipality (Chania) hosting the MRCF-landfill following direct trips to the facility without the use of a WTS.

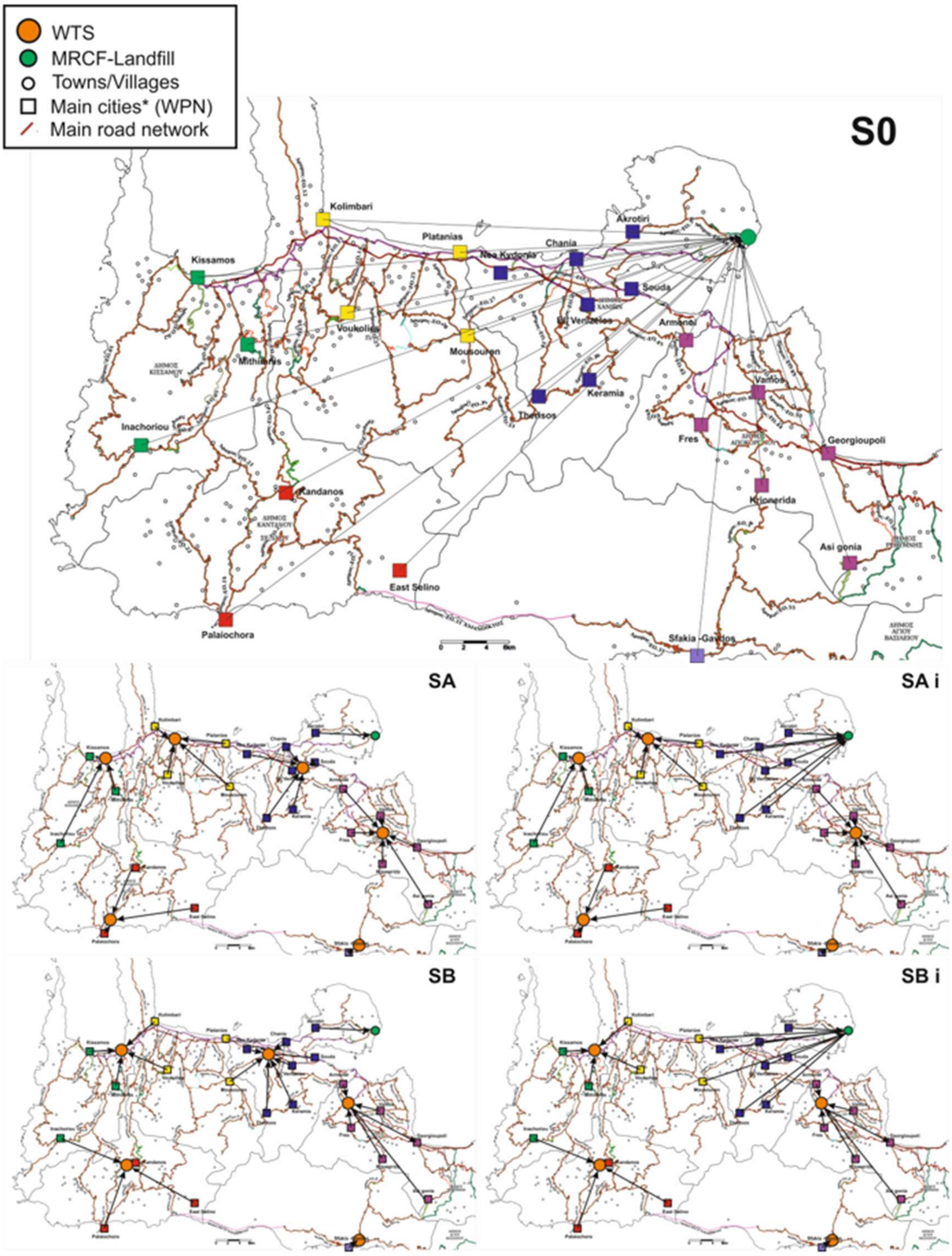


Figure 2 *illustrates the map and schematic diagram detailing five distinct waste management scenarios. SO) Baseline Scenario: Direct waste trips from vehicles to treatment facilities. SA) Integration of Waste Transfer Stations (WTSs) in all municipalities at the core of waste production nodes (WPN). SAi) Similar WTS locations as in scenario SA, except the municipality hosting the MRCF-landfill lacks a WTS. SB) Exploration of a scenario with a reduced number of WTSs located near the main road network. SBi) Precise WTS locations as in scenario SB, excluding the municipality with the MRCF-landfill. The map showcases the main road network, major towns, villages, MRCF-landfill location, proposed Mobile Transfer Stations (WTS), and waste destinations indicated by black arrows in each scenario.*

1.2 Data collection

The daily weighing data are obtained from the MRCF-landfill operated in the prefecture of Chania. The data include the date and time of the vehicle's arrival at the facility, license plate number, mixed and dead weight, and collection area. The fleet maintenance departments of the municipalities provide monthly fuel consumption data for the vehicles. Monthly travelled distances are obtained from the fleet monitoring software and cross-referenced with the distances recorded in the fleet maintenance logbooks. The collection routes are charted and separated into travelling and collection segments. Exact routes are recorded from the fleet management software of each agency where available or obtained through interviews with waste managers.

1.3 Calculation of fuel consumption

The local district hosting the waste facility is used as an area where fuel consumption for waste transport can be considered negligible. All fuel consumption for this area is attributed to waste collection. The resulting values for waste collection are then used to calculate the collection and transport consumptions for the rest of the local districts.

1.4 Calculation of emissions

Emission factors for fuel consumption are linked to every vehicle's Euro engine standard. The produced emissions are considered proportional to the fuel consumption by each vehicle. The diesel fuel mixture is based on Pulles et al. (2012). Euro VI engines are equipped with particle filter, it is considered to collect 50% of the heavy metal included in the fuel mixture (Franco González et al., 2021; Gioria et al., 2020; Pulles et al., 2012).

1.5 Assessment methodology

The LCIA is carried out using the dedicated LCA software EASETECH, which enables the evaluation of managing composite waste materials and facilitates monitoring mass and energy flows throughout the process chain (Clavreul et al., 2014). The recommended revised method, ILCD 2017 (Tobergte and Curtis, 2013), is used for the inventory assessment, which provides a standardised and harmonised framework, ensuring consistency and comparability across different life cycle assessment studies.

The examined systems are assessed against fourteen environmental impact categories presented in ***Table 2*** S1 supplementary information (European Commission, 2012). The LCIA results are presented in the corresponding units and, when necessary, are normalised into Person Equivalents (PE) for the reader to compare them. The results of those categories presenting negligible impacts are not presented.

Table 2 *ILCD 2013 updated impact categories (Tobergte and Curtis, 2013)*

Impact	Unit	Name
Global warming potential (climate change) with LT; 100 years IPPC2007	kg CO2-Eq	GWP100
Ozone depletion potential with LT,	kg CFC-11 eq.	ODP
Human toxicity, cancer effects, with LT	CTUh	HT CE
Human toxicity, non-cancer effects with LT	CTUh	HTnonCE
Particulate matter with LT	kgPM2.5-eq	PT
Ionising radiation human health with LT,	kBq U235 eq	IRP
Photochemical ozone formation, human health with LT	kg NMVOC	POF
Terrestrial acidification, Accumulated Exceedance	mol H^+ eq.	TAD
Eutrophication Terrestrial, Accumulated Exceedance	mol N eq.	EPT
Eutrophication Freshwater	kg P eq.	EPF
Eutrophication Marine with LT	kg N eq.	EPM
Ecotoxicity freshwater with LT	CTUe	ECF
Depletion of abiotic resources, mineral fossil & renewable	MJ	DAMF
Depletion of abiotic resources, elements (ultimate reserve)	kg Sb eq.	DAME

1.6 Sensitivity analysis

A sequence of sensitivity methodologies is used to evaluate the robustness of the developed model. Contribution analysis, perturbation analysis, uncertainty analysis, comparative analysis, and discernibility analysis methodologies are used to present, analyse, evaluate and interpret the produced results (Bisinella et al., 2016; Ripa et al., 2017). The LCA results are decomposed into their process contributions and sub-systems for contribution analysis, providing a quick overview of the significant contributors (Clavreul et al., 2012). For perturbation analysis, the sensitive parameters are identified by shifting each input parameter one at a time by a small percentage of 10% and evaluating whether it induces a significant change in a selected result based on the methodology presented in Bisinella et al. (2016). Since uncertainty analysis is devoted to systematically studying input propagation into output uncertainties, the Monde Carlo simulation methodology produces random sampling and analytical formulas (Groen et al., 2017, 2014). Every input parameter is considered a stochastic variable with a specified probability distribution. The LCA model is constructed with one particular realisation of every stochastic parameter, and the LCA results are calculated with this specific realisation. The above steps are repeated several times (1,000 in this study), and the sample of LCA results is investigated as to its statistical properties, mean, standard deviation, and confidence interval. Finally, the sensitivity concludes with discernibility analysis, whereby one scenario preference is quantified over another. The result is based on pairwise comparisons of results for individual Monte Carlo samples of S0 to SA and SB scenarios, presented as percentages representing the probability of one system performing more favourable results for the environment than the baseline scenario (Bisinella et al., 2016).

2. Results and discussion

2.1 Inventory analysis

The analysis of weighing data reveals that the waste collection vehicles employed in this study exhibited a volume of 16 m^3 with a carrying capacity ranging from 2,928 ± 451 kg for recycling and 6,413 ± 1,595 kg for mixed waste. The long-haul tractor used for material transfer demonstrated a load volume of 56 m^3 with carrying capacities of 12,310 ± 1,084 kg and 18,168 ± 1,691 kg for recycling and mixed waste, respectively. Based on the comprehensive data analysis, Table 3 presents the calculated fuel consumption in Liters (L) per Megagram (Mg) for the collection phase in both urban and rural areas for recycling and mixed waste and the fuel consumption in L per Mg and kilometer for the transport phase considering standard and long haul vehicles. It also presents the fuel consumed during the compaction process in the WTS. The collection phase exhibits variations between urban and rural areas. In urban areas with high population density and low vehicle speeds (ranging from 10 to 40 km/h), the average diesel consumption is calculated at 3.69 L/Mg for mixed waste and 9.62 L/Mg for the recyclable fragment. In rural areas with lower population density, the containers dispersed at greater distances and longer travel routes result in higher fuel consumption of 7.45 L/Mg and 12.8 L/Mg, respectively. The differences in fuel consumption between rural and urban waste collection have been reported in several studies, although the calculation methodologies and inclusion of transport-related fuel consumption may vary (Larsen et al., 2009; Nguyen and Wilson, 2010; Thanh and Matsui, 2013). It should be noted that the reported collection results exhibit high standard deviation due to several factors that influence fuel consumption, such as the selected vehicle routes and driver behaviour (Friedrich and Trois, 2013). Furthermore, the load capacity of waste collection vehicles impacts fuel efficiency, as both overloading and underloading can decrease fuel efficiency (González et al., 2021).

Table 3 *Collection and transport consumption calculated for the vehicle routes analysis for the region of Chania*

Collection	L of diesel per Mg of collected material (L/Mg)
Collection RF Urban [a]	9.62 ± 3.91
Collection RF Rural [a]	12.8 ± 4.00
Collection MW Urban [a]	3.69 ± 0.84
Collection MW Rural [a]	7.45 ± 0.14
Transport [b]	L of diesel for the transport of 1 Mg of material for 1 km (L/Mg km)
RF long haul truck 56 m^3 [c]	0.0338 ± 0.0048
MW long haul truck 56 m^3 [c]	0.0266 ± 0.0161
RF conventional truck 16 m^3 [c]	0.0572 ± 0.0150
MW conventional truck16 m^3 [c]	0.0314 ± 0.0158
WTS RF consumption [d]	0.454 L/Mg ± 0.054
WTS MW consumption [d]	0.544 L/Mg ± 0.345

a- Fuel consumed for material loading, compaction and travel during the collection phase.
b- The values account for km by a factor of 2 since they include the return travel of the vehicle.
c- The values refer to the fuel consumed only for transporting the material.
d- Fuel consumed from the tractor material compaction system during the loading and unloading of waste

The transport phase represents the second step in the waste collection process. In this study, the metric employed to quantify this stage environmental impact refers to the volume of diesel fuel required to transport one Mg of material over a distance of one km, encompassing both the outbound and return routes of the vehicle (in the calculations the travel distances have to be doubled to be representative). As presented in Table 3, traditional waste transfer methods entail considerable fuel consumption. However, using transfer stations, where materials are loaded on long-haul tractors, reduces travel distance by a factor of 4.2 for recyclables and 2.8 for mixed waste. It is important to note that loading the tractors at the WTS requires energy input in the form of diesel, which is proportional to the material mass exclusively and should be factored into the overall analysis.

In the alternative scenarios (SA and SB), a significant proportion of waste materials, representing 65.28% of the RF and 57.22% of the MW, are diverted to a WTS located at an average distance of 23 ± 5 kilometres from the MRCF- landfill. This diversion introduces an additional step in the transport process, which results in the consumption of 0.454 litres per Mg of RF and 0.544 litres per Mg of MW in terms of fuel equivalence. Based on the calculated fuel consumption for the extra step of loading and unloading the wastes in the WTS, to counterpart the extra fuel consumed, it was estimated that the distance between WTS and the MRCF - landfill has to be greater than 113.3 ± 6 km for mixed waste and 19.4 ± 3 km for recycling fragment. In a break even case, the fuel consumption is offset by the fuel saved owing to the reduced distance travelled by the tractor. However, in the SA and SB scenarios, the distance travelled is inadequate to achieve this breakeven point.

On the contrary, this excess consumption is avoided in the "i" scenarios, leading to improved efficiency and better outcomes. Scholars should exercise caution when considering the implementation of a Waste Transfer Station (WTS), as its installation might not entirely eradicate travel distances from specific regions to the WTS. This limitation could substantially elevate the breakeven distance and warrants careful consideration in research.

Concerning waste collection emissions, they are proportional to the vehicle's fuel consumption during collection and transport. Based on the European engine emission standard VI, Table S2 in the supplementary material shows the heavy-duty diesel engines' calculated values and implementation years. As of December 2012, the new European trucks must comply with the Euro VI standard. The emission standards are based on standardised test cycles that simulate various driving conditions, including engine speed, load, and temperature. However, since the emission standards represent standardised, average driving conditions, actual emissions may vary considerably depending on truck usage. The emission values represent the upper limit for release into the atmosphere.

Table 4 *Maximum emissions in kg per Liter of compacted diesel fuel for waste collection trucks in Greece (Engine EU Emission Standard VI)*

Engine standared	Euro VI
Date of Implementation	Dec 2012
CO kg/L of diesel	0.00525
HC kg/L of diesel	0.00046
NO_x kg/L of diesel	0.00140
NH_3 kg/L of diesel	0.03500
PM kg/L of diesel	0.00004
Arsenic kg/L of diesel	4.30E-08
Cadmium kg/L of diesel	2.15E-08
CO_2 fossil kg/L of diesel	2.669
Chromium kg/L of diesel	1.03E-05
Lead kg/L of diesel	2.58E-07
Mercury kg/L of diesel	1.98E-06
Nickel kg/L of diesel	8.60E-08
Selenium kg/L of diesel	8.60E-09
SO_2 kg/L of diesel	1.63E-05
Zinc kg/L of diesel	0

*1 liter of diesel = 0.0035 kWh Density of diesel = 860 kg/m^3. Values are based on the emission standards for heavy-duty diesel engines used in trucks and buses, defined in kilograms per litter of consumed diesel. The official category name is heavy-duty diesel engines, which include lorries and buses.

2.2 Life Cycle Inventory Assessment (LCIA)

Based on calculations, the net characterised results for the midpoint impact GWP, the total collection and transport to the MRCF of MW is 23.35 kg CO_2-eq per Mg of wet waste and for RF, 44.33 kg CO_2-eq per Mg of wet waste. GWP (also called climate change) is a significant midpoint impact in LCA studies. Its extended use among LCA studies makes the results comparable (Christensen et al., 2020; Papadaskalopoulou et al., 2019; Zeller et al., 2020). The climate change impact is determined based on the mass emission (kg) of three gases: N_2O, CH_4, and CO_2. These emissions are converted into (kg) CO_2-eq using the ILCD equivalent factors. Table S3 (supplementary information) presents the GWP impact for both collection streams in all scenarios.

To make the calculated impacts comparable, the net results are normalised in mili-person equivalent (mPE) for the different waste management scenarios per environmental impact (Aymard and Botta-Genoulaz, 2017; Tobergte and Curtis, 2013). A selective presentation of the highest-rated impact is shown in **Figure 4**, while all impacts and their normalised numeric values can be found in Figure S2 and Table S4 in the supplementary information. The comparison between the total environmental impacts of the two waste collection streams reveals that mixed waste processes result in lower impacts per unit collected when compared to their resource recovery (RF) counterparts. This variation arises from disparities in material densities, which enable trucks to transport larger masses of mixed waste within the same volume. This impact disparity spans various categories, ranging from 44% to 38% in GWP impact and 25% to 45% across other categories. All scenarios present significant environmental burdens for the GWP, HT-CE, HTno-CE, and DAMF impact categories with net results above zero. The impacts of ODP, IRP, EPF, and DAME are minor and considered insignificant.

Table 5 characterized regional results for GWP in kg CO2-eq per Mg of wet waste

Scenario	Materials	kgCO$_2$-eq/Mg ww	std	Variation
SO	MIX	23.35	2.43	5.89
SA	MIX	20.77	1.44	2.7
SAi	MIX	21.37	1.73	2.99
SB	MIX	20.87	1.37	1.89
SBi	MIX	21.70	1.88	3.53
SO	REC	44.33	12.62	159.28
SA	REC	38.07	9.17	84.18
SAi	REC	41.46	10.80	116.57
SB	REC	39.16	8.82	77.87
SBi	REC	41.00	11.15	124.29

In order to gain deeper insights into the impact of each scenario, we conducted a contribution analysis. The system is divided into three distinct process groups, each depicted in different colours and signifying the net cumulative effect of the constituent sub-processes for source recovery (RF) and municipal waste (MW). These process groups include collection (clt), transportation (trp), and the waste transfer station (wts), the latter encompassing all processes associated with the municipal transfer station except for travel-related elements. Results above zero indicate burdens imposed on the environment, signifying potential adverse impacts. Conversely, results below zero denote avoided emissions, signifying a positive environmental benefit, as detailed by (Blengini et al., 2012).

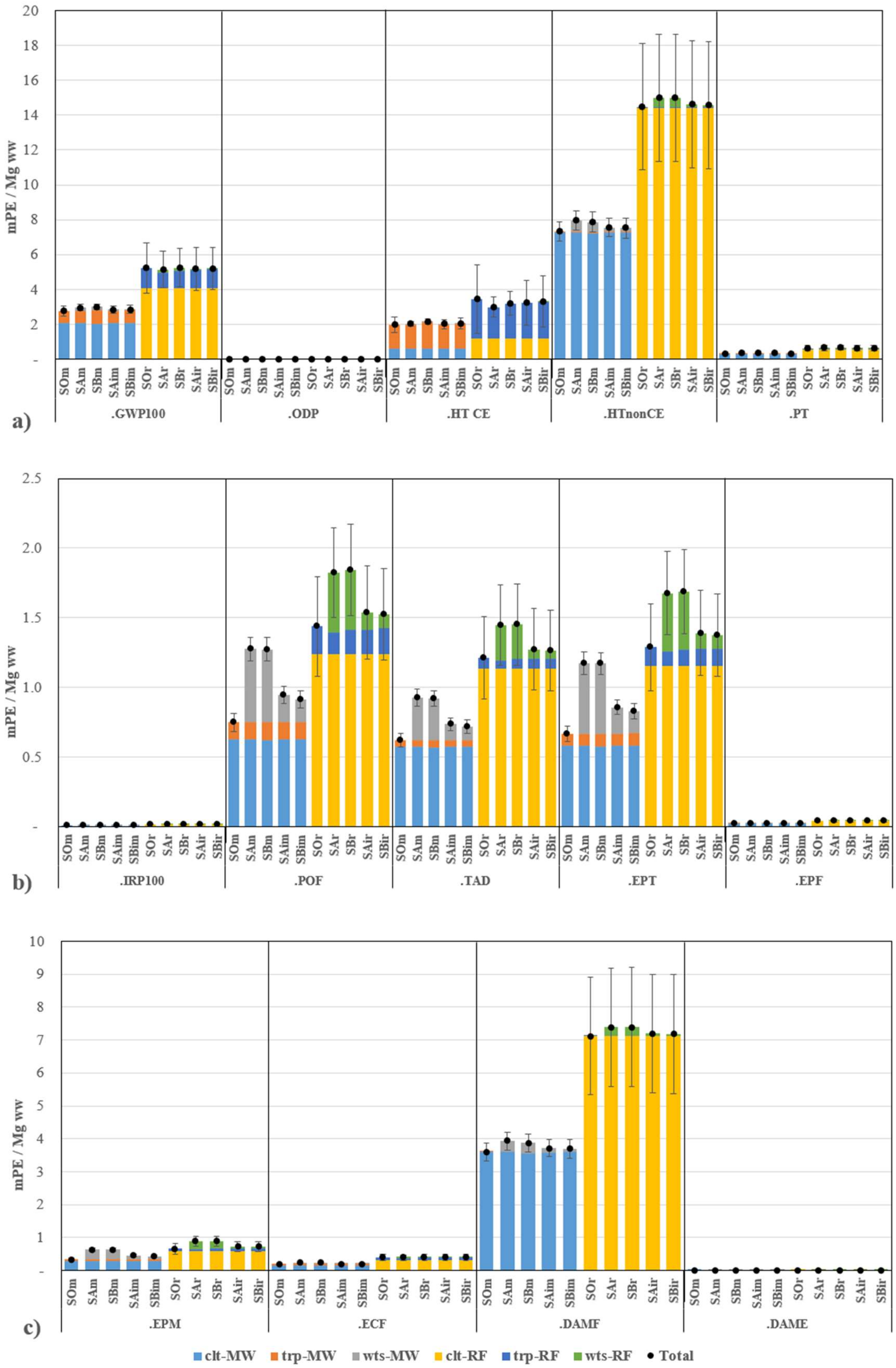


Figure 3 *Net normalised results presented as black dots in mili Person Equivalents per Mg of wet waste for mixed waste collection (scenarios marked with "m") and for recyclables (scenarios marked*

with "r") . Contribution analysis is presented as colour bars representing different groups of primary processes

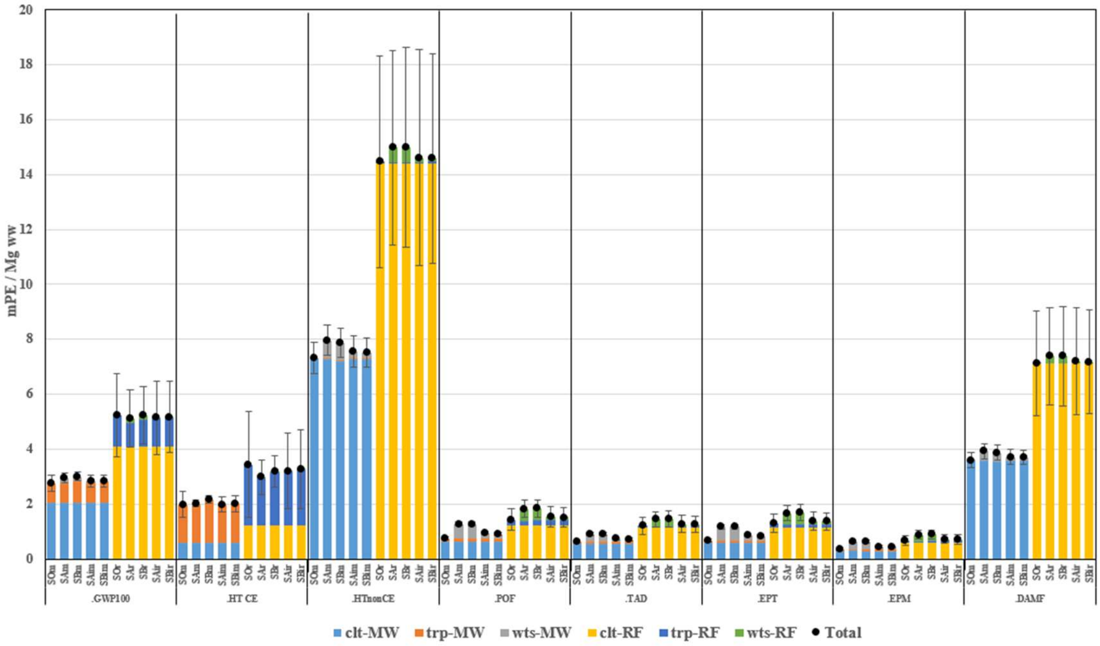


Figure 4 *Net normalised results presented as black dots in mili Person Equivalents per Mg of wet waste for mixed waste collection (scenarios marked with "m") and for recyclables (scenarios marked with "r"). Contribution analysis is presented as colour bars representing different groups of primary processes.*

The baseline scenario (SO) represents the conventional waste collection method of the two waste fragments. The alternative scenarios show higher net results in most impacts while assessing the effects of locating WTS within a few kilometres from the end of the collection area of each territory. The scenarios SA and SB yield the same difference across all impact categories compared to the SO scenario. The above suggests no significant difference in environmental impacts between the two scenarios due to the minor variations in the distances of locating the WTS in the two methodologies. The first alternative scenario, SA, uses the territory population and waste production data to pinpoint the WPN and locate the WTS closer to the average WPN. The normalised net result for this scenario is less beneficial than the baseline.

