

ISTANBUL TECHNICAL UNIVERSITY ★ GRADUATE SCHOOL

**ENVIRONMENTAL AND ECONOMIC
ASSESSMENT OF ZERO WASTE MANAGEMENT**



Ph.D. THESIS

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Department of Environmental Engineering

Environmental Sciences Engineering and Management Programme

DECEMBER 2023

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DECEMBER 2023

İSTANBUL TEKNİK ÜNİVERSİTESİ ★ LİSANSÜSTÜ EĞİTİM ENSTİTÜSÜ

**SIFIR ATIK YÖNETİMİNİN ÇEVRESEL
VE EKONOMİK SONUÇLARININ DEĞERLENDİRİLMESİ**

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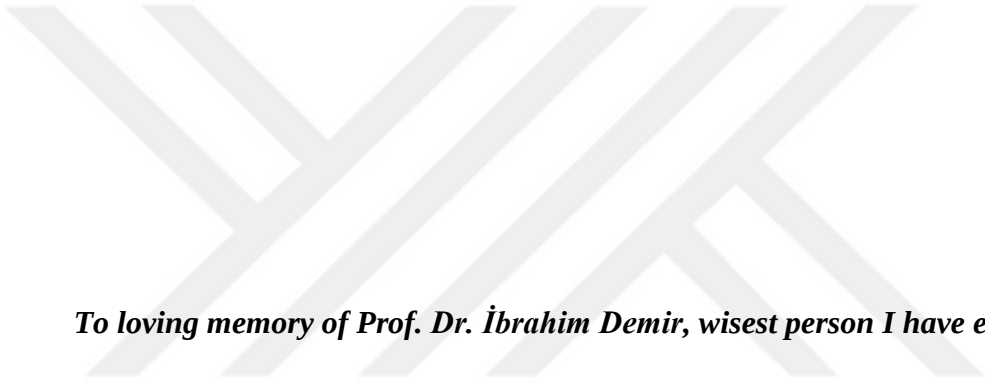
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To loving memory of Prof. Dr. İbrahim Demir, wisest person I have ever known.



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ABBREVIATIONS

AD	: Anaerobic Digestion
CE	: Circular Economy
CCS	: Carbon Capture and Storage
BAU	: Business as Usual
BM	: Business Model
DQA	: Data Quality Analysis
FU	: Functional Unit
FW	: Food Waste
GHG	: Greenhouse Gas
IMM	: Istanbul Metropolitan Municipality
IPPC	: Intergovernmental Panel on Climate Change
LCA	: Life Cycle Assessment
MCDM	: Multiple-Criteria Decision Making
MFA	: Material Flow Analysis
MRF	: Material Recovery Facility
MSW	: Municipal Solid Waste
NMVOC	: Non-methane Volatile Organic Compound
RDF	: Refuse Derived Fuel
SDG	: Sustainable Development Goals
UN	: United Nations
WM	: Waste Management
WtE	: Waste to Energy
ZW	: Zero Waste



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ENVIRONMENTAL AND ECONOMIC ASSESSMENT OF ZERO WASTE MANAGEMENT

SUMMARY

The waste management (WM) approach for the protection of resources known as "zero waste" (ZW) has become popular in recent years. A number of measures have been taken and facilities have been installed to facilitate improved management of municipal solid waste (MSW). ZW to landfill targets have gained significant traction worldwide, including in Türkiye. The aim of this thesis is to assess the environmental and economic results of WM pathways within the context of an increasingly applied ZW approach. This study evaluates the sustainability of municipal solid WM through the ZW goal on multiple scales (Istanbul and Ayazağa Campus).

The thesis provides a framework that enables institutions to develop a goal-oriented WM strategy using material flow analysis (MFA) and life cycle assessment (LCA). In addition to presenting the framework, a case study was conducted on a campus scale by using primary and secondary data. The framework assumes that no prior data is available, and the study will begin by collecting primary data on campus. For providing primary data; waste characterization and recycling potential of the Istanbul Technical University (Türkiye) Ayazağa Campus before (2019) and after (2022) the ZW management strategy the campus was divided into four distinctive groups, which are (i) academic (ii) administrative (iii) residential/dormitory and (iv) cafeteria. First, initial field study was conducted afterwards (for waste characterization and recycling potential), the new containers were placed. Students and campus personnel have been trained in within the scope of ZW management practices through both in-person and online seminars. The final phase of the study, the second field work, was completed. The results demonstrate that the waste generation rate in the pilot areas fluctuates between 0.045-0.190 kg/cap/day, but it decreases to 0.011-0.117 kg/cap/day in the second field study. The first field study had a potential recycling rate of 76.3%, but then it dropped to 68.2% in the second study. The MFA results indicate that landfill diversion ranges between 29.8% to ~99-100%, some residuals or ash from the incineration plant will still be disposed in landfill. Furthermore, simply diverting waste from landfill does not necessarily lead to circularity or directly address sustainable consumption and public attitudes towards ZW goals. The study framework aims to address potential challenges in campus-based, goal-oriented WM studies. Future case studies from other institutions and their campuses could help validate and improve this methodology. Based on the Ayazağa waste characterization results, the efforts to establish a ZW management system also led to a reduction in waste generation and increase in recycling performance. However, further studies are still required to assess the ZW's public awareness activities.