The scenarios SAi and SBi exhibit lower mPE across most impact categories, suggesting that they have less environmental impact than the SA and SB but still higher than the baseline scenario. Specifically, the SBi scenario consistently exhibits the lowest impacts across all categories, indicating that it may be the most environmentally beneficial among the four alternative scenarios.

The results of the contribution analysis performed are also presented in **Figure 4Σφάλμα! Το αρχείο προέλευσης της αναφοράς δεν βρέθηκε.** as columns and numerically in Table S4 in the supplementary information. The waste collection holds approximately 68-73% for MW and % 78-80% for RF (SA, SB) to 75% and 78% (SO), respectively, of the produced impacts for the GWP, while its contribution to other impacts is also significant. In Human toxicity for non-cancer effects, this contribution reaches 98% (SA) to 99.5% (SO), and the same trend is shown for DAMF and DAME, concluding that the collection part is the most resource-demanding and consuming. Table S4 (supplementary information) presents the GWP impact for all scenarios. Madden et al. (2022) stated that kerbside collection was responsible for approximately 88.6% of all fuel consumed and is a significant contributor to emissions. The transport process contributes to 24.3% of the GWP, 67.9% of the HT CE and less than 15% of the impacts in the other categories. In the alternative scenarios, the contribution of the WTS processes to the impacts is not negligible. The WTS contributes to 5.6% of the GWP impact for the SA and SB scenario, while it is reduced to 2.2% for SAi and 1.8% for SBi. For ODP, it reaches 7.3% for SA and SB and 2.8% and 2.3% for SBi and SBi, respectively.

The standard deviation derived from the obtained data is a vital metric indicating the variability in the impact of different routes. This measure effectively captures the influence of various factors such as seasonal variations in waste prodaction, collection and transport, as well as driving habits. When considering alternative scenarios involving the implementation of Waste Transport Systems (WTS), the observed standard deviation exhibits reduced fluctuations. This phenomenon can be attributed to the enhanced efficiency of waste transport within WTS, where tractors are consistently loaded to capacity during each trip. This stands in contrast to conventional collection trucks, which adhere to predetermined schedules irrespective of the presence of waste, leading to more inconsistent outcomes.

***Table S6** Net normalised results in mili Person Equivalents per Mg of wet waste for mixed waste collection (scenarios marked with "m") and for recyclables (scenarios marked with "r"). Contribution analysis is presented as colour bars representing different groups of primary processes. Contribution normalized results and Net characterized results obtained from primary data set for midpoint and endpoint impacts for the S0, SA, and SB scenarios. Mean, standard deviation, and variances obtained for every impact, based on primary data uncertainties put to the test using Monte Carlo simulation for 1000 runs*

Impact	Process in mPE							Monte-Carlo net results	
Scenario and stream	clt-MW	trp-MW	wts-MW	clt-RF	trp-RF	wts-RF	Net result	m=	d=
GWP100									
SOm	2.067	0.706					2.773	2.798	0.299
SAm	2.068	0.704	0.189				2.961	5.263	1.515
SBm	2.044	0.782	0.187				3.012	2.546	0.214
SAim	2.065	0.700	0.071				2.836	4.853	1.332
SBim	2.067	0.715	0.059				2.840	2.475	0.170
SOr				4.087	1.152		5.238	4.604	1.041
SAr				4.087	0.890	0.155	5.132	2.584	0.210
SBr				4.087	0.993	0.155	5.236	4.931	1.308
SAir				4.087	1.024	0.043	5.154	2.465	0.170
SBir				4.087	1.055	0.036	5.178	4.545	1.047
ODP									
SOm	0.000	0.000			-		0.000	0.000	0.000
SAm	0.000	0.000	0.000				0.000	0.001	0.000
SBm	0.000	0.000	0.000				0.000	0.000	0.000
SAim	0.000	0.000	0.000				0.000	0.001	0.000
SBim	0.000	0.000	0.000				0.000	0.000	0.000
SOr				0.001	0.000		0.001	0.001	0.000
SAr				0.001	0.000	0.000	0.001	0.000	0.000
SBr				0.001	0.000	0.000	0.001	0.001	0.000
SAir				0.001	0.000	0.000	0.001	0.000	0.000
SBir				0.001	0.000	0.000	0.001	0.001	0.000
.HTCE									
SOm	0.613	1.370			-		1.982	2.022	0.479
SAm	0.613	1.365	0.051				2.029	3.382	1.944
SBm	0.606	1.516	0.051				2.173	1.425	0.271
SAim	0.612	1.358	0.019				1.990	2.652	1.364
SBim	0.613	1.386	0.016				2.015	1.096	0.134
SOr				1.212	2.233		3.445	2.020	0.616
SAr				1.212	1.726	0.042	2.981	1.517	0.287
SBr				1.212	1.926	0.042	3.181	2.781	1.444
SAir				1.212	1.985	0.012	3.209	1.130	0.145
SBir				1.212	2.046	0.010	3.268	1.926	0.577
HTnonCE									
SOm	7.289	0.037			-		7.325	7.346	0.573
SAm	7.294	0.036	0.635				7.966	14.709	3.845
SBm	7.207	0.040	0.629				7.877	7.547	0.562
SAim	7.283	0.036	0.238				7.557	14.540	3.930
SBim	7.289	0.037	0.197				7.522	7.924	0.550
SOr				14.413	0.060		14.473	14.818	3.556
SAr				14.413	0.046	0.523	14.982	7.512	0.543
SBr				14.413	0.051	0.523	14.988	14.595	3.814
SAir				14.413	0.053	0.145	14.611	7.810	0.540
SBir				14.413	0.055	0.120	14.588	14.772	3.644
PT									
SOm	0.292	0.033			-		0.326	0.327	0.026
SAm	0.292	0.033	0.036				0.362	0.640	0.164
SBm	0.289	0.037	0.036				0.362	0.325	0.024
SAim	0.292	0.033	0.014				0.339	0.619	0.162
SBim	0.292	0.034	0.011				0.337	0.338	0.022
SOr				0.578	0.055		0.632	0.621	0.143
SAr				0.578	0.042	0.030	0.650	0.325	0.023
SBr				0.578	0.047	0.030	0.655	0.623	0.157
SAir				0.578	0.048	0.008	0.635	0.335	0.022
SBir				0.578	0.050	0.007	0.635	0.617	0.146

Impact	Process in mPE							Monte-Carlo net results	
Scenario and stream	clt-MW	trp-MW	wts-MW	clt-RF	trp-RF	wts-RF	Net result	m=	d=
SOm	0.006	0.000			-		0.006	0.006	0.000
SAm	0.006	0.000	0.001				0.006	0.012	0.003
SBm	0.006	0.000	0.001				0.006	0.006	0.000
SAim	0.006	0.000	0.000				0.006	0.012	0.003
SBim	0.006	0.000	0.000				0.006	0.006	0.000
SOr				0.012	0.000		0.012	0.012	0.003
SAr				0.012	0.000	0.000	0.012	0.006	0.000
SBr				0.012	0.000	0.000	0.012	0.012	0.003
SAir				0.012	0.000	0.000	0.012	0.006	0.000
SBir				0.012	0.000	0.000	0.012	0.012	0.003
POF									
SOm	0.627	0.121			-		0.748	0.753	0.066
SAm	0.627	0.120	0.525				1.273	1.450	0.381
SBm	0.620	0.133	0.520				1.274	0.894	0.060
SAim	0.626	0.120	0.197				0.943	1.481	0.362
SBim	0.627	0.122	0.163				0.912	1.186	0.084
SOr				1.240	0.197		1.437	1.729	0.314
SAr				1.240	0.152	0.433	1.824	0.869	0.057
SBr				1.240	0.170	0.433	1.842	1.478	0.353
SAir				1.240	0.175	0.120	1.535	1.181	0.087
SBir				1.240	0.180	0.099	1.519	1.715	0.316
TAD									
SOm	0.574	0.048			-		0.621	0.624	0.049
SAm	0.574	0.047	0.303				0.924	1.228	0.314
SBm	0.567	0.053	0.300				0.920	0.715	0.048
SAim	0.573	0.047	0.114				0.734	1.248	0.314
SBim	0.574	0.048	0.094				0.715	0.888	0.058
SOr				1.134	0.078		1.212	1.400	0.282
SAr				1.134	0.060	0.249	1.443	0.699	0.045
SBr				1.134	0.067	0.249	1.450	1.247	0.305
SAir				1.134	0.069	0.069	1.272	0.882	0.060
SBir				1.134	0.071	0.057	1.262	1.391	0.286
EPT									
SOm	0.582	0.084			-		0.666	0.670	0.055
SAm	0.582	0.084	0.507				1.174	1.302	0.336
SBm	0.575	0.093	0.502				1.171	0.821	0.054
SAim	0.581	0.084	0.190				0.855	1.350	0.327
SBim	0.582	0.085	0.157				0.824	1.111	0.080
SOr				1.151	0.138		1.288	1.606	0.291
SAr				1.151	0.106	0.418	1.675	0.795	0.051
SBr				1.151	0.119	0.418	1.687	1.344	0.318
SAir				1.151	0.122	0.116	1.389	1.107	0.083
SBir				1.151	0.126	0.096	1.373	1.593	0.293
EPF									
SOm	0.020	0.000			-		0.020	0.020	0.002
SAm	0.020	0.000	0.002				0.022	0.041	0.011
SBm	0.020	0.000	0.002				0.022	0.021	0.002
SAim	0.020	0.000	0.001				0.021	0.041	0.011
SBim	0.020	0.000	0.001				0.021	0.022	0.002
SOr				0.040	0.000		0.040	0.041	0.010
SAr				0.040	0.000	0.001	0.042	0.021	0.002
SBr				0.040	0.000	0.001	0.042	0.041	0.011
SAir				0.040	0.000	0.000	0.041	0.022	0.002
SBir				0.040	0.000	0.000	0.041	0.041	0.010

Impact	Process in mPE							Monte-Carlo net results	
Scenario and stream	clt-MW	trp-MW	wts-MW	clt-RF	trp-RF	wts-RF	Net result	m=	d=
EPM									
SOm	0.299	0.048			-		0.347	0.349	0.029
SAm	0.299	0.048	0.287				0.634	0.676	0.175
SBm	0.295	0.053	0.284				0.632	0.434	0.029
SAim	0.298	0.048	0.108				0.454	0.704	0.169
SBim	0.299	0.049	0.089				0.436	0.598	0.044
SOr				0.590	0.079		0.669	0.848	0.150
SAr				0.590	0.061	0.236	0.887	0.419	0.027
SBr				0.590	0.068	0.236	0.894	0.700	0.165
SAir				0.590	0.070	0.066	0.726	0.596	0.046
SBir				0.590	0.072	0.054	0.716	0.842	0.151
ECF									
SOm	0.166	0.040			-		0.206	0.208	0.019
SAm	0.166	0.040	0.014				0.220	0.397	0.107
SBm	0.164	0.045	0.014				0.222	0.194	0.015
SAim	0.166	0.040	0.005				0.211	0.372	0.099
SBim	0.166	0.041	0.004				0.211	0.192	0.013
SOr				0.328	0.066		0.394	0.359	0.082
SAr				0.328	0.051	0.011	0.390	0.196	0.015
SBr				0.328	0.057	0.011	0.396	0.377	0.097
SAir				0.328	0.059	0.003	0.390	0.191	0.013
SBir				0.328	0.060	0.003	0.391	0.355	0.083
DAMF									
SOm	3.602	0.000			-		3.602	3.612	0.283
SAm	3.604	0.000	0.322				3.926	7.240	1.898
SBm	3.561	0.000	0.319				3.880	3.722	0.278
SAim	3.599	0.000	0.121				3.719	7.168	1.941
SBim	3.602	0.000	0.100				3.701	3.918	0.272
SOr				7.122	0.000		7.122	7.318	1.757
SAr				7.122	0.000	0.265	7.387	3.703	0.269
SBr				7.122	0.000	0.265	7.387	7.193	1.884
SAir				7.122	0.000	0.074	7.196	3.861	0.267
SBir				7.122	0.000	0.061	7.183	7.297	1.801
DAME									
SOm	0.009	0.000			-		0.009	0.009	0.001
SAm	0.009	0.000	0.001				0.009	0.017	0.005
SBm	0.009	0.000	0.001				0.009	0.009	0.001
SAim	0.009	0.000	0.000				0.009	0.017	0.005
SBim	0.009	0.000	0.000				0.009	0.009	0.001
SOr				0.017	0.000		0.017	0.017	0.004
SAr				0.017	0.000	0.001	0.018	0.009	0.001
SBr				0.017	0.000	0.001	0.018	0.017	0.005
SAir				0.017	0.000	0.000	0.017	0.009	0.001
SBir				0.017	0.000	0.000	0.017	0.017	0.004

- Process groups (clt) collection, (trp) transportation, (wts) waste transfer station. (MW) mixed waste, (RF) Recyclable fragment

In an authentic waste management system, accounting for the population distribution within the study area is essential. Areas with high population density exhibit elevated waste production, significantly impacting the outcomes and the distance to the main destination facility. Within the context of the Chania prefecture, a substantial portion of population activity is concentrated in the northern region, in close proximity to the MRCF landfill. A more precise analysis can be achieved by focusing on distinct regions separately. For instance, conducting the methodology independently for two municipalities—one characterised by high population density and another, specifically the municipality of Kandanos—can yield clearer and more meaningful results.

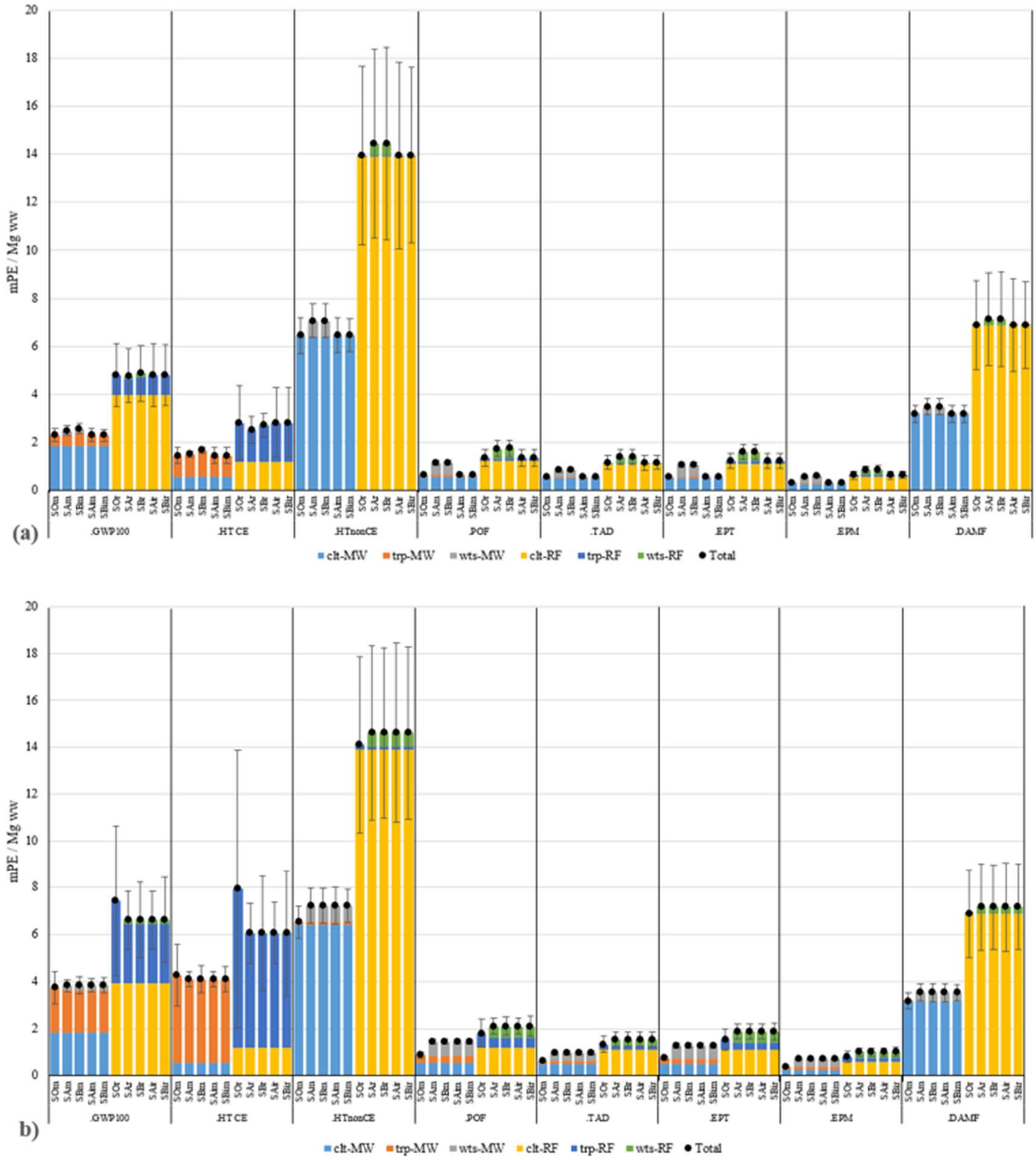


Figure 5. *Net normalised results for the hight rated impacts, for the municipalities' of Chania (a) and Kandanos (b), presented as black dots in mili Person Equivalents per Mg of wet waste for mixed waste collection (scenarios marked with "m") and for recyclables (scenarios marked with "r"). Contribution analysis is presented as colour bars representing different groups of primary processes.*

Figure 3a illustrates the results specific to the municipality of Chania. The MRCF-Landfill is an average distance of 10km from this municipality's Waste Processing Node (WPN). Among the scenarios examined (S0, SAi, and SBi), those excluding a Waste Transport System (WTS) exhibit the lowest net results for Municipal Waste (MW) stream collection. Introducing a WTS in the area intensifies the challenges, necessitating additional travel from the WTS to the MRCF and associated impacts. However, concerning Recycling Fragments (RF) transport, the SA scenario, which integrates a WTS, demonstrates low improvement in the corresponding impacts. Reducing the standard deviation in both waste streams (MW, RF) in the alternative scenarios is regarded as a significant advantage associated with the efficiency of the transport process, rendering these scenarios favourable, especially in the case of RF. Regarding MW, the slight deviation from the baseline scenario can be compensated by factors such as refused trips and driving hours in this context.

In Figure 3b, the data for the municipality of Kandanos indicates a 6% reduction in transportation impacts between the scenarios. The net results are balanced by the additional impacts from the Waste Transport System (WTS), rendering the alternative scenarios acceptable. Furthermore, reducing standard deviation provides a clear advantage to these scenarios. In the case of the Recycling Fragments (RF) stream, the transportation impacts are noticeably reduced by 29%, accompanied by a decrease in standard deviation. This highlights the significant benefits of implementing the WTS specifically for the RF stream.

Comparing the outcomes between the two municipalities, it is evident that the contribution from waste collection is nearly identical in both study cases, with variations attributed to the urban characteristics of the Chania municipality. The travel-related contribution accounts for 18-20% in MW and 14-17% in RF of the net results for Chania. In contrast, it rises significantly to encompass 47-51% in MW and 37-47% in RF for the remote municipality of Kantanos. This underscores the paramount importance of Waste Transfer Stations (WTSs) in the context of these findings.

Table 7 *Selective presentation of parameters with the highest influence on the studied scenarios for each endpoint impact. The most sensitive parameters for each of the presented midpoint impacts for each scenario*

Scenario	Parameter name	GWP	ODP	HT CE	HTnonCE	PT	IRP	POF	TAD	EPT	EPF	EPM	ECF	DAMF	DAME
SO	clt_mx_r	100%	100%	40%	100%	100%	100%	80%	100%	77%	100%	70%	100%	100%	100%
SO	clt_mx_u	49%	49%	19%	49%	49%	49%	39%	49%	37%	49%	34%	49%	49%	49%
SO	clt_rc_r	21%	21%	8%	21%	21%	21%	17%	21%	16%	21%	15%	21%	21%	21%
SO	clt_rc_u	19%	19%	8%	19%	19%	19%	15%	19%	15%	19%	13%	19%	19%	19%
SO	kmtt_chn	18%	0%	48%	0%	6%	0%	8%	4%	6%	0%	6%	13%	0%	0%
SO	trp_mx_cv	12%	0%	33%	0%	4%	0%	6%	3%	4%	0%	4%	9%	0%	0%
SO	trp_mx_ts	38%	0%	100%	1%	13%	0%	17%	9%	12%	0%	13%	27%	0%	0%
SO	wts_mx	14%	13%	5%	13%	18%	13%	100%	78%	100%	13%	100%	12%	13%	13%
SO	wts_rc	2%	1%	1%	1%	2%	1%	11%	9%	11%	1%	11%	1%	1%	1%

Scenario	Parameter name	GWP	ODP	HT CE	HTnonCE	PT	IRP	POF	TAD	EPT	EPF	EPM	ECF	DAMF	DAME
SA	clt_mx_r	100%	100%	40%	100%	100%	100%	80%	100%	77%	100%	70%	100%	100%	100%
SA	clt_mx_u	49%	49%	19%	49%	49%	49%	39%	49%	37%	49%	34%	49%	49%	49%
SA	clt_rc_r	21%	21%	8%	21%	21%	21%	17%	21%	16%	21%	15%	21%	21%	21%
SA	clt_rc_u	19%	19%	8%	19%	19%	19%	15%	19%	15%	19%	13%	19%	19%	19%
SA	kmtt_chn	18%	0%	48%	0%	6%	0%	8%	4%	6%	0%	6%	13%	0%	0%
SA	trp_mx_cv	12%	0%	33%	0%	4%	0%	6%	3%	4%	0%	4%	9%	0%	0%
SA	trp_mx_ts	38%	0%	100%	1%	13%	0%	17%	9%	12%	0%	13%	27%	0%	0%
SA	wts_mx	14%	13%	5%	13%	18%	13%	100%	78%	100%	13%	100%	12%	13%	13%
SA	wts_rc	2%	1%	1%	1%	2%	1%	11%	9%	11%	1%	11%	1%	1%	1%

Scenario	Parameter name	GWP	ODP	HT CE	HTnonCE	PT	IRP	POF	TAD	EPT	EPF	EPM	ECF	DAMF	DAME
SAi	clt_mx_r	100%	100%	56%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%
SAi	clt_mx_u	49%	49%	27%	49%	49%	49%	49%	49%	49%	49%	49%	49%	49%	49%
SAi	clt_rc_r	21%	21%	12%	21%	21%	21%	21%	21%	21%	21%	21%	21%	21%	21%
SAi	clt_rc_u	19%	19%	11%	19%	19%	19%	19%	19%	19%	19%	19%	19%	19%	19%
SAi	kmm_cchn	13%	0%	47%	0%	4%	0%	7%	3%	5%	0%	6%	9%	0%	0%
SAi	trp_mx_cv	27%	0%	100%	0%	9%	0%	15%	7%	12%	0%	13%	20%	0%	0%
SAi	trp_rc_cv	7%	0%	26%	0%	2%	0%	4%	2%	3%	0%	3%	5%	0%	0%
SAi	trp_rc_ts	3%	0%	10%	0%	1%	0%	2%	1%	1%	0%	1%	2%	0%	0%
SAi	wts_mx	5%	5%	3%	5%	7%	5%	47%	29%	49%	5%	54%	5%	5%	5%
SAi	wts_rc	0%	0%	0%	0%	1%	0%	4%	2%	4%	0%	4%	0%	0%	0%

Scenario	Parameter name	GWP	ODP	HT CE	HTnonCE	PT	IRP	POF	TAD	EPT	EPF	EPM	ECF	DAMF	DAME
SB	clt_mx_r	100%	100%	36%	100%	100%	100%	80%	100%	77%	100%	70%	100%	100%	100%
SB	clt_mx_u	49%	49%	17%	49%	49%	49%	39%	49%	37%	49%	34%	49%	49%	49%
SB	clt_rc_r	21%	21%	7%	21%	21%	21%	17%	21%	16%	21%	15%	21%	21%	21%
SB	clt_rc_u	19%	19%	7%	19%	19%	19%	15%	19%	15%	19%	13%	19%	19%	19%
SB	kmtt_vamc	27%	0%	62%	0%	9%	0%	12%	6%	9%	0%	9%	19%	0%	0%
SB	trp_mx_cv	13%	0%	31%	0%	4%	0%	6%	3%	4%	0%	4%	9%	0%	0%
SB	trp_mx_ts	43%	0%	100%	1%	14%	0%	19%	10%	14%	0%	14%	31%	0%	0%
SB	wts_mx	14%	13%	4%	13%	18%	13%	100%	78%	100%	13%	100%	12%	13%	13%
SB	wts_rc	2%	1%	0%	1%	2%	1%	11%	9%	11%	1%	11%	1%	1%	1%

Scenario	Parameter name	GWP	ODP	HT CE	HTnonCE	PT	IRP	POF	TAD	EPT	EPF	EPM	ECF	DAMF	DAME
SBi	clt_mx_r	100%	100%	50%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%
SBi	clt_mx_u	49%	49%	24%	49%	49%	49%	49%	49%	49%	49%	49%	49%	49%	49%
SBi	clt_rc_r	21%	21%	10%	21%	21%	21%	21%	21%	21%	21%	21%	21%	21%	21%
SBi	clt_rc_u	19%	19%	9%	19%	19%	19%	19%	19%	19%	19%	19%	19%	19%	19%
SBi	kmm_cchn	13%	0%	42%	0%	4%	0%	7%	3%	5%	0%	6%	9%	0%	0%
SBi	kmtt_kolv	10%	0%	31%	0%	3%	0%	5%	2%	4%	0%	5%	7%	0%	0%
SBi	trp_mx_cv	31%	0%	100%	0%	10%	0%	17%	7%	13%	0%	15%	22%	0%	0%
SBi	trp_mx_ts	21%	0%	67%	0%	7%	0%	12%	5%	9%	0%	10%	15%	0%	0%
SBi	wts_mx	4%	4%	2%	4%	6%	4%	39%	24%	40%	4%	44%	4%	4%	4%
SBi	wts_rc	0%	0%	0%	0%	0%	0%	3%	2%	3%	0%	4%	0%	0%	0%

Parameter addreviations: clt_mx_r, Fuel consumption for collection of mixed waste in rural areas; clt_mx_u, Fuel consumption for collection of mixed waste in urban areas; clt_rc_r, Fuel consumption for collection of recyclable fraction in urban areas; clt_rc_u, Fuel consumption for collection of recyclable fraction in rural areas; kmm_cchn, Distance from Chania municipality waste production node to the MRCF-landfill for the specific scenario; kmtt_chn, Distance from Chania municipality waste Transfer station to the MRCF-landfill for the specific scenario; kmtt_kolv, Distance from Kissamos municipality waste Transfer station to the MRCF-landfill for the specific scenario; kmtt_vamc, Distance from Chania municipality waste Transfer station to the MRCF-landfill for the specific scenario; trp_mx_cv, Fuel consumption for collection of mixed waste with convectional vehicle; trp_mx_ts, Fuel consumption for collection of mixed waste with Long haul vehicle; trp_rc_cv, Fuel consumption for collection of recycling fragment with convectional vehicle; trp_rc_ts, Fuel consumption for collection of recycling fragment with Long haul vehicle; wts_mx, Fuel consumption for loading and unloading of mixed waste in the waste transfer station; wts_rc, Fuel consumption for loading and unloading of recycling fragments in the waste transfer station.

2.3 Sensitivity analysis

The sensitivity analysis highlights the main influencing parameters for each scenario (Table S5). The ranking is based on the methodology calculating the Normalised Sensitivity Ratio (NSR) for each system parameter, as proposed by Andreasi Bassi et al. (2017). The highest ranked parameter in GWP, ODP, HT - nonCE, PT, IRP, TAD, EPF, ECF, DAMF, and DAME is the fuel consumption per Mg for the collected mixed waste for rural areas, followed by the corresponding value for urban collection. This value is influenced by various parameters concerning the number of stops, driving conditions, road conditions and altitude variations of the terrain (Liu et al., 2022). The parameters concerning material transport are also considered necessary, especially for HT-CE, due to releasing NO_x into the environment (Friedrich and Trois, 2013).

The uncertainty propagation results are extracted from the Monte-Carlo simulation performed for 1000 runs in all five scenarios and every midpoint impact. The comparison between the studied scenarios is sufficient and confirms the robustness of the used parameters. The mean, standard deviation, and variance are quoted with the net-characterised results in Figure 4**Σφάλμα! Το αρχείο προέλευσης της αναφοράς δεν βρέθηκε.** and Table S3. An insignificant deviation of the mean values from the net results is observed in most impact categories.

The results of the discernibility analysis are presented in Figure 6**Σφάλμα! Το αρχείο προέλευσης της αναφοράς δεν βρέθηκε.** and Table S6. It shows the times in 1,000 runs of Monte-Carlo simulation, where the alternative scenarios perform better (have a value above zero) than the baseline scenario. The quantification of these results shows the percentage of times each scenario (SA, SAi, SB, SBi) is predicted to be more beneficial than the baseline for different environmental impacts through Monte Carlo analysis.

For the Global Warming Potential (GWP) and Human Toxicity - Carcinogenic Effects (HT-CE), the Monte Carlo analysis predicted the scenarios to be more beneficial compared to the baseline less frequently, only around 20% and less than 5% of the time, respectively. The depletion of elemental and fossil abiotic resources categories also has moderate probabilities, ranging from approximately 27% to 76%.

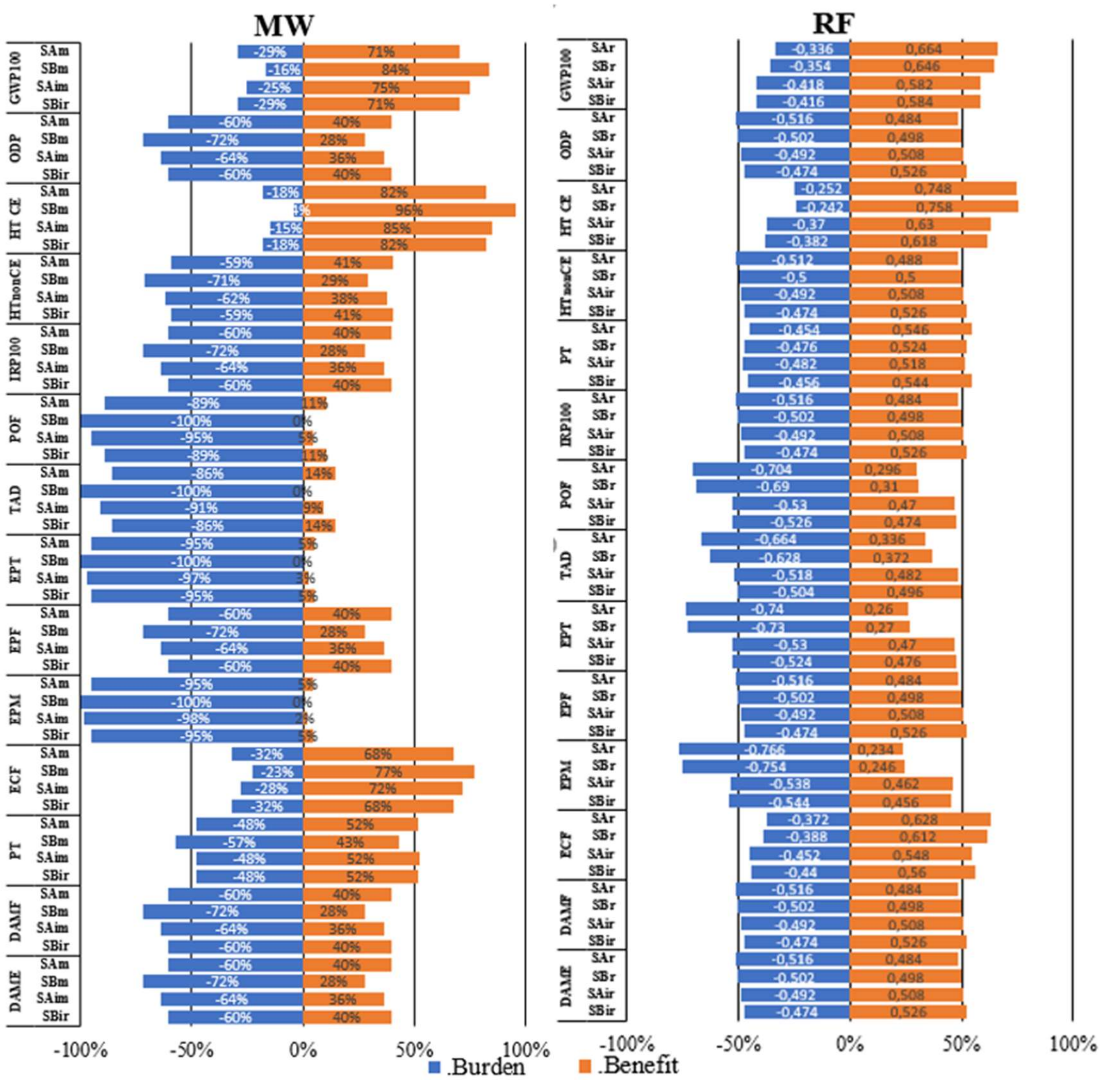


Figure 6*. Monte Carlo simulation presents the times the alternative scenario outperforms the baseline scenario for MW and RF.*

Remarkably, for the impact categories of Photochemical Ozone Formation (POF), Terrestrial Acidification (TAD), Freshwater Eutrophication (EPT), and Marine Water Ecotoxicity (EPM), all scenarios were predicted to be more beneficial 100% of the time, indicating a consistent potential for improvements in these categories compared to the baseline scenario. It is important to note that for each impact category, the SAi and SBi consistently outperform the SA and SB. The "i" scenarios that avoid the installation of a WTS at close distances from the WPC present lower impacts in all the impact categories, making these scenarios more effective at reducing environmental impacts.

***Table 8** Monte-Carlo analysis calculates the times each scenario presents beneficial results than the baseline per impact (results for 1000 runs).*

Impact	*SA*	*SAi*	*SB*	*SBi*
GWP	19.6%	21.6%	21.6%	21.6%
ODP	74.0%	76.6%	76.6%	76.6%
HT CE	3.6%	3.4%	3.4%	3.4%
HTnonCE	72.2%	75.4%	75.4%	75.4%
PT	61.8%	64.2%	64.2%	64.2%
IRP	74.0%	76.6%	76.6%	76.6%
POF	100.0%	100.0%	100.0%	100.0%
TAD	99.6%	99.8%	99.8%	99.8%
EPT	100.0%	100.0%	100.0%	100.0%
EPF	74.0%	76.6%	76.6%	76.6%
EPM	100.0%	100.0%	100.0%	100.0%
ECF	27.2%	29.8%	29.8%	29.8%
DAMF	74.0%	76.6%	76.6%	76.6%
DAME	74.0%	76.6%	76.6%	76.6%

3. Conclusions

The study reveals that the transport of collected waste contributes to 24-30% of the total impacts. Using collection trucks for waste transport benefits the environment when the distances between collection nodes and waste treatment facilities or landfills exceed 56.7 ± 6 km for mixed and 9.7 ± 3 km for recyclable waste. When these distances are extended, WTS become increasingly advantageous, offering additional benefits such as time and human resources savings and reduced maintenance costs. The location of these WTS must comply with various criteria and is heavily influenced by the geographical location of the involved Waste Production Nodes (WPN) and disposal sites, as well as the distribution and configuration of the primary and secondary road networks.

Utilising Life Cycle Assessment (LCA) in conjunction with the generated environmental outcomes can yield precise data for assessing various Waste Transfer Station (WTS) positioning scenarios. Broadening the scope of examined scenarios to encompass additional waste treatment options can enhance the decision-making process related to waste management for both waste management professionals and policymakers.

One key aspect of waste transport is the efficient utilisation of WTSs, enabling managers to optimise trips by ensuring each journey is conducted only when the vehicle is fully loaded. This approach contrasts with conventional vehicles and can significantly reduce environmental impacts. Reducing fuel consumption in waste collection calls for a multi-dimensional approach involving selecting fuel-efficient vehicles, optimising collection routes, promoting efficient and safe driving behaviours, and meticulous vehicle maintenance.

Standard deviation (Std) serves as a valuable indicator, reflecting the variability inherent in the waste collection and transport processes. The decrease in this indicator within the final net results can be interpreted as a beneficial outcome of the calculated results.

This research reaffirms that site-specific aspects and local socio-economic constraints profoundly influence the LCA applied to an integrated waste management system. In current case study, the terrain morphology and the road network to the northern part of the region do not provide alternative routing or flexibility for various WTS alternative locations. Therefore, such results should not be generalised. The implication from a waste management LCA should not be the final results but indication for system improvement looking at several aspects such as cost, terrain, climate, needs, etc. timising waste collection vehicle routes is a critical challenge that needs to be addressed. This issue, called the Waste Collection Vehicle Routing Problem (WCVRP) is significant in waste management. Embracing the Life Cycle Assessment (LCA) approach as a practical decision-making instrument will enable the acquisition of insights into environmental and energy-related impact levels when designing integratedwaste management systems.

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4. Chapter

Modeling the Life Cycle Inventory of a Centralized Composting Facility in Greece.

Abstract

This chapter aims to create a life cycle inventory (LCI) based on the mechanical recycling and composting facility in Chania (Crete, Greece). The objectives are to model the composting unit by mapping the fragmental mass balance between its sub-processes, to monitor the release of C and N as emissions to the environment, and to record the water, electricity, and fuel consumption for the treatment of one Mg of OFMSW introduced in the facility. Two-year waste sampling and data collection are comprehensively analyzed. The outcomes from this study can be used as a tool for waste management practitioners to foresee the outputs and cost of treating OFMSW and source-segregated biowaste.

1. Methodology

The LCI model is developed using actual and local data from two-year monitoring study of the composting unit. Material flow analysis (MFA) software STAN is utilized to fill in missing and not-accessible data, while the LCA software EASETECH is used as a tool for the elemental pathway of C and N in the process. The case study facility is in Chania on the island of Crete (Greece). The composting unit is part of the integrated "Mechanical Recycling and Composting Facility—Landfill" of Chania. It serves 156,585 inhabitants (EL.STAT, 2014) and annually treats approximately 91,500 Mg of urban solid waste (Prefecture of Crete, 2016). The OFMSW is collected from the mechanical recycling facility, and the process is classified as a simplified pretreatment method (Cecchi et al., 2003). Briefly, the comingled waste is fed in the mechanical sorting system, passing throw a bag opener and an automatic rotary sieve (trommel) with a 70 mm diameter mess. A conveyor then drops the undersized material to a magnet for ferrous metals removal, and the remaining is obtained as OFMSW. The oversized material exited from the trommel is driven for recyclables recovery in the facility, while the rejects are disposed of in the nearby landfill.

The system boundary for this study is shown in **Figure 1**. It includes the composting subsystems (aerobic composting tank, refinery unit, open windrows-maturation), which act as the operational processes after the wastes are delivered to the composting plant. The methodology is based on an in-depth analysis of all of the consisting fragments of OFMSW. The waste fragments are comprehensively characterized throughout the subprocesses until their degradation to greenhouse gas (GHG) emissions released to the environment, rejects disposed of to the landfill, or CLO production. Initially, every subprocess and flow are recognized and recorded, while the monitoring period is two years (2018–2019).

The greenhouse gases (GHG) are also considered and studied as C and N transformations along with the main waste flows. The energy is calculated in the form of electricity, fuel in diesel consumed, and the water consumed in the subprocesses. The green waste (GW) consists of tree branches collected from the municipality bulky collection system. It is shredded in the facility and used as a bulking agent in a ratio of 1:4 by volume. The functional unit is 1 Mg of wet mass OFMSW mixed with green waste entering to the composting unit.