In addition, a case example was developed based on ITU Ayazağa campus, Türkiye, with annual separated food waste of 577 tonne per year to provide a more circular and decarbonised economy. A LCA was conducted using the EASETECH software. Four scenarios were evaluated: anaerobic digestion, composting, incineration, and landfill. Of these, incineration resulted in the highest CO₂-eq savings (-192 kg CO₂-eq/tonne FW), but lacked decoupling and circularity of resources. Conversely, anaerobic digestion demonstrated the highest circularity and lowest toxicity. Based on these findings, anaerobic digestion was selected for further investigation. Economic transactions for the anaerobic digestion system's business models were analysed,

including revenues, municipality fees and operating costs. The new economic model is expected to align with circular economy strategies and promote stakeholder collaboration as a significant social outcome.

The thesis study attempted to understand whether Istanbul's ZW to landfill targets are sufficient to achieve carbon decoupling (i.e. to understand whether carbon emissions decrease as waste increases) by 2050. A novel approach combining MFA and LCA is employed to analyze WM decoupling in Istanbul. The reference flow is the annual waste production in Istanbul, projected at 7.4, 9.2, and 11.2 million tonnes in 2020, 2035, and 2050, respectively. Future WM strategies include landfill diversion of food and packaging waste, enhancing landfill gas capture, increasing recycling rates and establishing anaerobic degradation and incineration facilities. The baseline scenario indicates a decrease in landfill diversion (15.5% to 11.1%) and an increase in carbon emissions (2.06 million tonne CO₂/FU to 3.68 million tonne CO₂/FU) from 2020 to 2050. In contrast, the improved scenario exhibits a significant increase in landfill diversion (15.5% to 78.0%) and a reduction in carbon emissions (2.06 million tonne CO₂/FU to -1.38 million tonne CO₂/FU), resulting in a net saving. However, the improved scenario may not meet the 60% and 100% landfill diversion targets by 2035 and 2050, respectively. All future scenarios have lower net cost (€/ton) than the business-as-usual (BAU) 130€/ton for 2035. This is primarily due to the reduction in landfill cost and revenues coming from AD and incineration plant. All future scenarios have lower net cost (€/ton) than the BAU 160€/ton for 2050. These consist primarily of waste from municipal waste recycling, energy recovery from AD facilities and incineration.

In addition to the environmental and economic assessments the methodological multiple-criteria decision analysis (MCDM) framework was described in the PhD study. Because identifying optimal WM strategies requires coordination of every level of stakeholders. WM systems consist of long-lived infrastructure, decision-makers need to proactively plan for changing the composition of waste and policy requirements and an changing the energy system for cost-effective and sustainable management of future solid waste. The indicators have been identified based on the following requirements: meaningful, clear in content, appropriate in scale, robust and reproducible, comparable describe and understandable. The environmental indicators included in the study were those typically considered in LCA studies. For the MCDM study; circularity, climate change, particulate matter, eutrophication terrestrial and eutrophication marine were selected as environmental sub-indicators. Capital/investment costs, operation cost, public fee and revenue were considered for the economic sustainability assessment. Social indicators; considered within the decision-support framework were classified into five sub-categories namely; partnership potential, job creation, urban space requirement, public acceptability and public health. Finally, technical indicators; were classified into three categories as, flexibility, reliability and qualified employee. A variety of WM options were assessed using (AHP and TOPSIS methods) suggested framework to select the most suitable food WM technology in Istanbul. In conclusion, the organic fraction of MSW can be processed by setting up an AD facility in Istanbul as it is the best option in terms of environmental, economic, technical and social aspects.