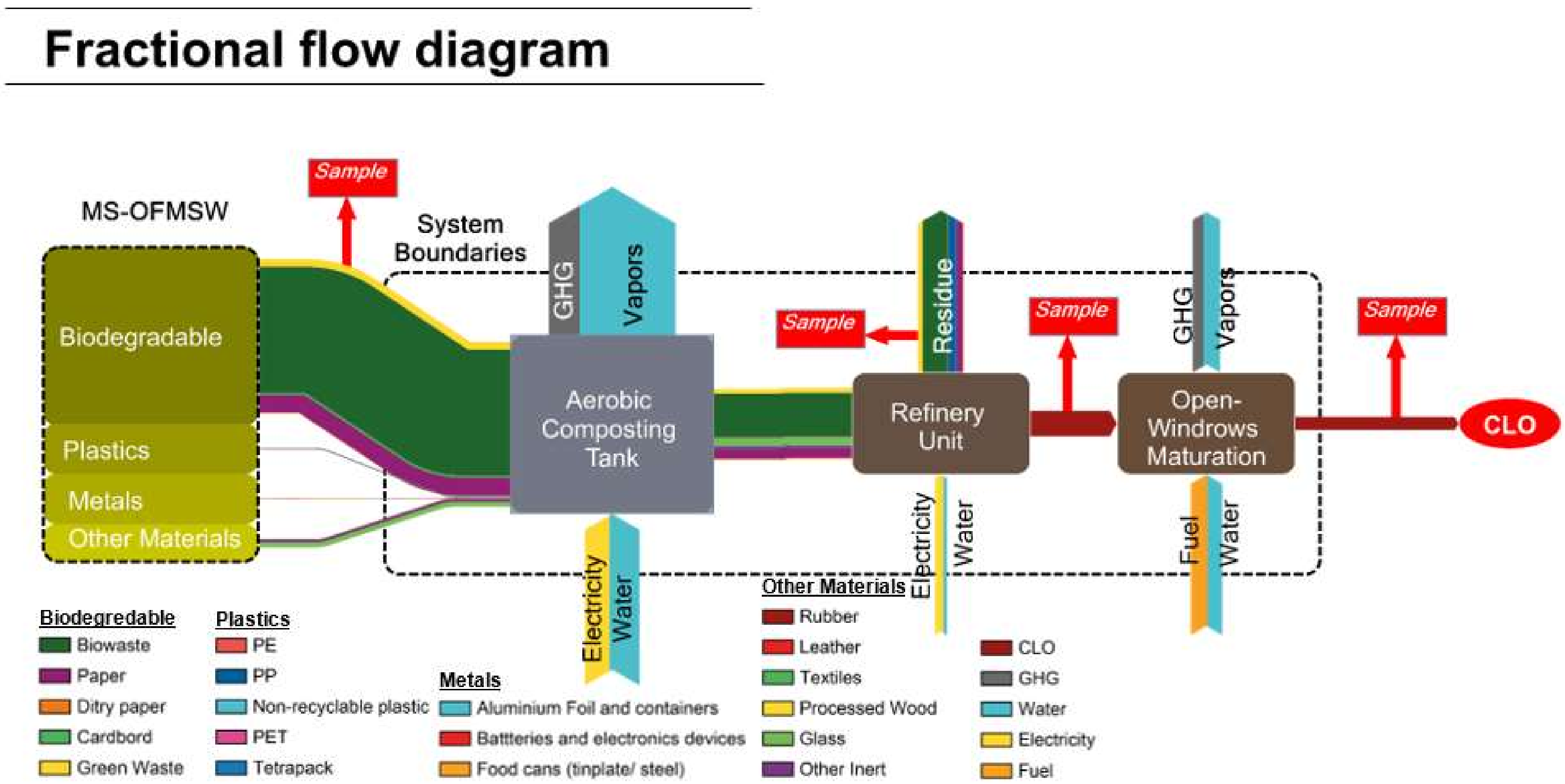

***Figure 1** Sankey's graphic representation of the system and its boundary includes the composting process and inputs/outputs. The subprocesses (aerobic composting tank, refinery unit, open windrows-maturation) are presented. Colored lines represent the different fractions of MS-OFMSW, while the thickness of the lines is proportional to the mass of each fragment. The resources used (electricity, fuel, and water) are shown with yellow and orange arrows, and the emissions to the atmosphere are shown with dark gray arrows. Water addition and evaporation are shown with light blue arrows. The red arrows indicate the sampling points.*

1.1. Composting Units

The composting process is divided into three sub-units: (1) the aerobic composting tank (ACT), which is a continuous flow reactor; (2) the refinery unit (RFU); and (3) the open windrows (OPW) for compost maturation. **Table 1** provides the main information about the composting conditions. The primary composting process in the ACT sustains aerobic conditions with bottom-up aeration and a leachate draining system. A deodorization system with a biofilter is connected to the air exhaust system of the ACT. The turning, water addition, and movement of the material inside the reactor are controlled by an overhead-suspended bridge system with four screw-shaped turners. A fifth screw turner at the end of the reactor transfers the composted materials to conveyor belts towards the RFU. The RFU comprises flip-flop sieves and gravimetric separators to remove bulky and non-compostable materials. The rejects are diverted for landfilling (landfill cover), and the refined material is sent for secondary composting and maturation at OPW. A hook lift truck, a backhoe loader, a wheel loader, and a compost turner handle the transportation and mixing of rejects and maturation windrows. All above vehicles are considered to use diesel (Euro 5 emission standard engines). The unit employs one (1) senior engineer as operation manager, one (1) heavy machinery operator, and two (2) workers daily, while one (1) truck driver and two maintenance technicians (electrician and mechanic) from the nearby MRF are also involved part-time.

***Table 1** Composting conditions and involved personnel for CLO production from mechanically separated OFMSW and green waste.*

Composting Conditions	Value—Factor	
Mixing ratio by volume (OFMSW:GW)	1:4	
Retention time in the composting tank	4 weeks	
Refinery unit	Yes	
Residue after treatment	Used for landfill cover	
Maturation time	6–9 months	
Personel	Number of People	Involvement
Operation manager (senior engineer)	1	Full-time
Front line stuff (workers)	2	Full-time
Heavy machinery operator	1	Full-time
Mentainence mechanic	1	Part-time
Mentainence electrician	1	Part-time
Truck driver	1	Part-time

1.2. Sample Collection and Characterization

Sampling was carried out for two consecutive years (2018–2019) and represented the average composition of each material. The samples were collected from the inputs and outputs of each sub-process, and the sampling points are shown in **Figure 1**. Each sample was then reduced in size using the 'coning and quartering' method at approximately 100 kg and subjected to fragmentation analysis in a nearby area. The samples were sorted into four major categories (biodegradable, plastics, metals, and others-not compostable) and 17 sub-categories based on the research (Edjabou et al., 2015) and the recovery potential at the MRF: biowaste, paper, dirty paper, cardboard, garden waste, soft plastic, hard plastic, non-recyclable plastic, aluminum foil and containers, batteries and electronic devices, food cans (tinplate/steel), rubber, leather, textiles, processed wood, glass, and other inert.

Each sample was analyzed for water content and total solids after drying at 105 °C for 24 h (CSN EN 12048), ash content, volatile solids by igniting the dried sample at 550 °C until steady weight (CSN EN 13039), and Kjeldahl Nitrogen in the dried samples using the Kjeldahl method (CSN EN 13654.01).

1.3. Life Cycle Inventory (LCI) Modeling

Material flow analysis is used to fill in the missing information. The processed weighting data are fed to the sophisticated software STAN v2.6 (4-109ubstance flow Analysis, 2.6), which is developed by the Technical University of Wien (Cencic and Rechberger, 2008; Sevigné-Itoiz et al., 2015) to complete the missing stream flows and convert them in accordance with the functional unit of the study (Henriksen et al., 2019). The results are fed to the EASETECH model (Environmental Assessment System for Environmental TECHnologies, v2.4.5) to calculate for each fraction the degradation factors and transfer coefficients. EASETECH is a waste-LCA model focusing on managing complex waste streams (Christensen et al., 2020), and it can handle the flow of complex heterogeneous fractions in various bioprocess systems. The framework and calculation structure have been described in detail by (Clavreul et al., 2014). In EASETECH software, the degradation is defined as the reduction of organic dry mass during the composting process. The degradation factor of the fragment 'α' for the process Df(α) is defined as the % reduction of the total mass of vs. ascribed to biogenic carbon reduction. In comparison, the transfer coefficient of fragment 'α' is considered the reduction of the total wet mass due to mechanical separation Tf(α).

1.4. GHG Emissions

GHG emissions related to the processes can be defined to direct emissions, indirect upstream emissions, and indirect downstream emissions. Direct emissions are linked to the composting site and its activities, including waste degradation and emissions from machinery used on the site (fuel consumption). The indirect upstream emissions are related to activities for fuel production, provision of electricity used in the site, and the construction of infrastructure and machinery. Indirect downstream avoided emissions are considered from peat substitution for fertilizer production and the carbon sequestration in the soil when compost is applied to land (Boldrin et al., 2009; Favoino and Hogg, 2008). The indirect emissions related to fuel and electricity production were selected after an extensive literature review to reflect the local fuel and energy mixture. **Table 2** shows the emission factors (Efs) used in this study (Fruergaard et al., 2009; Koffi et al., 2017). Mass flow analysis was employed to calculate the gases released during the degradation of the materials for the direct and indirect downstream emission, while EASETECH software native database provided machinery emission factors based on the engine euro standard.

Table 2 *Emission factors (Efs) relevant to GHG during composting*

Type of Process/Emission	Emission Factor	Reference
Provision of diesel oil	0.306 kg CO2-eq/liter diesel	(Favoino and Hogg, 2008)
Combustion of diesel oil	2.7 kg CO2-eq/liter diesel	(Clavreul et al., 2014)
Provision of electricity	0.810 kg CO2-eq/kWh	(Favoino and Hogg, 2008)

1.5. Site-Specific Data

Valuable data from the facility operation are also collected. They concern the primary input and output of each composting unit for the monitoring period, which include weighing data from the daily treated materials, rejects and outputs of the refinery process, daily routes, working hours, annual diesel fuel consumption (L) of every vehicle involved in the composting process, daily electricity consumption from the composting unit (kWh), and daily water consumption (L) in the composting process.

The weighting data are annually averaged for every flow and diverted to the appropriate functional unit and sub-process. The annual electricity consumption is attributed, respectively, to each sub-process and divided by the annual wet mass of the treated material of the specific sub-process. The vehicle diesel consumption is calculated by dividing the annual fuel consumption by attributing working hours and routes for the needs of the composting process. Water is attributed to each subprocess and divided by the mass of the treated materials.

1.6. Life Cycle Inventory Boundaries

The LCI boundaries assume a zero-burden approach (Djuric Ilic et al., 2018; Nakatani, 2014) for the received materials at the entrance of the composting facility. Therefore, the facility environmental footprint is not included in the calculations. This excluded component includes emissions from the construction of the facility, equipment, vehicles, and post-processing of the initial material. Also, this study does not consider the environmental impacts associated with the construction of windrow composting facility (equipment and infrastructure).

1.7. Sensitivity Analysis

Sensitivity analysis is conducted to examine sensitive inputs and analyze whether the assumptions made in the model influence the results (Laurent et al., 2014a). For this reason, this study uses perturbation analysis, and uncertainty propagation methodology (Andreasi Bassi et al., 2017; Clavreul et al., 2012).

Perturbation analysis identifies the most sensitive parameters of the model. The method calculates each parameter sensitivity ratio (SR) and observes the effect of low but countable changes in the results. Every parameter of the studied system is raised, one at a time by 10% (Δ parameter), the new calculated net result is referred as (Δ result). The SR is the ratio between the relative change of the result and the relative change in the parameter. It is calculated as:

$$SR = \frac{\frac{\Delta\, result}{initial\, result}}{\frac{\Delta\, parameter}{initial\, parmeter}} \quad NRS_i = \frac{RS_i}{max|RS_i|}$$

To compare the different SRs in various outputs of the model, the concept of the normalized sensitivity ratio (NSR) has been developed and calculated for each SR. NSR is defined as the ratio of one parameter in one system output divided with the maximum absolute value among all of the SRs in the same output. The concept is a modified adaptation of the methodology of NSRs introduced by (Andreasi Bassi et al., 2017).

Uncertainty propagation consists in propagating input uncertainties to calculate the result uncertainty. Before propagating them, the practitioner chooses a representation for these input uncertainties. The probability theory was adopted in this case study, and the sampling propagation method of Monte Carlo analysis was selected (Hung and Ma, 2009).

2. 3. Results

2.1. Waste Composition

Table 3 presents the material fraction distribution of the mechanically separated organic fraction of municipal solid wastes (MS-OFMSW) and green waste received for composting. Water constitutes 52.5% of the total wet mass which is higher than MS-OFMSW in other studies (Burnley, 2007; Riber et al., 2007). The main compostable fragments can be categorized as biowaste (76.5%), paper-like materials (paper, dirty paper, cardboard) (12.9%), and green waste (4.64%). Since mechanical sorting is based on sizing and gravimetric properties, the presence of foreign non-biodegradable materials is justified. According to (Alvarez et al., 2009), paper waste and cardboard in various proportions consist of 12–27% of the dry mass of MS-OFMSW treated in similar composting facilities in Spain. Carbon content and its origins, biogenic or fossil, are taken from (Riber et al., 2009). These estimates consider that some foreign material may be present in each fragment as suggested by the IPCC 2006 Guidelines (Eggleston et al., 2006). The main greenhouse gases that contribute to global warming are CH4 and N2O, and their release depends on the technology, the waste input, and the management of the process. The above carbon origin is of immense importance in most LCA methodologies since biogenic carbon, when released in the form of CO2 to the environment, is not counted in the impacts, in contrast, when the same portion of the carbon is released in the form of methane in a landfill, for example, it is counted (Christensen et al., 2009; Saer et al., 2013). In summary, the initial material chemical composition without the green waste has a TS of 47.16%, vs. 77.03% of TS, ash content of 22.97, biogenic C of 43.82%, and TN of 2.48%. Although the above values vary compared to literature, they are within the same order (Carabassa et al., 2020; Wei et al., 2017)

Table 3 *ultimate analysis for each fraction, and carbon content (divided to biogenic and fossil origin).*

Fraction	OFMWS Composition (%)	OFMSW Water Content (%)	TS (%)	VS (%)	Ash (%)	C Bio (%)	C Fossil (%)	N(%)
Biodegradable								
Biowaste	76.49 ± 10.55	56.04	37.40	90.00	10.00	54.60	0.60	3.72
Paper	11.40 ± 10.91	33.29	87.00	72.30	27.70	37.60	0.20	0.18
Dirty paper	0.29 ± 0.63	53.30	75.50	91.10	8.90	44.60	0.91	0.30
Cardboard	1.24 ± 17.06	39.33	89.50	84.90	15.10	41.10	0.30	0.24
Green waste	4.64 ± 1.62	47.00	53.00	93.00	7.00	43.02	0.00	0.15
Plastics *								
Soft plastic (PE)	0.30 ± 2.90	28.25	85.89	95.60	4.40	0.41	81.60	0.20
Hard plastic (PP)	0.29 ± 2.05	22.83	96.80	97.80	2.20	0.40	79.50	5.50
Non-recyclable plastic	0.30 ± 3.90	0.00	92.90	94.50	5.50	0.36	70.60	0.50
Metals								
Aluminum foil and containers	0.14 ± 0.47	24.95	81.20	23.90	76.10	13.70	1.52	0.40
Batteries and electronic devices	0.14 ± 0.93	9.72	91.10	14.20	85.80	4.35	4.35	0.10
Food cans (tinplate/steel)	0.15 ± 1.57	7.03	86.82	0.00	100.00	0.00	0.00	0.00
Other materials								
Rubber	0.11 ± 3.58	34.42	92.30	90.30	9.70	52.30	13.10	0.60
Leather	0.11 ± 3.58	34.42	93.30	87.40	12.60	30.70	30.70	0.30
Textile	0.11 ± 3.58	34.42	94.00	96.40	3.60	39.10	13.00	3.20
Processed wood	0.11 ± 3.58	34.42	84.60	96.30	3.70	49.40	0.00	0.00
Glass	3.17 ± 2.05	2.23	99.70	1.20	98.80	0.00	0.00	0.10
Other Inert	1.81 ± 2.87	34.71	63.40	2.30	97.70	0.65	0.65	0.00

* PET and Tetra pack packaging were monitored but not found.

2.2. Material Flow Analysis

The overall process with flow dynamics and mass balance is presented in **Figure 2**. The estimations of C and N flows are displayed in **Figure 3** and **Figure 4**, respectively, assuming that carbon is 99% oxidized to CO2 while nitrogen is released to the air as NH3 at the ACT process (Boldrin et al., 2009). The modeling of the composting system follows all fractions throughout the processes based on two assumptions: (1) the mass can be transferred between processes, and (2) the carbon of biogenic origin in biodegradable materials is biologically degraded to gases with dominant carbon dioxide. The above transfers and transformations are expressed as degradation factors and transfer coefficients.

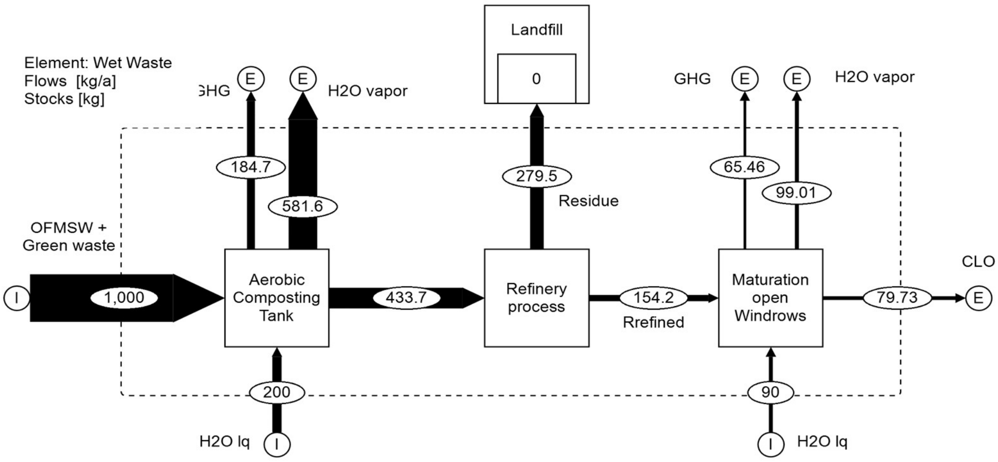

Figure 2 *Sankey diagram of mass balance for OFMSW + green waste treatment in kg (wet waste) (the lines are proportional to the mass of each flux).*

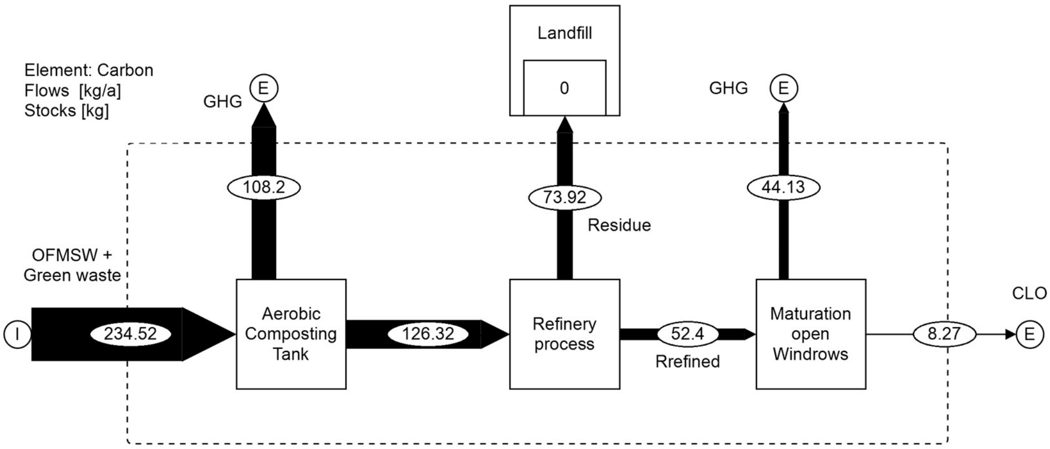

Figure 3 *Sankey diagram for carbon mass balance in kg for OFMSW + green waste composting processing. The lines are proportional to the mass of each flux.*

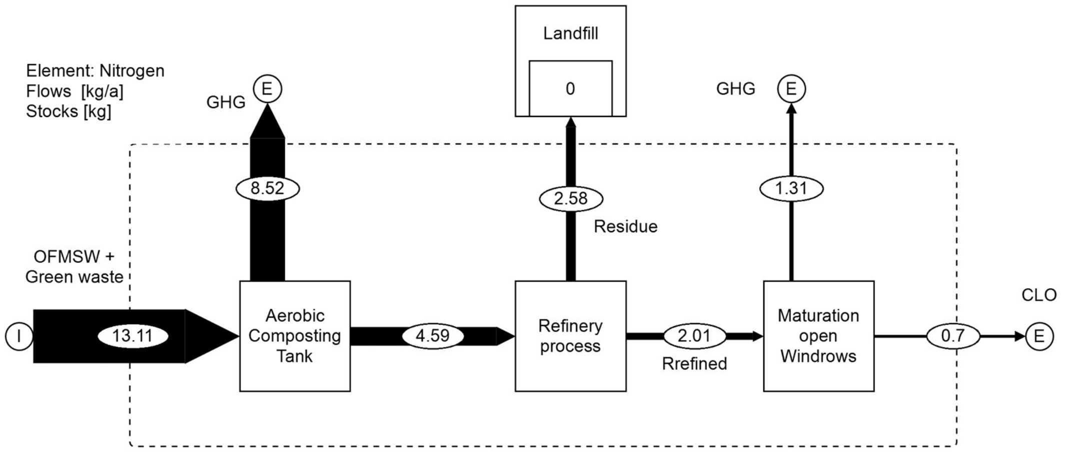

***Figure 4** Sankey diagram for nitrogen mass balance in kg for OFMSW + green waste composting processing (the lines are proportional to the mass of each flux).*

The primary process occurs in the continuous-flow aerobic composting tank (ACT), where the materials enter daily, are mechanically mixed, and transferred across the tank. Controlled conditions are provided with aeration, water adjustment, and temperature management. The processed material that exits daily continues to the refinery unit for separation. The calculated retention time of the materials in the composting tank is four to five weeks. Several studies follow a general approach when modeling a composting system and consider the treated materials as a single homogenous mixture appointing one degradation factor (Graça et al., 2021; López et al., 2010). That is justified since most studies refer to source-segregated OFMSW (Campuzano and González-Martínez, 2016; Graça et al., 2021; Gutiérrez et al., 2015; Sailer et al., 2021) and only a few to mechanical sorted OFMSW (Cecchi et al., 2003). This research considers the individuality of each of the consisting fragments and its different degradation rates. The degradation factors are calculated utilizing MFA methodology with data of the ash content and mass loss of each fragment in the input and output at the ACT.

In **Table 4**, the degradation factors are referred to the biodegradable materials. Paper-like materials present the higher degradation factor with printed paper (61%), cardboard (51%), and dirty paper (43.3%). These materials present high VS, and they consist primarily of cellulose and lignin, organic polymers difficult to decompose.

However, the high decompositions are justified by the screening, mixing and the elevated temperatures found during the thermophilic phase of composting process that contribute to the rapid degradation of lignocellulose (Tuomela et al., 2000). Similar high degradation rates have been observed elsewhere (Alvarez et al., 2009) concerning various paper-like materials present in OFMSW composting. It is also stated that if the paper-like materials do not exceed 27% of OFMSW, degradations of 36–65% are feasible in controlled conditions with a retention time of 45 days. Concerning biowaste, the degradation factor has an average 48%, while the green waste 15%.

The above results suggest that the retention time in the ACT is not sufficient for the complete decomposition of the biodegradable materials of biowaste. Green waste serves as a bulking agent and is shredded to medium-sized particles presenting a low degradation factor (Yuan et al., 2017). Physical characteristics such as bulk density, particle size, and porosity are important factors for fragment decomposition level. The structure of green waste (containing lignocellulosic materials) appears difficult decomposition and requires specific lignocellulosic microorganisms (and enzymes) to improve it degradation and retention time more than 12 weeks (Cerda et al., 2018; Davis, 2005). In order to produce mature compost, (Boldrin et al., 2009) stated that degradations of 40–83% of the carbon contained in the biowaste are required, while (Komilis and Ham, 2006) reported 62–66% and 66–77% degradation of carbon for garden waste and food waste, respectively.

Table 4 Degradation factors (% ww) for the volatile solids of waste fractions in the aerobic composting tank process.

Fraction	Degradation Factor (%)
Biowaste	48
Paper	61
Dirty paper	43.3
Cardboard	51
Green waste	15
Soft plastic (PE)	10
Other inert	5
Other materials	0

Plastic materials present close to zero degradation for this process; only LDPE abandoned in the form of shopping bags can be accounted to have a 10% degradation. Although LDPE films have presented some degradation only in the harsh environment of the composting process (Davis, 2005), and the degraded portion is meager. The mechanical processes employ throughout the composting process, such as turning mixing and screening, can result in polymers being sheared into smaller fragments during the conventional composting process and could explain the above degradation factor, as it is often apparent in household and commercial organic waste (Gui et al., 2021; Vithanage et al., 2021). An additional issue that must be considered is the rapid increase in biodegradable plastic materials that have started to replace the traditional PE film. Their biodegradability is dependent on the composting conditions and the chemical composition of each material (Narancic et al., 2018).

The second sub-process (refinery unit-RFU) is a mechanical separation stage based on sizing the material using a ‘flip-flop’ sieve with 10 by 10 mm mesh holes, followed by a gravimetric air separator in line with a gravimetric air cyclone to collect the lightweight material. **Table 5** presents the transfer coefficients for each material. Bulky and heavy materials are mainly rejected into residue. Water content is critical in this step since it adds excess weight if not adjusted correctly in the previous process, leading to discarding compostable materials as residue. The fact that the exiting material is collected by its size and gravimetric properties and not its chemical characteristics is advantageous. It provides optimal mechanical characteristics on the collected materials, although it does not prevent the infiltration of unwanted dissolved chemicals such as heavy metals. The collected materials have TS (75.03%), vs. (75.4) of TS, and 19.3% TS carbon of biogenic origin. The rejected material of the process consists of bulky and non-compostable materials. The same principle is followed for the transfer coefficients of the refinery process, and the above assumptions allow experimentation with variations of composition with the same system, providing a handy tool for further research.

Table 5 *Refinery process transfer coefficients total mass (% ww) for open windrow composting and maturation.*

Fraction	Transfer Coefficients (%)
Biowaste	45
Paper	60
Cardboard	100
Green waste	5
Soft plastic (PE)	2
Hard plastic (PP)	1
Non-recyclable plastic	1
Other materials	0

The material that continues to the final composting/maturation stage has a homogenous texture; the origin fractions are hard to recognize, only some paper-like remains, and some wood fraction with particle size lower than 10 mm are notable. The total vs. is high (69%). The material is accounted as concentrated biodegradable fraction, which justifies the intense composting stage, followed by a prolonged maturation state (composting windrows). It must be stated that open-windrow composting can be challenging owing to variable weather conditions that advance or delay the composting process. Intensive mechanical mixing and constant windrow temperature monitoring which occurs once a week during this process minimize the number of anaerobic pockets in the composting mass. However, it is reasonable to assume that an inevitable release of CH4 occurs. Hence, adopting the lowest emissions values, 0.8–2.5% of degraded C is released as CH4, which seems reasonable [11]. Concerning nitrogen-based GHG, (Hellmann et al., 1997) stated that there is no production of N2O during the thermophilic phase since autotrophic nitrifier activity ceases above 40 °C. Since the maturation phase is considered a continuation of the primary composting process, GHG production is only scarce at the final stages of the process. For this reason, 0.1–0.7% of degrading N is accounted to transform to N2O. Mixing and water addition ensure a partially controlled maturation elongating to six or nine months until the desired physicochemical characteristics are reached.

The quality of MSW compost is dependent on many parameters, including the composting facility design, feedstock source and proportions used, composting procedure, and duration of maturation (Hargreaves et al., 2008). The maturation typically requires minor active management. It is a crucial final stage that facilitates the conversion of potentially toxic NH4 to NO3, allows the loss of phytotoxic volatile compounds, and stabilizes the microbial community. At this state, mesophilic fungi and actinomycetes colonize the compost, which is thought to be responsible for the breakdown and transformation of humic substances and lignin. Although, maturation is a vital stage frequently given insufficient time, or is even missed out altogether, to save space and increase the throughput of composting plants. In this case study, the corresponding sub-process can be chronically adjusted depending on the aiming physicochemical characteristics of the final product. The average decomposition rate is calculated to be 75% of the total volatile solids of the initial material. The resulting CLO has 37% water content, vs. of 57%, while the C and N contents are calculated to 56% and 1.9%, respectively. Carabassa et al. (2020) (Carabassa et al., 2020) presented CLO with similar physicochemical characteristics ranging from 65 to 70% TS, 44.5 to 64% VS, and 1.4 to 2.17% N, while (Malamis et al., 2017b) produced CLO with similar characteristics.

2.3. Mass Balance

Material and substance flow analyses are performed based on mass balances. The composting unit is then built graphically and displayed as Sankey plots. **Figure 2** presents the mass flows of wet waste throughout the processes and the loss of material and compounds to the atmosphere (in kg). The water content is significant for the proper accounting of the total mass balance. Since it is added during composting and maturation and accounts for 200 L per Mg of treated materials in the ACT and 90 L per Mg of treated materials during maturation, the quantities are not insignificant (while its use is threefold). It provides temperature control by reducing heat due to its evaporation, it acts as a medium for the dilution and exchange of elements. Finally, it regulates the aerobic conditions in the composting mass. During the aerobic process, the evaporated water and mass loss is calculated to 766.3 kg per Mg w/w, plus 164.5 kg for the maturation state, while an amount of 279.5 kg is rejected. The resulting CLO material is calculated to be 79.73 kg.

The carbon balance is presented in **Figure 3**. During the two sub-processes where organic matter degradation occurs, 64.96% of the initial carbon is released into the environment. The primary composting process releases 46.14% of the initial carbon in gaseous form. A significant portion of the initial carbon (31.52%) is diverted to the landfill and contributes to carbon sequestration (Kumar and Sharma, 2014; Staley and Barlaz, 2009). Finally, 3.52% is included in the CLO destined for land use. Several studies have investigated the degradation of organic matter and C fate during composting. Production of mature compost requires degradation of 40–83% of the carbon contained in the compost (Boldrin et al., 2009). Most of this carbon is emitted as biogenic CO2, and a relatively small portion is emitted as CH4 created in anaerobic pockets in the composting mass.

The total nitrogen loss during the main composting process (ACT) is 64.99% (**Figure 4**). A portion (19.68%) of the initial nitrogen is landfilled, and 5.3% is bound to the CLO produced mass. The rest is released in gas form during the maturation phase. The controlled conditions in the ACT provide a stable temperature profile of 45–65 °C, favoring the thermophiles phase. The above conditions inhibit the nitrification of produced ammonium to NO2 while the dissociation constant (pKa) of NH4+ decreases with increasing temperature, meaning that higher temperatures favor evaporation of

NH3. Eventually, ammonia is the most emitted form of N (Amlinger et al., 2008; Andersen et al., 2011). However, other by-products have not been investigated (i.e., for ammonia the oxidized forms NO and N2O are not considered, although aerobic microorganisms may form them). These gases potentially impact the environment. NO may result in ozone depletion in the stratosphere, and N2O is an effective greenhouse gas (Clemens and Cuhls, 2003).

2.4. Estimation of Resources Consumed

2.4.1. Electricity

Aeration, deodorization, mixing, transfer, and refining of compost are the main electricity-consuming processes in the ACT and RFU resulting in a 34.56 kWh electricity consumption per Mg of the wet treated material (**Table 6**). This number is the average electricity consumption for every sub-process for a given volume of the treated material. According to (Boldrin et al., 2009), electricity consumption depends mainly on technology use and is higher on closed composting systems, especially reactor technologies, ranging between 9 and 65 kWh/Mg w/w versus 0.023–19.7 kWh/Mg w/w for open technologies. The research by (Liu et al., 2020) attributes a fourfold electricity consumption to reactor technology than windrow composting, stating that the benefit of reactor composting is covered from N loss by preventing organic contaminants, higher degradation rates, and lower composting periods. Another research in large-scale bioconversion systems based on the aerobic treatment of organic waste implies that the reduction of the produced leachate due to controlled air supply is reduced by 75–99% (Themelis and Kim, 2002).

***Table 6** Heavy machinery involved in the composting process (fuel consumption), electricity, and water consumption*

Row Labels		**Process Attributed**	**Unit/Mg of Material Treated in the Corresponding Process Material**
Backhoe loader (L of diesel)	MIDI wheel loader (liters/Mg)	Maturation	2.201
Wheel loader (L of diesel)	Wheel loader (liters/Mg)	Maturation	1.096
Other tractor (L of diesel)	Hook lift (liters/Mg)	Maturation	0.311
Other drivable machines (L of diesel)	Compost turner (liters/Mg)	Maturation	0.098
Marginal Electricity Consumption (kWh)	Electricity (kWh/Mg)	Aerobic composting Tank and Refinery	34.56
Water consumption for composting process in aerobic composting tank	liters/Mg entering main composting	Aerobic composting Tank and Refinery	200.0
Water consumption for maturation state in open windrows	liters/Mg material in windrows	Maturation	583.7

2.4.2. Fuel

The transportation of the residue to the landfill and the refined material to the maturation area employs a hook-lift truck consuming 0.311 L per Mg of the transferred material. For the management and treatment of the maturing windrows, two wheel-loaders, and one compost turner are involved (**Table 6**). The fuel consumption for each vehicle is calculated to be 2.201, 1.096, and 0.098 L of diesel consumed, respectively, per Mg of maturing material. The engine technology for all of the vehicles follows the standard of Euro 5 as it has been classified from the European emission standards for heavy-duty diesel engines. The conversion of fuel consumption to the initial wet mass of MS-OFMSW is 0.658 L of diesel per Mg, while the literature review presents a range of 0.4–0.5 L per Mg for similar processes (Boldrin et al., 2009).

2.4.3. Water

Water consumption is 200 L per Mg for the ACT and 90 L per Mg of refined material during windrow composting. In many LCA methodologies, water consumption is not included [11]. During the aerobic tank composting, the water addition is constant to substitute the water loses of high composting rates and prevent the compost from overheating, while the only water source is the embedded irrigation system. On the other hand, open composting is exposed to weather conditions and precipitation contributes to the windrows irrigation system.

2.4.4. Sensitivity Analysis

Sensitivity analysis is performed to check the model's robustness and assess overall uncertainty (Groen et al., 2014). The parameters tested in the perturbation analysis include the degradation factors for aerobic composting and the transfer coefficients. All parameters are raised, one at a time, by 10% and the resulting change in the three key outputs of the system: the two exits of the refinery process (rejects and refined materials) and the at the end of maturation phase (produced CLO) are recorded as the quantity of wet mass in contrast to the initial quantities. The results as NSRs are presented in **Table 7**. The resulting NSRs reveal the sensitivity of the model to the degradation rate of biowaste and the maturating CLO.

Table 7 Perturbation analysis of NRSs for the main parameters of the model

	Refinery Output	Rejects Output	Final CLO
Degradation Factor			
Aerobic composting tank			
Biowaste	0.8	0.9	0.2
Paper	0.2	0.1	0.1
Dirty paper	0.0	0.0	0.0
Cardboard	0.0	0.0	0.0
Green waste	0.0	0.0	0.0
Soft plastic (PE)	0.0	0.0	0.0
Other Inert	0.0	0.0	0.0
Transfer coefficient			
Refinery process			
Biowaste	0.0	0.0	0.0
Paper	1.0	1.0	0.5
Dirty paper	0.3	0.3	0.2
Cardboard	0.0	0.0	0.0
Green waste	0.0	0.0	0.0
Soft plastic (PE)	0.0	0.0	0.0
Degradation Factor			
Open windrows			
Degradation Factor	-	-	1.0

The second part of the sensitivity analysis is performed regarding the overall uncertainty propagation for the system outputs. Monde Carlo simulation (MCS) is initialized to generate pseudo-random numbers from the set of the studied parameters. The model degradation parameters and transfer coefficients are attributed with uncertainties of 10% in the form of normal distribution, and the MCS iteration value is set to 10,000 times to obtain the sample distribution of the output parameter (Groen et al., 2017, 2014; Helton et al., 2006; Henriksen et al., 2019). The results are 6.1% variation for the rejects output, 9.4% for the refinery output, and 15% for the compost output.

4. Discussion on Chapter 4

This chapter identifies the dynamics of recovering significant quantities of biogenic carbon and benefitting from the produced compost-like output as a soil conditioner. Although the quality of the produced CLO has several uncertainties due to the origin of the materials, a significant amount of the initial materials is discarded as rejects and usually ends up in landfills. Considering the circular economy perspective, the sustainable treatment of OFMSW requires it to be separated from residual waste at the source to eliminate contaminants remaining in the initial materials (European Environment Agency et al., 2020). In Spain, samples of produced CLO from 10 MBT plants in Castile and Leon showed heavy metal concentrations below the limits set by the national legislation. However, the percentage of inert impurities, such as plastic or glass, was excessively high, exceeding in some cases the legal limit [77,78]. The same issues were concerned for the CLO produced from the MBT of Attica (Malamis et al., 2017a). The elimination of reject based on the absence of the contaminants mentioned above increases the produced quantities since, in other cases, rejects could be further processed. The restrictions applied to CLO uses do not apply for source segregated biowaste produced compost. The use of the produced material to agriculture, soil improvement, and fertilizer substitution should not be overlooked.

In early 2017, Europe had about 570 active MBT plants with a treatment capacity of 55 million tons (Doing, 2017). According to the 2020 report from the European Environment Agency (EEA) concerning bio-waste treatment in Europe, the most common treatment methods for biowaste, in line with circular economy principles, were composting and anaerobic digestion. The second was the most preferable in some cases due to benefits from the recovery of material and energy. However, the 22 EU countries average favor composting, with Greece utilizing only composting (van der Linden and Reichel, 2020). In the highest biowaste treatment capacities ranking, Sweden and Croatia present more than 370 Kg/capita, followed by Austria, Slovenia, and France near 300 kg/person, while Greece shows the lowest capacities. In the same ranking, comparing source segregation (versus not separately collected biowaste), Greece mainly applies the collection of mixed waste. At the same time, Austria leads the trends with close to 200 kg/capita on separate biowaste collection. Concerning Greece until 2020, six MBT facilities had been constructed and in operation, and ten more are under construction (Hellenic Repablic, 2020).

The LCA study performed by Abeliotis (Abeliotis et al., 2012) for the MBT of west Attica was based on data provided from the regional administration of Attica, and the native database of the LCA software was used to calculate the produced emissions. In a global LCA review (Laurent et al., 2014a) until 2014 (222 case studies), the dominant monitored waste stream was household mixed waste, 70% of the studies concern cases in European countries. Most of the inventory data sources were taken from the literature without addressing the appropriateness of the data used, such as representativeness in time or space of the extracted data compared to the studied system. Composting was the most favorable among the biological treatment methods used in 74 of the above studies. In contrast, anaerobic digestion was used in 53 cases.

In more recent studies, the life cycle inventory analysis is the most time and resource-demanding for the LCA partitioners (Laurent et al., 2014b). The evolution of advanced LCA software with ready-made modules for the composting process may save time and resources. However, it may lead to fault results making mandatory an evaluation step of the primary LCI data.

The goals for a more circular economy in EU by the new revised Waste Framework Directive introduced a new requirement for bio-waste separation. By 31 December 2023, bio-waste must either be separated and recycled at the source or collected separately and not mixed with other types of waste (European Parliament and Council, 2018). In addition, as of 2027, compost derived from mixed municipal waste will no longer count towards achieving compliance with the recycling targets for municipal waste. From an LCA perspective, the impacts of a transition from mechanical sorting to source segregated biowaste collection has not yet been studied.

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5. Chapter

Material flow and environmental performance of the source segregated biowaste composting system

Abstract

In this chapter, a Life cycle assessment (LCA) is performed to investigate the environmental impacts of two alternative approaches in a biowaste management system. The system inventory follows the previous chapter's research and is based on actual data and on-site sampling for two consecutive years at the mechanical and biological treatment (MBT) facility at the prefecture of Chania (Greece). The facility pertains as MBT for household waste and material recycling (MR) for the recyclable fractions in two different process lines. The mass balances and environmental performance are assessed from waste generation to end-use. The LCA and ReCiPe 2016 methodology estimate the endpoint environmental impacts on human health, ecosystem quality and resource scarcity. The results show that biowaste source segregation in an integrated waste management system not only significantly benefits its recoverability potential it also improves its environmental performance. Impacts on human health (HH) has been reduced by 4.6 times, on freshwater ecosystem quality (EQf) by 6.3 times and resource scarcity (RS) usage by 2.5 times when biowaste is combined with compost production and use, material recovery and reprocessing for fertilizer and raw material substitution.

1. Methodology

1.1. Case study

The Prefecture of Chania (Crete) waste management system is used as a case study. Chania was one of the first to a ties sustainable waste management in Greece following the directive Landfill of Waste 1999/31/EC. It has enforced a four-stream segregation system for the MSW collection. It is based on colour-coded kerbside containers: i) the Recyclable Fractions (RF) of Municipal Solid Waste that includes separated dry fractions (primary plastics, aluminium, tins and cans, paper, and cardboard) are collected in blue colour containers; ii) Packaging Glass that includes bottles and jars that collected in yellow-coloured bell shape collection containers; iii) the household wastes (HHW) is collected in green colour containers; and iv) green and bulky waste consisting of tree branches, garden waste, furniture, and oversize materials are loaded to open collection trucks. The total waste generation was 113,271 tonnes and 117,296 tonnes for 2018 and 2019, respectively (data provided by the waste managers).

All waste materials are transported to the Mechanical Recycling and Composting Facility – Landfill (25 km away from Chania city center). Mechanical Recycling is designed to treat HHW and RF separately, one at a time, in the same infrastructure. It uses mechanical sieves, optical separators, density separators, and magnets to collect cardboard, printed paper, Tetrapak packaging, PE, PP, PET plastics, aluminium (Al), and ferrous (Fe) material during its daily operation. The recovered recyclable materials are pressed and baled for storage temporally and then loaded to tractors to be sent for recycling and substitute new materials in industries outside the Prefecture of Chania.

The HHW is fed in a bag-opener and two rotating cylindrical sieves, and it is separated by size into three fractions: i) the oversize materials; ii) the medium size material that is subjected to the same process as the RF for the recovery of plastics, paper, and metals; and iii) the undersize < 70 mm materials after passing through an electromagnet for ferrous metals removal consist the organic fraction of MSW (OFMSW). The resulting OFMSW is mixed with shredded green waste and delivered to the composting unit. The residues from all processes are disposed of in the nearby landfill. In the composting unit, the primary composting occurs in the continuous flow aerobic composting tank for five weeks. The exiting material is subjected to screening and passing throw flip-flop sieves in line with density separators to reject non-compostable materials (Chazirakis et al., 2022). The fine compost-like output (CLO) continues for maturation in open piles for several months, depending on the weather conditions. The schematic diagram of the case study waste management is presented in supplementary information **Figure 1**.

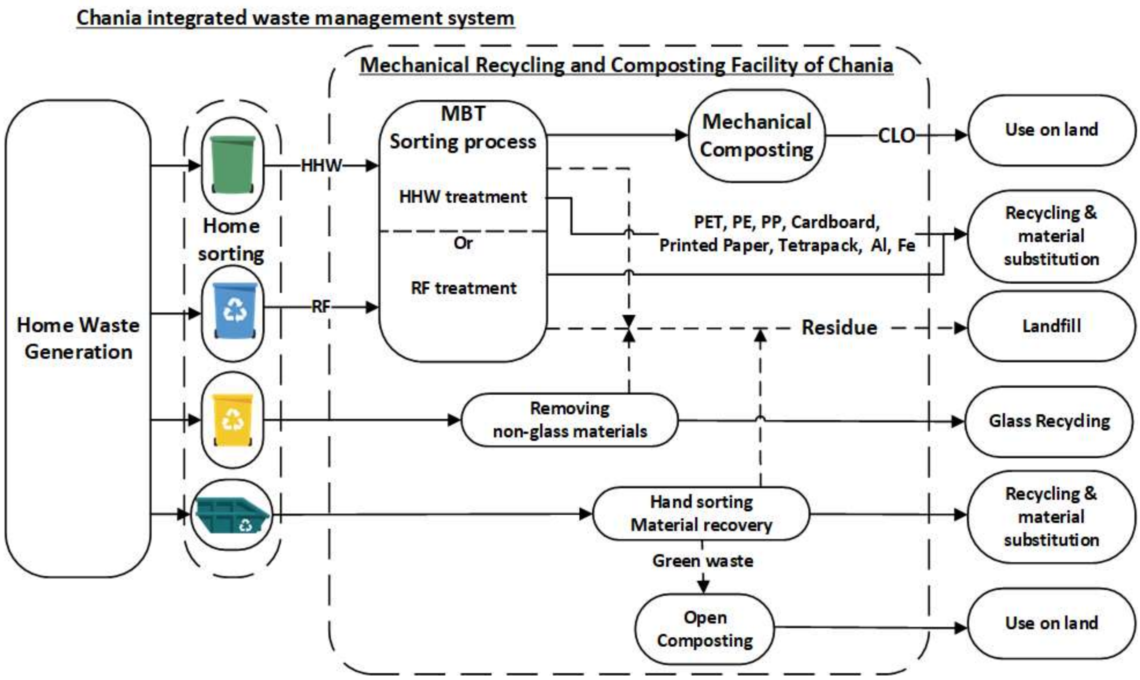

***Figure 1** Chania integrated waste management system*

1.2. Waste generation flow analysis and site-specific data collection

Accurate data about the facility operation are gathered concerning the main inputs and outputs for a certain period (***Table 5***). The data used in this study are obtained from periodic sampling in the collection trucks and the inputs and outputs of the facility during 2018-2019. They include weighting data from the daily treated materials, rejects, and outputs of the sub-processes, daily routes, working hours, annual diesel fuel consumption (L) for every vehicle involved, and daily electricity consumption at the facility (kWh). The weighting data are annually-averaged for every flow and diverted to the appropriate functional unit and sub-process. The annual electricity consumption is attributed to each sub-process and divided by the annual wet mass of the treated material for the specific sub-process. The vehicle diesel consumption (L) is calculated by dividing the yearly fuel consumption by working hours and routes for the needs of each sub-process. All environmental flows are considered for 100 years. Products generated within the system are modelled using attributional modelling. (Edwards et al., 2018a).The composition of HHW and RF is monitored through monthly sampling in the collection trucks arriving at the facility. Sampling is held based on the CSN EN14899 sampling methodology. All fractions are measured for humidity and ash content following CSN EN 12048 and CSN EN 13039.

The samples are sorted into four main categories (biodegradable, plastics, metals, and other not compostable) and 19 sub-categories (biowaste, paper, dirty paper, cardboard, garden waste, branches, plastic bottles, soft plastic, hard plastic, juice cartons (carton/plastic/aluminium), non-recyclable plastic, aluminium foil and containers, batteries and electronic devices, food cans (tinplate/steel), rubber, leather, textiles, processed wood, glass and other inert. The chemical composition of the waste fractions is taken from EASETECH software native database (Clavreul et al., 2014). Mass flow calculations are conducted on a dry matter basis, and water content is attributed based on the corresponding literature to overcome the above issue. Mass flow analysis is performed to calculate the different waste separation streams. The specialized software STAN v2 developed at the Wein Technical University is used to complete the missing waste stream flows and convert them according to the functional unit of the study (Sevigné-Itoiz et al., 2015). The EASETECH software is used to carry out the LCA study (Clavreul et al., 2014). The software allows modelling the reference flow as a collection of material fractions, tracking their composition throughout the modelled technologies (similar to material flow analysis)

1.3. Fuel and electricity calculation

Various vehicles (wheel loaders, forklifts, and hook-lifts) are used in the facility. All vehicles are considered EURO 4 emission standard concerning the released emissions. Annual fuel and daily weighting data are collected for every involved vehicle. Since some vehicles are used for more than one process (RF or HHW treatment and composting), the fuel is allocated accordingly. Two methods are used to attribute fuel consumption, depending on the vehicle use, material transport, or material handling. For vehicles used in material handling, the annual fuel consumption is divided by the yearly sum of the weight of the treated material in the corresponding process. The two-year average consumption is used in the inventory. For vehicles involved in material transport, the annual fuel consumed is allocated based on the total number of trips for each process and then divided by the weight of the treated material in the specified process. The consumption of each vehicle for every treatment process is calculated in L of diesel per Mg of treated material in wet weight.

The electricity consumption (kWh) is calculated based on actual measurements using each process's annual consumption data. The above consumptions are converted into consumption per Mg of treated material using the annual daily input for each process. The resulting data are then modelled in the EASETECH LCA software.

1.4. Goal and scope definition

A consequential LCA is developed to compare biowaste separate collection environmental performance. The LCA methodology is designed following the 'ISO 14040: Principles and Framework' and 'ISO 14044: Requirements and Guidelines for international environmental standards' (ISO 2006). The LCA boundaries include foreground and background processes (generation of HHW, segregation process for OFMSW or biowaste, recovery of recyclables in the mechanical facility, substitution of raw materials, use of compost on land or fertilizer substitution, rejects end up in the landfill).

A zero burden approach is also considered where all processes before the waste generation are identical (Gala et al., 2015; Liu et al., 2017; Nakatani, 2014) and do not affect the directional outcomes of the study. The addition of biowaste segregation, collection, and transportation does not affect the studied system since the total collected weight of waste is the same. The functional unit is 1 Mg of generated waste (wet basis). The environmental impacts of all scenarios are estimated and compared. In addition, a sensitivity analysis is performed to evaluate the robustness and integrity of the results.

1.5. Scenario description

Three scenarios shown in **Figure 2** are developed and compared. The scenarios follow the treatment of 1 Mg household mix waste generated in the prefecture of Chania (composition is shown in **Table 2**). In the first scenario (baseline scenario or S0), 1 Mg of household wet waste is collected in the green container and delivered to the MBT for treatment. The process uses fuel, electricity consumption, and recovery rates from the HHW treatment process of the Chania case study. The outcoming materials are OFMSW (81.5% biowaste), recyclable materials (plastic, paper, ferrous, aluminium), and residues sent for landfilling. The OFMSW is then biologically treated in the composting unit for CLO production and use in soil. The recovered recyclable materials are sent for recycling and raw material substitution.

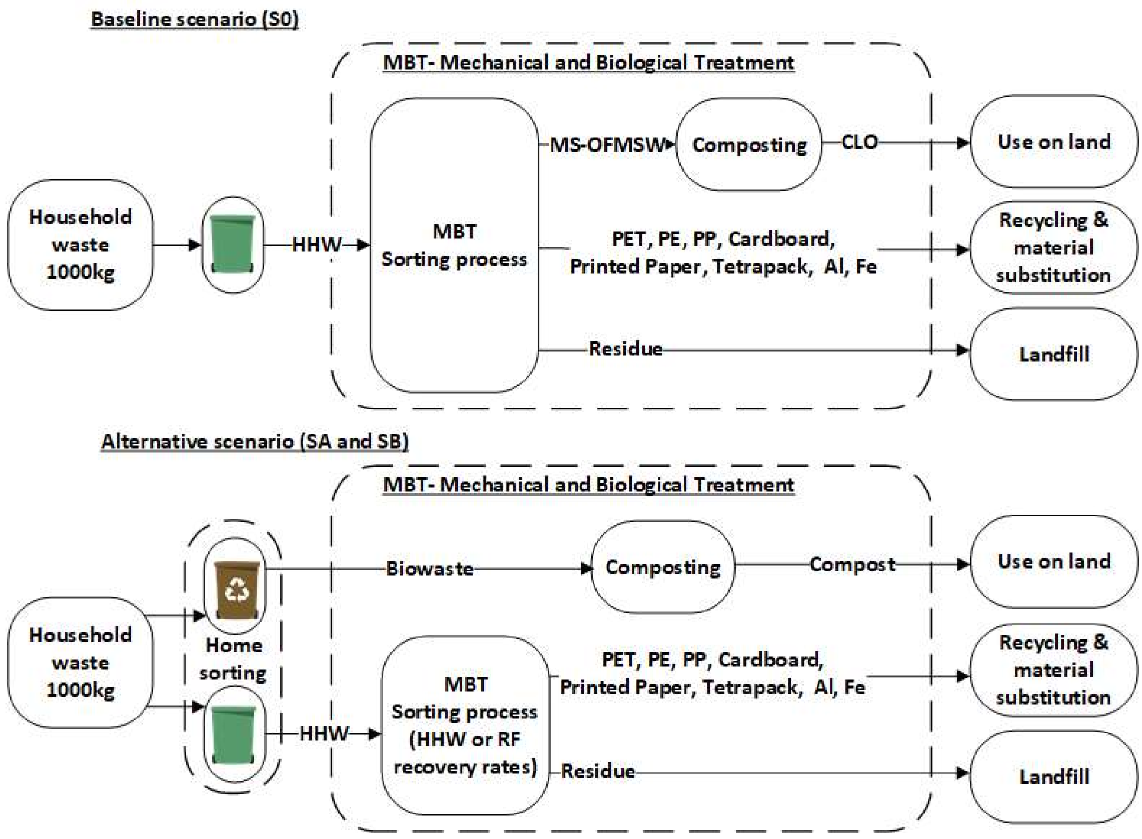

Figure 2 Flow diagrams of Baseline (S0) scenario with recyclable material recovery for raw material substitution and OFMSW recovery for CLO production for land use. The alternative scenarios SA and SB use source segregation for biowaste and compost production for land use and fertilizer substitution. The remaining HHW is processed for recyclable recovery and raw material substitution in the SA scenario uses recovery rates, energy and fuel consumption based on the HHW process of the case study, while in the SB scenario uses recovery rates, energy and fuel consumption based on the RF process of the case study.

The second scenario (SA) inserts the brown container used for biowaste segregation from the HHW stream. The quantity of the collected biowaste is considered the same as in S0 (equal to OFMSW). Similar collection rates have been reported elsewhere (Bueno et al., 2015). No extra burdens to the environment are accounted for sorting, collection, and transportation for the new stream. The biowaste arrives at the facility separately and is directly composted in the composting unit, producing compost used in agriculture for fertilized substitution. The remaining home waste is collected in the green container and processed in the MBT using the same configuration (recovery rates, fuel, and electricity consumption) as in the S0 scenario for HHW treatment. The recovered recyclable materials are sent for recycling in recycling facilities for raw material substitution. The residues from all the processes are disposed of at the nearby landfill.

Table 1 *Data input to LCA-EASETECH assessment software*

Functional unit 1000kg	
Foreground data	
HHW composition	in situ measurements- CSN EN14899
Waste characterization	
Water and Ash content	in situ measurements ISO CSN EN 12048, CSN EN 13039
Elemental composition	(Clavreul et al., 2014)
Home sorting	in situ measurements/collected data
MRF mixed/recycling configuration	
electricity consumption	in situ measurements/collected data
fuel consumption	in situ measurements/collected data
separation coefficients	in situ measurements/collected data
Composting modelling	(Chazirakis et al., 2022)
Background process	
Inventory and substitution data	
Emission factors (Efs) relevant to GHG	
Provision of diesel oil	(Favoino and Hogg, 2008)
Combustion of diesel oil	(Clavreul et al., 2014)
Provision of electricity	(Favoino and Hogg, 2008)
Landfill modelling	(Manfredi and Christensen, 2009)
Recycling substitution	(Cremiato et al., 2018)
land use of CLO/compost	(Bruun et al., 2006)
Fertilized substitution	(Edwards et al., 2018b)

The third scenario (SB) has the same concept as the SA scenario, following a separate biowaste collection (brown container) and green container processing for recyclable materials recovery. However, it assumes that the biowaste sorting from the HHW changes its component properties and eventually improves the recoverability of the recyclable materials of this stream. For this scenario, the MBT uses the RF configuration (recovery rates fuel and electricity consumption) based on findings for the case study of Chania.

1.6. Life Cycle Inventory (LCI)

1.6.1. 2.6.1 Foreground processes

1.6.1.1. Mechanical and Biological Treatment Facility (MBT)

The MBT facility operates daily and processes 32,000 Mg of HHW and RF annually in the HHW configuration. Both streams are processed in the same line, but not at the same time. The mechanical recovery units use a combination of physical processes to separate the materials, while the OFMSW is aerobically composted in the composting unit.

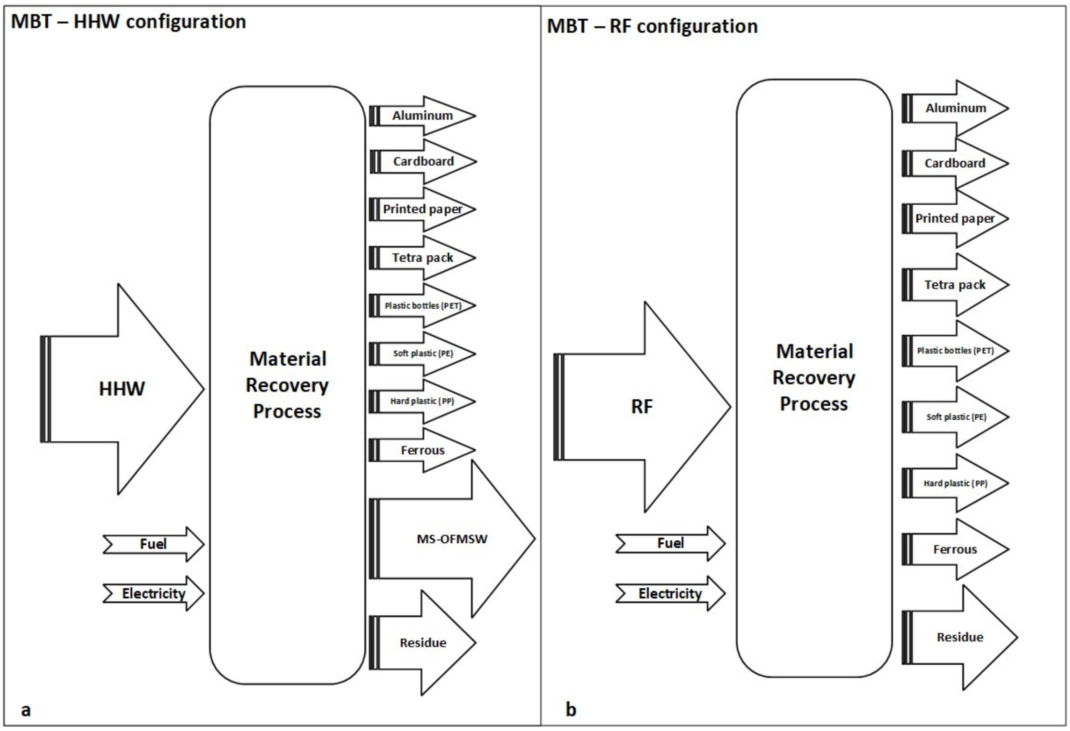

***Figure 3** Material recovery unit process, flowchart for a) HHW confeguration, b) RF conguration*

1.6.1.2. Mechanical Recovery

Initially, waste pre-sorting takes place to remove oversized items, which could damage the downstream equipment. The materials are then transferred with cranes and conveyor belts to bag openers, releasing the materials enclosed into bags. Two trommels (pore size 250 and 70 mm) separate the material to oversize, intermediate, and undersize. A combination of near-infrared (NIR) spectra optical separators, ballistic separators, magnets, eddy current separators, and handpicking are used to recover: OFMSW (undersize material that passes through an electromagnet), ferrous and aluminium metals, plastics (PET, PE, PP), tetrapak, cardboard, and printed paper (**Figure 3**), while all residues are directed to the landfill. The calculated recovery rates and fuel and energy consumption for each configuration are presented in**Table 1**.

Table 2 *Fractional composition and mass use in the scenarios for the initial household waste, the recovered MS-OFMSW in the baseline scenario, and the source segregated biowaste in SA and SB scenarios.*

Fraction name	Initial Waste composition (%) [a]	MS-OFMSW composition (%) [b]	Biowaste composition (%) [c]
Biodegradable			
Biowaste	51.57 ±5.93	81.55	89.31
Green waste	0.43 ±0.01	0.87	10.69
Printed paper [e]	8.81 ±0.39	9.91	-
Dirty paper	0.22 ±0.01	0.38	-
Cardboard [e]	3.51 ±1.31	0.77	-
Plastics			
Plastic bottles (PET) [e]	2.39 ±0.03	0.00	-
Soft plastic (PE) [e]	6.27 ±0.12	0.27	-
Hard plastic (PP) [e]	4.01 ±0.07	0.25	-
Tetrapak [e]	0.79 ±0.01	0.00	-
Non-recyclable plastic	1.02 ±0.01	0.66	-
Metals			
Aluminium foil and containers [e]	1.09 ±0.01	0.12	-
Batteries and electronic devices	0.74 ±0.09	0.16	-
Ferrous (tinplate/steel) [e]	1.86 ±0.02	0.16	-
Other materials			
Rubber [d]	1.71 ±0.02	0.08	-
Leather [d]	1.69 ±0.02	0.08	-
Textiles [d]	1.68 ±0.02	0.08	-
Wood processed [d]	1.85 ±0.02	0.00	-
Glass	2.54 ±0.06	2.98	-
Other non-combustibles	7.82 ±0.52	1.67	-
Total material in Kg	1000 [f]	492 [g]	402 [h]

(a) Data collected from site measurements.
(b) Composition calculated
(c) Composition adjusted to equal the MS-OFMSW biowaste recovery in the baseline scenario
(d) Leather, Wood, Textiles and Rubber (LWTR) were sampled as one fragment and divided into equal parts to calculate the initial waste composition and the attribution of the physicochemical characteristics.
(e) Fragments that are considered recyclable for this study and can substitute new materials
(f) Initial kg of household waste generated and used in every scenario
(g) kg of mechanically separated OFMSW recovered in scenario S0. It consists of 402 kilograms of biowaste and green waste
(h) kg of biowaste and green waste used in scenarios SA and SB, the quantities are the same recovered in scenario

1.6.1.3. Composting unit

The composting unit consists of three parts: a continuous flow aerobic composting tank, the refinery unit to remove un-composted materials, and the maturation phase (**Figure 4** in supplementary information) (a detailed inventory of this unit is presented in Chazirakis et al. (2022)). The refinery process residue is diverted to the landfill as cover material in the baseline scenario. In the alternative scenarios, residue consists of uncompostable biowaste and bulky green waste (assuming it is not contaminated with foreign materials). It is subjected to another windrow maturation process. The resulting compost is used in agriculture, substituting fertilizer use.

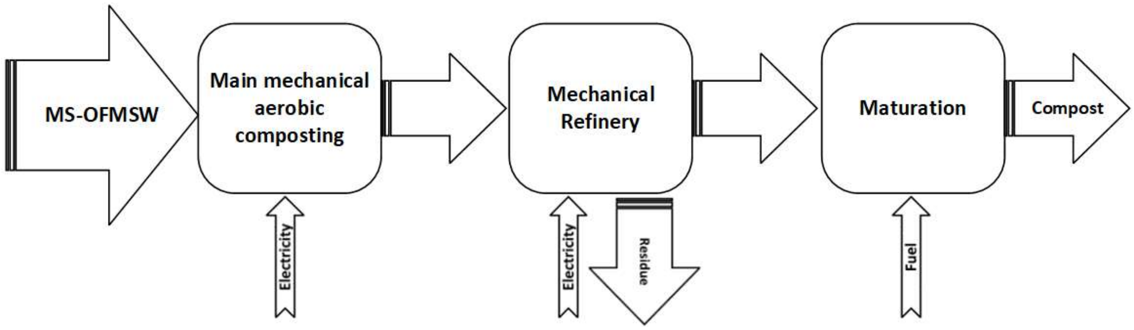

Figure 4 Composting unit process flowchart (Chazirakis et al., 2022).

1.6.2. Background processes

1.6.2.1. Landfilling

The residues from the above processes and operations are sent to the landfill. The sanitary landfill is modelled based on Manfredi and Christensen (2009). No biogas utilization is considered for all the scenarios, while a flare is used to burn the produced biogas. The impacts are calculated with a time horizon of 100 years. Carbon sequestration is the amount of biogenic carbon that, after 100 years, remains in a landfill as stored carbon (De la Cruz et al., 2013) or, in the case of land spreading of solid organic residues from biological treatment is bound to soil (Turner et al., 2016).

1.6.2.2. Recycling of recovered materials

Recovered materials are transported to a recycling facility to produce secondary materials for reprocessing. Such materials are regarded as substituting and avoiding virgin material use (plastics, papers, and metals). Cellulosic fibres can be reprocessed in a paper mill to produce low-grade paper (not blenched), and the ferrous materials are reprocessed in a steel manufacturing plant. At the same time, polymers are addressed to facilities where PET, PE, and PP are granulated and remelted. The recovered materials are assumed to substitute natural products with the following efficiency: paper manufacturing 0.83, polymers reprocessing 0.81, metal and aluminium 0.90. The net emissions for the material reprocessing are reported in Cremiato et al. (2018).

1.6.2.3. Use on land

The compost-like output (CLO) can be added to land for soil rehabilitation to restore quarries, dumping sites, or road slopes (Carabassa et al., 2020; Dawn Stretton-Maycock, 2009). The compost from source segregated biowaste can be used in agriculture as a soil conditioner and fertilizer substitute. Eventually, it acts as production avoidance (Boldrin et al., 2009, 2010; Zeller et al., 2020). Concerning compost used on land, data from agronomic modelling are used to estimate the carbon and nitrogen fate (Bruun et al., 2006). When the produced compost is applied to land, it is considered to substitute N in fertilizers at a ratio of 1:5 and P and K fertilizers at a ratio of 1:1 (Papadaskalopoulou et al., 2019). The produced compost in the SA and SB scenarios can be used in agriculture (Cremiato et al., 2018); therefore, the system boundaries for these scenarios are expanded to include fertilizer substitution (Edwards et al., 2018a; Seruga and Krzywonos, 2021).

1.7. Life cycle impact assessment (LCIA)

The LCIA is carried out according to the LCA ReCiPe2016 method. This method generates a complete picture of the ecological impacts. This methodology is preferred because it quantifies the environmental impacts in two groups: midpoints and endpoints. The first group comprises 17 midpoint impacts and relevant indicators, with ecological burdens such as global warming, acidification, and ozone depletion. The second group translates the environmental impacts into issues of concern (typically reflect damage in one of three protection areas: human health, ecosystem quality, and resources) (Ripa et al., 2017; Oliveira et al., 2017). The endpoint impacts are referred to in DALYs (disability-adjusted life years) as relevant for human health and represent the years lost or when a person is disabled due to a disease or accident. The unit for ecosystem quality is the local species loss integrated over time (species year). The unit for resource scarcity is dollar ($), which represents the extra cost involved for future mineral and fossil resource extraction. Hauschild and Huijbregts (2017) stated that the endpoint characterization is easier to interpret in terms of the relevance of the environmental flow.

1.8. Sensitivity analysis

A sequence of sensitivity methodologies is used to filter and evaluate the robustness of the developed model. Contribution analysis, perturbation analysis, uncertainty analysis, comparative analysis, and discernibility analysis methodologies are used to present, analyze, evaluate and interpret the produced results (Ripa et al., 2017; Valentina Bisinella et al., 2016). The LCA results are decomposed into their process contributions and sub-systems for contribution analysis, providing a quick overview of the significant contributors (Clavreul et al., 2012). For perturbation analysis, the sensitive parameters are identified by shifting each input parameter one at a time by a small percentage of 10% and evaluating whether it induces a significant change in a selected result based on the methodology presented in Bisinella et al. (2016). Since uncertainty analysis is devoted to systematically studying input propagation into output uncertainties, the Monde Carlo simulation methodology is used to produce random sampling and analytical formulas (Groen et al., 2017, 2014). Every input parameter is considered a stochastic variable with a specified probability distribution. The LCA model is constructed with one particular realization of every stochastic parameter, and the LCA results are calculated with this specific realization. The above steps are repeated several times (1,000 in this study), and the sample of LCA results is investigated as to its statistical properties, mean, standard deviation, and confidence interval. Finally, the sensitivity concludes with discernibility analysis, whereby one scenario preference is quantified over another. The result is based on pairwise comparisons of results for individual Monte Carlo samples of S0 to SA and SB scenarios, presented as percentages representing the probability of one system performing more favourable results for the environment than the baseline scenario (Bisinella et al., 2016).

2. Results and discussion

2.1. Material flow analysis

Table 2 shows the initial household waste composition of the Chania Prefecture. MBT process in the S0 scenario recovers 492 kg ww of MS-OFMSW (**Figure 5**), where biowaste and green waste represent 82.42% of its wet weight. The remaining OFMSW materials consist of 11.06% paper-like materials that can be partially composted (Alvarez et al., 2009; Tandy et al., 2009) and other materials considered uncompostable and only contribute to the increase of the treated mass and the contamination of the end product (Baiano et al., 2021; Edo et al., 2022). 402 kg of biowaste and green waste are collected in the brown container and directed for composting in the SA and SB scenarios. The above amount derives from the biowaste and green waste quantities recovered in the baseline scenario, making the alternative scenarios comparable to the S0. The composition of this stream is also presented in **Table 2**. The waste composition

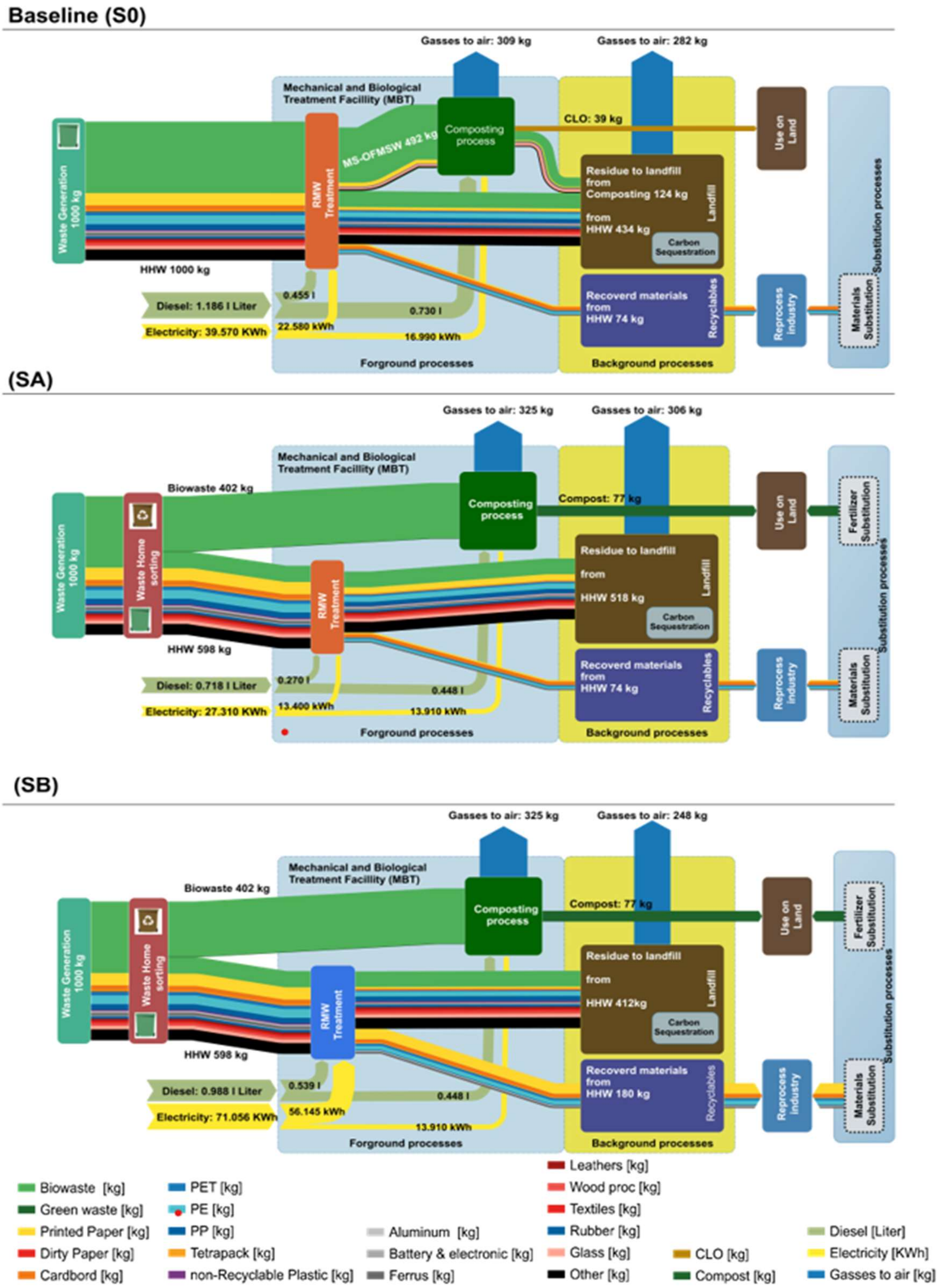

Figure 5 *Graphical representation of studied scenarios, mass balances, and resource flows (Sankey diagrams). The thickness of each line is proportional to mass.*

is essential for any waste management scenario, as well as the LCA study, and it significantly influences the final outcome. Slagstad and Brattebø (2013) stated that approximately ±15% change in the fraction of waste composition could result in a greater than 10% variation in GWP, human toxicity and nutrient enrichment (water impact categories). Hence, the LCA impacts are highly sensitive to uncertainties in waste composition.

Table 3 *MBT Facility Inventory for processing HHW and RF.*

	HHW configuration	RF configuration
Electricity consumed kWh/Mg ww		
Electricity	0.02258	0.0946
Vehicles type and diesel fuel consume [a] L/Mg ww		
Skid steer loader	0.10158	0.20234
Telehandler	0.08404	0.16762
Other drivable machines	0.26960	0.53970
Recovery rates of recyclable materials [b] (%)		
Cardboard	11.71 ±1.2	78.29 ±8.1
Printed paper	78.29 ±51.2	85.63 ±56.0
Tetrapack	11.71 ±1.2	62.87 ±6.5
Plastic bottles (PET)	39.31 ±0.2	68.25 ±0.4
Soft plastic (PE)	24.12 ±5.7	45.31 ±10.6
Hard plastic (PP)	4.62 ±1.2	13.3 ±3.3
Aluminum foil and containers	11.5 ±2.1	69.27 ±12.8
Ferrous (tinplate/steel)	47.19 ±2.1	76.99 ±3.4
Transfer coefficient's to MS-OFMSW [b] (%)		
Biowaste	77.74 ±8.2	-
Printed paper	55.29 ±6.0	-
Cardboard	10.76 ±1.8	-
Tetrapack	0	-
Dirty paper	84.22 ±0.5	-
Plastic bottles (PET)	0	-
Soft plastic (PE)	2.11 ±0.1	-
Hard plastic (PP)	3.09 ±0.1	-
Non-recyclable plastic	31.99 ±1.2	-
Aluminum foil and containers	11.75 ±0.1	-
Batteries and electronic devices	10.41 ±0.1	-
Ferrous (tinplate/steel)	4.34 ±0.1	-
Rubber, Leather, Textiles, Wood	2.27 ±0.1	-
Glass	57.7 ±1.2	-
Other non-combustibles	10.51 ±0.3	-

(a) All vehicles use a Euro 4 engine
(b) Recovery rates refer to % of the total fraction entering the facility

Table 3 presents the resulting inventory extracted from the study of the waste treatment process for the facility's two configurations (HHW and RF). The inventory data are used to build the three studied scenarios. Energy in the form of electricity is calculated in kWh per Mg of wet material treated in the facility. In the RF configureration, the material has higher recoverability than the HHW configuration due to lower water content and lower contamination from biowaste, making the pneumatic separation systems perform at higher rates consuming higher loads of energy. The same increase is observed for fuel consumption concerning the handling and transferring of the balkier materials during the RF treatment. The recovery rates are presented as a percent (%) of material collected from the total material in the treated waste. For example, in the HHW configuration, 11.71 % of the total cardboard in the waste stream is recovered for recycling, and 10.76% is in the MS-OFMSW sent for composting, the remaining 77.53% of cardboard is not recovered and ends up in the residue. In the HHW configuration, the plant exhibits lower recovery rates for the recyclable materials than in the RF configuration. Bourtsalas and Themelis (2022) reported similar data comparing six (6) European MBT plants in their study. Recoveries over 90% for most of the recyclables have been presented by Fitzgerald et al. (2012) and Pressley et al. (2015) for various material recovery facilities. The printed paper presents significant variations in its recoverability in both configurations due to its porous nature that absorbs liquids and subsequently alters its recycling quality and recovery potential. Other EU MBT plants have presented similar mass flow rates (Połomka and Jędrczak, 2019).

Material flow Sankey diagrams of the three scenarios are shown in **Figure 6**. Each colored line represents a different material, while the thickness of the line is proportional to its quantity. The size of the lines' for energy and fuel consumption are also proportional to their value. The material entering and treated in the facility in the baseline scenario is 1000 kg, from which 492 kg are mechanically separated as MS-OFMSW and proceed to the composting unit. The insertion of the brown container for home shorting of biowaste presents a 40.2% reduction in the amount of material entering the mechanical separation facility and an 18.3% reduction of the material that enters the composting unit in the alternative scenarios.

The brown container manages to collect only biodegradable materials (biowaste and green waste), plastics and paper; otherwise, co-segregated during mechanical sorting in the S0 scenario are avoided. Paper bags and compostable bags usually used in the collection of biowaste are not accounted for in the research. The composting process model created by Chazirakis et al. (2022) predicts the production of 77 kg of compost for agricultural use versus 39 kg of compost-like output in the S0 scenario. The absence of contaminants in the source materials allows for quantitative exploitation of the produced compost without any significant amount of rejects for landfilling.

Concerning the mechanical sorting in the SA scenario, the dynamics of the MFA modelling present a reduction of the incoming material in the MRF unit due to the diversion of biowaste, with a linear correlation to the electricity and fuel consumption. They are both reduced by 40.65% compared to the S0 scenario. Although the recoverable material quantities that enter the facility are the same in both S0 and SA scenarios, and since the same configuration is used in the collected recyclable materials, quantities are also the same in both the above scenarios. The SB scenario considers that by avoiding mixing biowaste with the rest of the home waste, the recoverable materials are not downgraded due to contamination, and the material entering the MRF unit simulates the RF material of the case study. The MFA for this scenario shows higher recovery rates for the recyclable fractions increasing the total amount of recyclable materials collected by 143.2%. At the same time, the electricity and fuel total consumption increased by 148.6% and 18.5%. Pressley et al. (2015) demonstrated that material recovery facilities (MRF) electricity consumption and performance can vary significantly depending on facility design and incoming waste composition, while the high contaminant rate in the mixed-waste MRF reduces the recovery efficiency and increases the residual rate.

The residue directed for landfilling is increased in SA compared to the S0 since it includes the portion that otherwise would be co-segregated mechanically in MS-OFMSW. In the high recycling rate scenario, SB residue discarded in the landfill is lower. Andreasi Bassi et al. (2017) stated that the high organic content of Greek household waste, compared to similar waste management scenarios in the EU, causes a significant environmental load in terms of global change due to methane emissions.

Concerning material flow, separate biowaste collection reduces the amount of processed material in the facility. It increases the produced compost production and quality while increasing the recycled material recoverability.

2.2. Life cycle impact assessment (LCIA)

The environmental impacts for the three scenarios are performed for endpoint impacts at three areas of protection, human health (HH) with damage to human health, resource scarcity (RS) with damage to resource availability, and natural environment with damage to terrestrial (EQT) and freshwater (EQF) ecosystem quality. The net results for each midpoint and endpoint impact are presented graphically in **Figure 6** and **Table 4**. The results above zero present burdens added to the environment and are considered damaging to the environment with adverse effects. The results below zero are considered avoided emissions and are taken to benefit the background with a positive impact (Blengini et al., 2012).

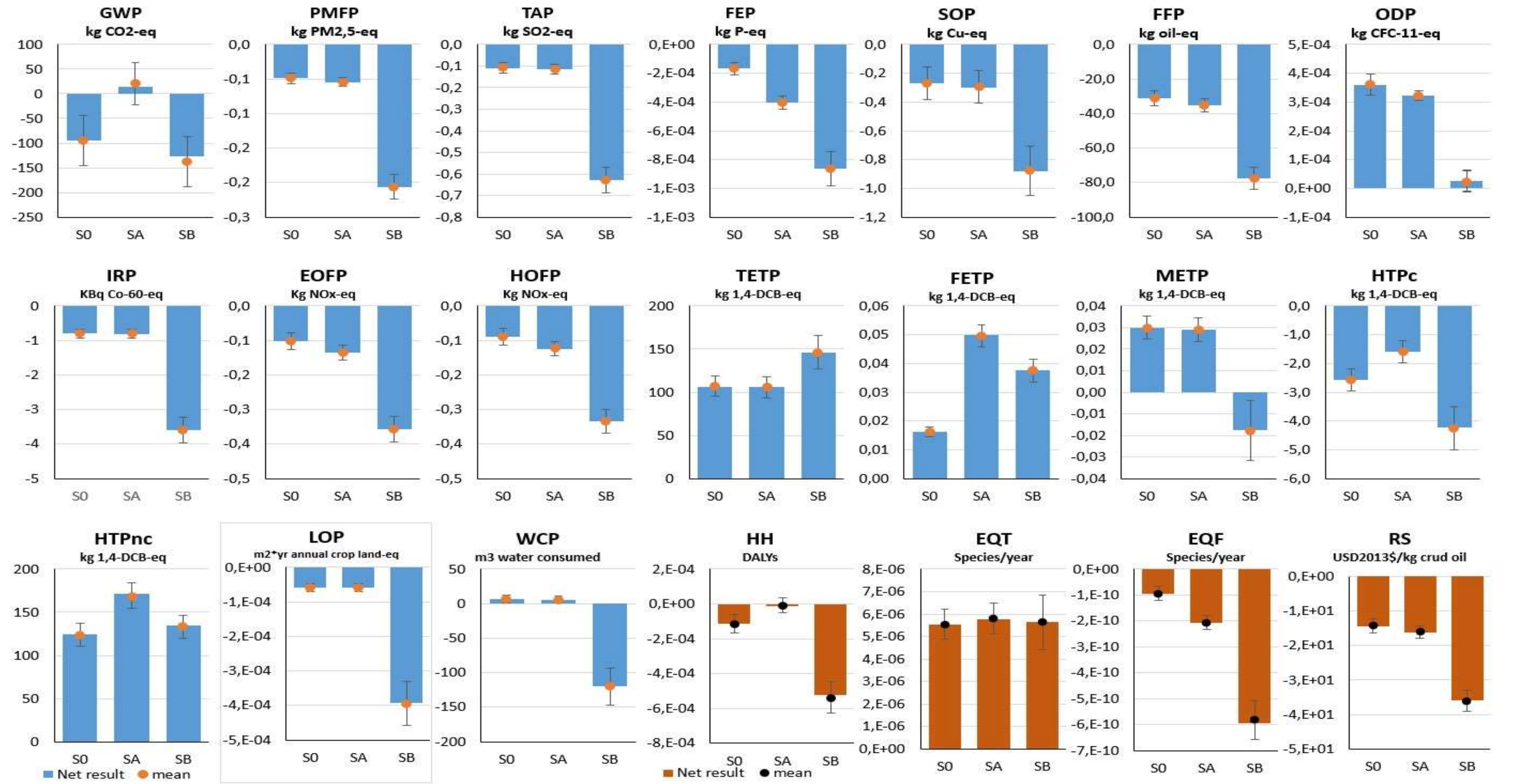

Figure 6 *Net characterized results obtained from primary data set for midpoint and endpoint impacts for the S0, SA, and SB scenarios. Mean, and standard deviation, obtained for every impact, based on primary data uncertainties put to the test using Monte Carlo simulation for 1000 runs.*

Human health is calculated in DALYs, and the assessment of all three scenarios produced scores below zero, meaning that the benefits to HH are more than the burdens. These results can be partially explained since all three scenarios include biowaste segregation and recyclable material recovery for raw material substitution, meaning that a significant proportion of negative impacts from new raw material production are avoided (Bourtsalas and Themelis, 2022; Blengini et al., 2012). However, comparing (SA) to the baseline scenario shows that the SA net scores are lower by order of 10 since the brown container for clean biowaste collection does not prevent a portion of paper material otherwise mechanically co-segregated to co-compost, instead it is led to the landfill. In contrast, biowaste segregation with an enhanced material recovery in the SB scenario shows 4.6 times more beneficial to HH than the S0. GWP as a midpoint impact plays a significant role in HH calculation, and its contribution is reflected in the above changes. Many researchers indicate special attention to climate change, also referred to as the global warming potential (GWP) midpoint impact, since it is the only impact category that can be easily compared with other studies (Christensen et al., 2020; Papadaskalopoulou et al., 2019; Zeller et al., 2020). The climate change impact is determined based on the mass emission (kg) of three gases: N_2O, CH_4, and CO_2. These emissions are transformed into (kg) CO_2-eq using the ReCiPe2016 equivalent factors at the midpoint level (hierarchism perspective). Christensen et al. (2020) calculated that the contribution to the global warming impact from the treatment of 1 Mg of wet waste for an MBT plant ranges from about 150 kg CO_2-eq, which strongly depends on the choice of technology used, referring to an attributional system. In S0, GWP presents a saving of -94.8 kg CO_2-eq. At the same time, in the alternative scenarios, the impact ranges from 14.2 kg to -143.0 kg CO_2-eq per wet waste, depending on the separation technology efficiency.

The RS impact shows benefits for all three scenarios presented as savings, including oil, with SA being 12.4% better than the baseline. At the same time, the ranking for SB was 148.3% better than the S0. The increase in the recycling rates due to biowastes segregation has a significant impact on this category.

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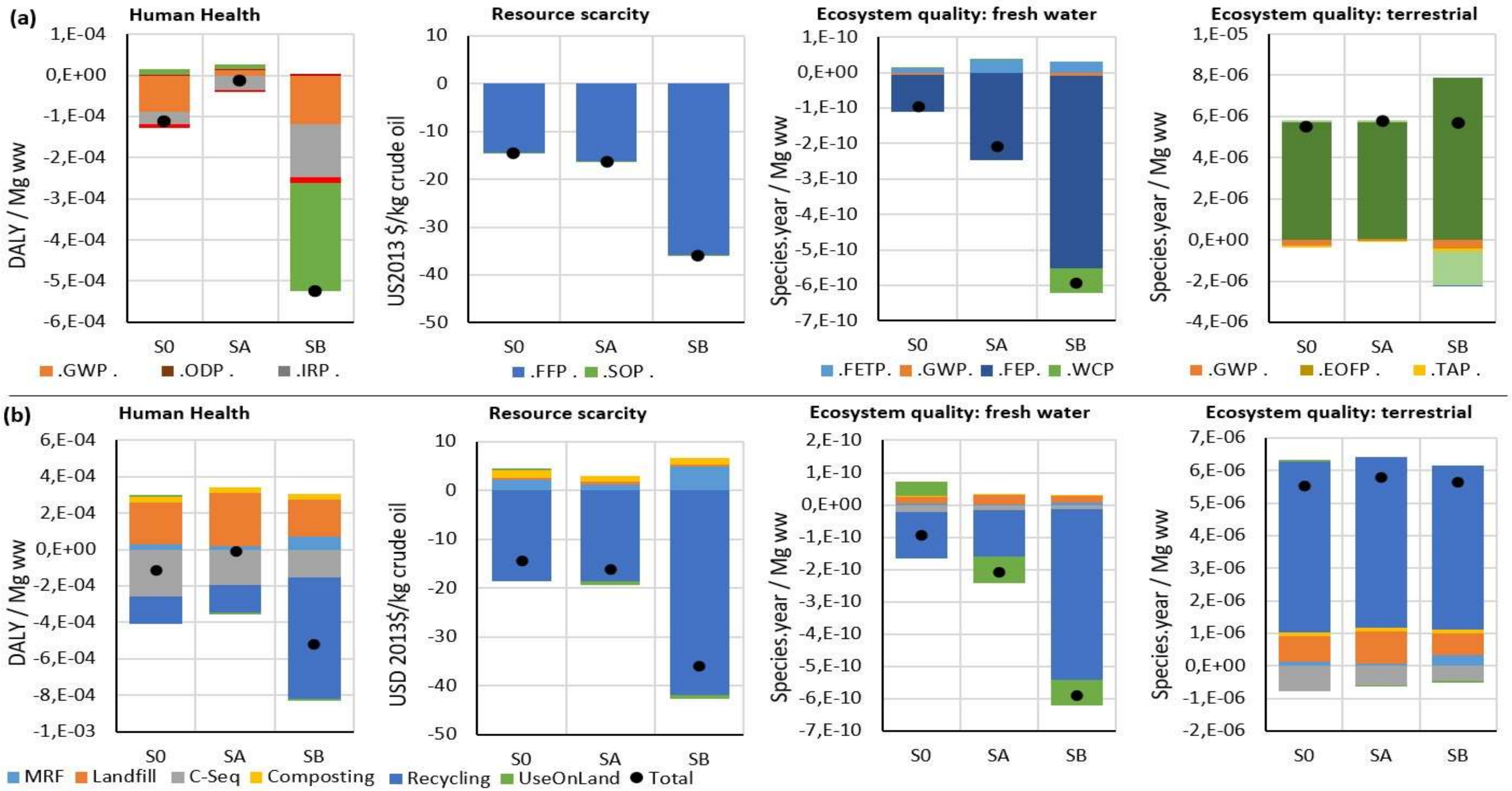

Figure 7 (a) *Contribution of Processes Endpoint impacts on Human Health (HH), Resource Scarcity (SC), and Ecosystem Quality for terrestrial (EQt) and freshwater (EQf) systems. The different colors indicate the proportion of the contribution of each midpoint impact or process. The black dot indicates the net result* ***(b)*** *Contribution of midpoint impacts to endpoint impacts on Human Health (HH), Resource Scarcity (SC), and Ecosystem Quality for terrestrial (EQt) and freshwater (EQf) systems. The different colours indicate the proportion of the contribution of each midpoint impact. The black dots indicate the net results.*

In the EQ category concerning freshwater, the impacts are very low. However, they are considered to benefit the environment, while SB has the best score, followed by the SA scenario. Concerning terrestrial EQ, the two scenarios present minimal deviations from the baseline (**Figure 6**). Further analysis of the endpoint impacts presented in Table S1 and **Figure 7a** shows each impact as the cumulative contribution of several midpoint impacts presented by contribution analysis. The results indicate that global warming potential (GWP), particle matter formation potential (PMFP), and water consumption potential (WCP) are significant impacts on HH

Biowaste separate collection in the SA and SB avoids the co-composting of foreign biodegradable materials (mostly cellulose and lignin in paper-like materials), otherwise mechanically co-segregated and composted during mechanical sorting. This portion represents 11.05% of the MS-OFMSW, while their fate in the landfill contributes to the increase of CH_4 release and the adverse impacts in the GWP category (García et al., 2016). In contrast, the extensive recovery of materials in the SB scenario is based on the hypothesis that the gradual reduction of biowaste in the treated HHW leads to higher recyclables recovery rates in the MBT, resulting in raw material substitution and avoiding WCP and PMFP impacts that significant influence HH. The above results are in agreement with similar LCA studies of mechanical sorted waste management systems (Bourtsalas and Themelis, 2022; Blengini et al., 2012), presenting enhanced levels of separate collection and subsequent recycling (Cimpan et al., 2015).

To better understand the resulting scores of each category, the system is divided into six main processes. Every subprocess is attributed and grouped, presented as materials recovery unit, biogas production during landfilling, carbon sequestration in the landfill, composting process, recycling of the recovered materials, and use of compost in land with fertilizer substitution. The breakdown of the endpoint results in **Figure 7b** highlights each system main process contribution. Landfilling offers both advantages and disadvantages. A portion of the carbon is assumed to be deposited and stored for a long time. This process is called carbon sequestration and is considered a benefit to the environment (De la Cruz et al., 2013; Finnveden et al., 2005; Zhao et al., 2011). On the other hand, carbon oxidation on the landfill cover released as GHGs into the air negatively results in several environmental impacts, primarily GWP. The two processes compete, producing a beneficial net score -3.09E-5 DALYs for S0 and burden with a net score of 9.56E-5 and 4.55E-5 DALYs for SA and SB, respectively.

The composting unit contributes negatively to all impacts because of the electricity and fuel consumed (Chazirakis et al., 2022). The amount in dry bases of biowaste composted in the unit for both scenarios SA and SB is the same as the biowaste mechanically recovered and processed in the S0 scenario, reducing the emissions related to the mechanical process of the compost. In contrast, landfill CH4 emissions are increased as materials enter the landfill. It is considered that a significant fraction (50-58%) of the carbon entering the landfill is composed almost entirely of biogenic carbon (Chazirakis et al., 2022).

Recycling of paper, plastics or metals are considered an avoided impact since the initial production of these materials from raw materials is directly linked to resource scarcity and water consumption. PMFP and WCP are also linked due to electricity and fuel consumption from the above production processes (De Feo et al., 2021). Similarly, De Feo et al. (2019) stated that a 1% increase in the source separation could avoid the emission of 5 kg CO_2-eq and 5 g PM_{10} for every citizen. Chen et al. (2019) showed that increasing the proportion of mechanical recycling would reduce all environmental impacts, including up to 51.8% on PMFP. Cimpan et al. (2015) indicated the significance of system boundary choices, technology choices, and type or a mix of energy used in reprocessing and primary production are vital when modelling paper and cardboard recycling, compared to recycling other materials. Merrild et al. (2008) showed that the GWP associated with reprocessing is highly plant-related, and the choice of the dataset is an essential parameter in the system definition. Montejo et al. (2013) stated that the recommendation for upgrading and/or commissioning future plants is optimizing material recovery through increased automation.

Concerning the Compost-like output (CLO) and compost brown container collection, their use as a soil conditioner and fertilizer substitute is dependent primarily on the origin of the initial composted materials, source or mechanically segregated based on the EU Directive. Even though this process contributes low to the overall net score for HH, it shows benefits for SA and SB. These results are primarily attributed to the reduction in the use of fertilizers, electricity and water owing to compost applications, and therefore reducing the release of GHG, nutrients, and toxic chemicals to the environment (air, water, and soil) during production and use of these avoided inputs. The above beneficial impacts offset the GHGs, nutrients, and toxic substances released to the environment during the production and use of electricity and diesel required to produce and apply composted products (Sardarmehni et al., 2021; Sharma and Campbell, 2007).

2.3. Sensitivity analysis

The sensitivity analysis highlights the main influencing parameters for each scenario which are presented in **Table 4** . The ranking is based on the methodology of calculating the Normalized sensitivity ratio (NSR) for each system parameter proposed by Andreasi Bassi et al. (2017). The highest ranked parameter in GWP, TAP, and FETP impacts in all scenarios is the recovery rate of biowaste and OFMSW, which expresses the carbon fate throughout composting or landfilling and affects the assessment results owing to the different biogenic carbon accounting methodologies in each process (Laurent et al., 2014). The parameter's importance has been stated in other studies (Cimpan et al., 2015; do Carmo Precci Lopes et al., 2019) for aerobic composting systems and anaerobic treatment (Moreira de Oliveira et al., 2022). The parameters concerning recyclable materials recovery rates, and more particularly aluminum and PET recovery and substitution rates show increased sensitivity, receiving higher rankings. The avoided energy and resources emissions released for producing these materials compensate the negative impacts of landfilling and add benefits to the overall waste management system environmental behavior (Andreasi Bassi et al., 2017).

The uncertainty propagation results are extracted from the Monte-Carlo simulation performed for 1,000 runs in all three scenarios and every midpoint and endpoint impact. The mean, standard deviation, and variance are quoted with the net characterized results in **Figure 6** and ***Table 5***. In most of the impact categories, an insignificant deviation of the mean values from the net results is observed. The influence of material recovery rates is represented by the variance of the simulated results. The comparison between the studied scenarios is sufficient, and confirms the robustness of the used parameters. For instance, the impact categories dependent on biowaste quantity and quality parameters contribute in deviation more than 51% for the mean values in all the scenarios although the comparison presents the benefits of the SB scenario. To quantify the above results, discernibility analysis is performed as the last step of the sensitivity analysis.

Table 4 *Selective presentation of parameters with the highest influence on the studied scenarios for each endpoint impact. The most sensitive parameters for each of the presented midpoint impacts*

	Scenario		
Impact	**S0**	**SA**	**SB**
GWP	Biowaste recovery ratio in mechanical sorting (MBT)	Biowaste home sorting efficiency	Biowaste home sorting efficiency
PMFP	Aluminum recovery ratio in mechanical sorting (MBT) PET recovery ration in mechanical sorting (MBT) Substitution ratio of Aluminum materials (Recycling). Substitution ratio of PET materials (Recycling).	Aluminum recovery ratio in mechanical sorting (MBT) PET recovery ration in mechanical sorting (MBT) Substitution ratio of Aluminum materials (Recycling). Substitution ratio of PET materials (Recycling).	Aluminum recovery ratio in mechanical sorting (MBT)
TAP	Biowaste recovery ratio in mechanical sorting (MBT))	Biowaste home sorting efficiency	Aluminum recovery ratio in mechanical sorting (MBT)
FEP	Hard plastic recovery ration in mechanical sorting (MBT)	Hard plastic recovery ration in mechanical sorting (MBT)	Hard plastic recovery ration in mechanical sorting (MBT)
FFP	Soft plastic (PE) recovery ration in mechanical sorting (MBT) Substitution ratio of PE materials (Recycling).	Soft plastic (PE) recovery ration in mechanical sorting (MBT) Substitution ratio of PE materials (Recycling).	Soft plastic (PE) recovery ration in mechanical sorting (MBT)
TETP	Ferrous recovery ratio in mechanical sorting	Ferrous recovery ratio in mechanical sorting	Ferrous recovery ratio in mechanical sorting
FETP	Biowaste recovery ratio in mechanical sorting (MBT) Recovery ratio of biowaste in refinery process (Composting)	Biowaste home sorting efficiency	Biowaste home sorting efficiency
HTPc	Soft plastic (PE) recovery ration in mechanical sorting (MBT) Substitution ratio of PE (Recycling)	Soft plastic (PE) recovery ration in mechanical sorting (MBT) Substitution ratio of PE (Recycling)	Substitution ratio of PE (Recycling)
WCP	Aluminum recovery ratio in mechanical sorting (MBT) Substitution ratio of Al (Recycling) Substitution ratio of paper (Recycling)	Aluminum recovery ratio in mechanical sorting (MBT) Substitution ratio of Al (Recycling) Substitution ratio of paper (Recycling)	Aluminum recovery ratio from (MBT)

The presented parameters scored a NSR higher than 0.8, based to the methodology proposed by (Andreasi Bassi et al., 2017).

Table 5 *Net characterized results obtained from primary data set for midpoint and endpoint impacts for the S0, SA, and SB scenarios. Mean, standard deviation, and variances obtained for every impact, based on primary data uncertainties put to the test using Monte Carlo simulation for 1000 runs.*

Impact	Scenario Unit	S0 *Net result*	mean	st.deviation	variance	SA *Net result*	mean	st.deviation	variance	SB *Net result*	mean	st.deviation	variance
Midpoint													
GWP	kg CO2-eq	*-9.48E+01*	-9.49E+01	5.14E+01	2.64E+03	*1.43E+01*	2.05E+01	4.29E+01	1.84E+03	*-1.26E+02*	*-1.37E+02*	*5.06E+01*	*2.56E+03*
PMFP	kg PM2,5-eq	*-4.89E-02*	-4.89E-02	7.57E-03	5.72E-05	*-5.48E-02*	-5.46E-02	6.49E-03	4.22E-05	*-2.06E-01*	*-2.06E-01*	*1.75E-02*	*3.08E-04*
TAP	kg SO2-eq	*-1.09E-01*	-1.09E-01	2.52E-02	6.37E-04	*-1.16E-01*	-1.16E-01	2.23E-02	4.96E-04	*-6.27E-01*	*-6.29E-01*	*5.78E-02*	*3.34E-03*
FEP	kg P-eq	*-1.68E-04*	-1.67E-04	4.24E-05	1.79E-09	*-4.07E-04*	-4.03E-04	4.49E-05	2.02E-09	*-8.64E-04*	*-8.62E-04*	*1.20E-04*	*1.43E-08*
SOP	kg Cu-eq	*-2.67E-01*	-2.70E-01	1.13E-01	1.27E-02	*-2.98E-01*	-2.95E-01	1.15E-01	1.32E-02	*-8.80E-01*	*-8.78E-01*	*1.73E-01*	*2.99E-02*
FFP	kg oil-eq	*-3.13E+01*	-3.13E+01	4.33E+00	1.88E+01	*-3.52E+01*	-3.49E+01	3.91E+00	1.53E+01	*-7.77E+01*	*-7.78E+01*	*6.46E+00*	*4.17E+01*
ODP	kg CFC-11-eq	*3.60E-04*	3.60E-04	3.71E-05	1.38E-09	*3.22E-04*	3.22E-04	1.73E-05	2.99E-10	*2.69E-05*	*2.47E-05*	*3.72E-05*	*1.38E-09*
IRP	kBq Co-60-eq	*-8.00E-01*	-8.01E-01	1.30E-01	1.69E-02	*-8.07E-01*	-8.06E-01	1.33E-01	1.78E-02	*-3.59E+00*	*-3.60E+00*	*3.66E-01*	*1.34E-01*
EOFP	kg NOx-eq	*-1.02E-01*	-1.02E-01	2.45E-02	6.02E-04	*-1.36E-01*	-1.35E-01	2.12E-02	4.51E-04	*-3.58E-01*	*-3.57E-01*	*3.59E-02*	*1.29E-03*
HOFP	kg NOx-eq	*-9.00E-02*	-9.01E-02	2.40E-02	5.75E-04	*-1.24E-01*	-1.23E-01	2.07E-02	4.27E-04	*-3.34E-01*	*-3.34E-01*	*3.48E-02*	*1.21E-03*
TETP	kg 1,4-DCB-eq	*1.06E+02*	1.07E+02	1.18E+01	1.38E+02	*1.06E+02*	1.06E+02	1.22E+01	1.49E+02	*1.46E+02*	*1.46E+02*	*1.94E+01*	*3.75E+02*
FETP	kg 1,4-DCB-eq	*1.62E-02*	1.62E-02	1.57E-03	2.45E-06	*4.99E-02*	4.96E-02	3.77E-03	1.42E-05	*3.77E-02*	*3.76E-02*	*3.97E-03*	*1.58E-05*
METP	kg 1,4-DCB-eq	*2.97E-02*	2.99E-02	5.37E-03	2.89E-05	*2.88E-02*	2.90E-02	5.53E-03	3.05E-05	*-1.75E-02*	*-1.78E-02*	*1.39E-02*	*1.93E-04*
HTPc	kg 1,4-DCB	*-2.58E+00*	-2.57E+00	3.80E-01	1.44E-01	*-1.60E+00*	-1.59E+00	3.92E-01	1.54E-01	*-4.23E+00*	*-4.26E+00*	*7.47E-01*	*5.58E-01*
HTPnc	kg 1,4-DCB	*1.24E+02*	1.24E+02	1.31E+01	1.71E+02	*1.71E+02*	1.69E+02	1.49E+01	2.21E+02	*1.35E+02*	*1.33E+02*	*1.31E+01*	*1.72E+02*
LOP	m^2*yr annual crop land-eq	*-5.86E-05*	-5.88E-05	1.04E-05	1.08E-10	*-5.86E-05*	-5.86E-05	1.07E-05	1.15E-10	*-3.92E-04*	*-3.94E-04*	*6.35E-05*	*4.04E-09*
WCP	m^3 water consumed	*6.65E+00*	6.46E+00	5.41E+00	2.93E+01	*5.85E+00*	5.81E+00	5.55E+00	3.08E+01	*-1.20E+02*	*-1.20E+02*	*2.65E+01*	*7.03E+02*
Endpoint													
HH	DALYs	*-1.13E-04*	-1.12E-04	5.23E-05	2.73E-09	*-1.36E-05*	-7.02E-06	4.40E-05	1.94E-09	*-5.24E-04*	-5.36E-04	9.07E-05	8.22E-09
EQT	Species /year	*5.52E-06*	5.55E-06	6.73E-07	4.53E-13	*5.78E-06*	5.81E-06	6.83E-07	4.67E-13	*5.66E-06*	5.63E-06	1.23E-06	1.50E-12
EQF	Species /year	*-9.44E-11*	-9.41E-11	2.66E-11	7.08E-22	*-2.08E-10*	-2.06E-10	2.76E-11	7.63E-22	*-5.93E-10*	-5.82E-10	7.53E-11	5.67E-21
RS	USD 2013$ /kg crude oil	*-1.45E+01*	-1.44E+01	1.99E+00	3.98E+00	*-1.63E+01*	-1.61E+01	1.80E+00	3.24E+00	*-3.60E+01*	-3.60E+01	2.98E+00	8.86E+00

The graphical results of the discernibility analysis are presented in **Figure 8**. It shows the times in 1,000 runs of Monte-Carlo simulation, where SA or SB scenarios perform better (have a value above zero) than the baseline scenario. The quantification of these results shows that the SB scenario in all the simulations (100%) is more beneficiary than the S0 in HH, EQF, and RS in EQT, the simulation showed 51.9% of runs that the SB outperformed by S0 due to variations in TETP, TAP, and WCP impacts related to recycling recovery and energy consumed in MBT. Comparing S0 with the SA scenario, there are cases (6.0% of the Monte-Carlo simulation runs) where the SA outperforms the baseline scenario in HH. In EQT, the results showed 49.1% of the runs that, SA prevails, while for EQF and RS, the results are 99.9% and 75.0%, respectively. The reason is attributed to the high variability of the MBT recovery parameters, justifying the significance of recovering technologies (Christensen et al., 2020).

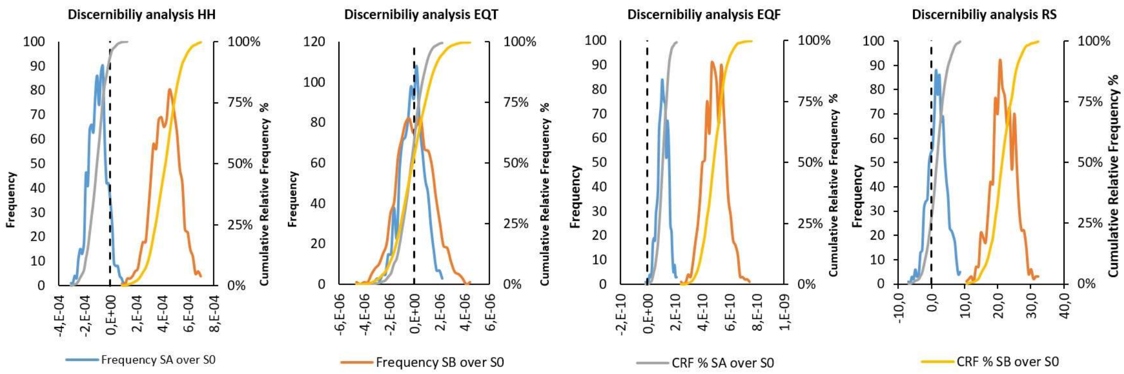

Figure 8 Discernibility analysis for the endpoint impacts for the two alternative scenarios (the comparison between results expressed as probability distributions).

4. Conclusions on Chapter 5

In this chapter we evaluated the implications in human health, ecosystem quality and resource scarcity of a transition from a mechanically segregated OFMSW system a gradually extended biowaste separate collection waste management system. LCA confirmed that biowaste source collection combined with downstream recycling is the most effective approach to lower environmental impacts. The human health impacts are improved by a factor of 3.5, the ecosystem quality by 5.8 and the resource scarcity by 1.4. In addition, recovering materials from mixed waste enhances the possibility of contaminants being adsorbed and reduces its substitution potential. Ideally, municipal collection systems should include source separation of organic wastes to avoid contamination with other waste streams and improve the quality of both recyclables and compostables. The prime argument for large-scale implementation of biowaste collection is whether the recoverability properties of the remaining HHW would be gradually improved, leading to higher recovery rates of recyclable materials. Such transitions involve complex effects beyond changes in waste flows, and after their management, this is due to interactions with existing treatment infrastructure recyclable recoverability. The degree of this improvement should be further investigated.

Landfilling of residual waste is the dominant source of GHG burdens for the existing system. The mechanical sorting of OFMSW manages to send for composting along with the biowastes, a countable portion of paper materials that in other cases would be landfilled, contributing to the GHG production. In contrast, the alternative scenarios cannot benefit from this composted fraction of fibrous material; on the other hand, paper material that is led for landfilling contributes to environmental burdens unless the efficiency of recycling is increased. Inert and materials hard to separate are also collected during mechanical shorting and co-composted, providing the conditions for contaminants and pollutants to migrate and be absorbed by the end product making it in some cases unsuitable for organic farming. The transition towards a waste management system based on separate comprehensive collection could be advantageous even without the cascading effects engaging waste imports.

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6. Chapter

1. General conclusions

In the study's initial phase, a critical examination of waste transport practices shows that collection trucks significantly contribute to environmental impacts. The strategic placement of Waste Transfer Stations (WTS) proves essential when considering factors such as distance to treatment facilities and road networks. Life Cycle Assessment (LCA) emerges as a valuable tool, providing precise data for evaluating WTS scenarios and aiding decision-making in waste management. Emphasising the efficient use of WTS and adopting practices to reduce fuel consumption aligns with sustainability objectives in waste collection. Additionally, the standard deviation of net results can serve as a reliable estimator of the efficiency of the collection and transport processes. Standard deviation (Std) emerges as a valuable indicator, reflecting the variability inherent in the waste collection and transport processes, as it can serve as an estimator of the efficiency of these processes. The numeric redaction in this indicator within the final net results can be interpreted as a beneficial outcome for the calculated results.

In the prospect of waste treatment, the study investigates the implications of transitioning compost prodaction and use from mechanical biowaste recovery to a biowaste separate collection waste management system. Life Cycle Assessment in consequential prospect reveals significant improvements in human health, ecosystem quality, and resource scarcity through biowaste source collection and downstream recycling. The study weighs the importance of municipal collection systems incorporating source separation to enhance the recoverability of recyclable materials and prevent contamination of the revered materials. While recognising the potential for improved recoverability of Household Waste in large-scale biowaste collection, the study acknowledges the complexity of interactions with existing treatment infrastructure, necessitating further investigation. The advantages of accurate, local, and updated data have proved significant elements for implementing this type of investigation. At the same time, uncertainty propagation in the context of sensitivity analysis allows for more precise and enlightening results. Ultimately, LCA tools proved valuable in evaluating the transition to a waste management system focused on separate collection.

2. Discussion

The advancements in analytical chemistry over the last 30 years have significantly improved our ability to trace the pathways of elements in nature, even in lower quantities, across numerous samples, and in less time. These advancements have enabled the collection of a vast amount of information, providing insights not only on a global scale but also on local waste schemes. At the same time, the progress in computational power has allowed for a more comprehensive analysis of various aspects concerning the fate of elements and compounds in nature, enhancing our understanding of their interactions and their beneficial or adverse effects.

The production and release of various components are closely associated with waste production. Despite accumulating substantial information, scientists struggle to comprehend the mechanisms, pathways, and environmental interactions related to waste. Consequently, there is a need for further research and exploration in this field. ISWM system poses a significant challenge, especially in rapidly expanding urban areas. Implementing sustainable ISWM is critical in attaining diverse, sustainable development goals, encompassing clean water and sanitation, sustainable cities, climate change mitigation, and sustainable consumption (Abubakar et al., 2022). However, urban areas encounter obstacles in waste management due to population growth, consumerism, and resource constraints. Insufficient awareness, technologies, financial limitations, and governance issues impede effective solid waste management practices. Developing countries, in particular, face challenges such as low waste collection rates and environmental risks.

Accurate data is essential for effective urban solid waste planning. In the case of the Chania region, there was a lack of a current inventory with up-to-date data that could capture the specific local conditions and characteristics. The generated information highlighted the importance of using local data rather than relying solely on global data, as it allows practitioners to better understand the needs of the waste management system. This enables the development of more appropriate proposals tailored to the region's specific requirements.

To the best of the author's knowledge, no comprehensive study has been conducted to assess the fate of fragmental household waste throughout its entire life cycle from a life cycle assessment (LCA) perspective in the region of Greece. However, using the EASETECH software has allowed for more detailed and precise modelling of the integrated solid waste management system in the region, specifically at a fragmental level.

This approach has facilitated tracking waste from its initial generation to its final disposal or treatment. It provides valuable inventories related to mix-waste and recyclable fragments, waste collection, transport, and treatment. These inventories serve as a valuable resource for further studying various alternative scenarios.

The in-depth analysis of the mechanical composting process has resulted in the development of a valuable model tool that is customisable and adapted to the specific characteristics of the Mediterranean region. This model tool can be utilised to simulate and experiment with various scenarios related to composting. Furthermore, the inventory produced as part of this analysis provides a robust foundation for conducting simulations and experiments, enabling a better understanding of the environmental and economic implications of different approaches to composting. The flexibility and adaptability of the model tool and the comprehensive inventory contribute to its usefulness in supporting decision-making processes and optimising waste management strategies in the Mediterranean context.

The study also investigated the recovery and utilisation of biowaste, either mechanically or through source segregation, via the composting process. Encouraging results were obtained, indicating that such measures could enhance the environmental profile of the proposed waste management system without requiring significant alterations to the existing infrastructure.

3. Future research

The current Thesis has certain limitations that restrict the scope of the study, leaving several aspects of waste management unexplored. This presents an opportunity for further research and investigation, which holds significant environmental, scientific, and other relevant interests. Future goals for research include gaining a deeper understanding of the environmental impacts associated with advanced separation and recovery techniques that leverage the physicochemical properties of the recovered materials. Additionally, it aims to explore advancements in reducing collection emissions through source segregation methods. Furthermore, plans are made to assess the impacts of waste segregation and increase awareness of the recoverability and properties of recovered and residue materials. These lines of inquiry will contribute to a more comprehensive understanding of waste management practices and their potential environmental benefits.