As a result, MSW management in Istanbul changed from a simple model of combined landfilling and low recycling rate to in a combination of high recycling (MRFs), incineration with energy recovery and anaerobic digester facility. All of these improvements contribute transforming into a circular metabolism in Istanbul. As well

as improving recycling rates and reducing municipal solid WM emissions, attention must also be paid to the quality of secondary products derived from waste. The enhanced recycling rate increases the amount of secondary material returned to the metabolic process, thereby contributing to the circular economy. Improving the quantity and quality of materials recycling relies on public involvement and improving public awareness will be key to this enhancement. This study contributes to the existing literature by establishing quantitative relationships between WM emissions and goal-based WM strategies, emphasizing the importance of informed decision-making to achieve sustainable WM and decoupling of emissions. The thesis provided a map of the current knowledge of the WM system, detailed the practices and abilities required for the success of ZW initiatives and presented a practical and flexible context.





SIFIR ATIK YÖNETİMİNİN ÇEVRESEL VE EKONOMİK SONUÇLARININ DEĞERLENDİRİLMESİ

ÖZET

Katı atık üretimi, doğal kaynakların tükenmesi nedeniyle günümüzde önemli bir çevre sorunudur. Miktarı hızla artan atıkların yönetimi gün geçtikçe zorlaşmaktadır. Bu durum son yıllarda sıfır atık (SA) olarak adlandırılan yeni bir kavramın gelişmesine neden olmuştur. SA'nin temel amacı, atıkları maddenin yaşam sonu yerine bir ara aşama olarak ele alarak kaynakların korunmasıdır. SA için diğer bir yaklaşım, son yıllarda popüler olan “Düzenli depolama tesislerine (DDT) atık göndermeme veya SA gönderme veya atık saptırma” hedefidir. AB ülkeleri, 2035 yılına kadar belediye atıklarının en fazla %10'unu düzenli depolamaya göndermeyi ve geri dönüşüm oranını %50'den %65'e çıkartmayı hedeflemektedir.

Kentsel katı atıklar'ın (KKA) daha iyi yönetimini kolaylaştırmak için bir dizi önlem alınmış ve tesisler kurulmuştur. DDT'ye SA gönderimine dair hedefler, Türkiye'de dahil olmak üzere dünya çapında önemli bir ilgi kazanmıştır. Türkiye'deki durum incelendiğinde 2003-2017 yılları arasında DDT sayısının 15'ten 87'ye çıktığı görülmektedir. Vahşi depolama sorununa çözüm olarak inşa edilen DDT'ler aslında atık hiyerarşisinde en alt sırada yer almaktadır. Türkiye yılda yaklaşık 31,5 milyon ton atık üretmektedir. Buna rağmen 2020 yılında atıkların %17'si hala belediye çöplüklerine gönderilirken, yalnızca %13,2'si geri dönüştürülmüştür. Türkiye, Sıfır Atık Yönetmeliği ile 2019 yılında yeni bir hedefi gündeme getirmiştir. 2019 yılında yayınlanan Sıfır Atık Yönetmeliği, Türkiye'de eğitim kurumları da dahil olmak üzere birçok kurum için SA uygulamalarına başlamayı zorunlu hale getirmiştir. Ülkedeki mevcut SA uygulamaları atıkların DDT'den uzaklaştırılması üzerine yoğunlaşmaktadır. Bertaraf tesislerine gönderilen atık miktarının azaltılması önemli olmakla beraber, bu gelişmenin ne gibi çevresel ve ekonomik değişimlere neden olduğu incelenmelidir. Bu tezin amacı, giderek artan şekilde uygulanan SA uygulamaları bağlamında atık yönetim senaryolarının çevresel ve ekonomik sonuçlarını değerlendirmektir. Ayrıca bu çalışma, KKA'nın sürdürülebilirliğini SA hedefi aracılığıyla çoklu ölçekte (İstanbul ve Ayazağa Kampüsü) değerlendirmektedir.

Tez, kurumların malzeme akış analizi (MAA) ve yaşam döngüsü değerlendirmesi (YDD) araçlarını kullanarak hedefe yönelik bir atık yönetim stratejisi geliştirmelerine yardımcı olan bir çerçeve sunmaktadır. Çerçevenin sunulmasına ek olarak, birincil ve ikincil veriler kullanılarak kampüs ölçeğinde bir vaka çalışması yapılmıştır. Çerçeve, önceden veri bulunmadığını varsaymaktadır. Vaka çalışması, kampüsteki birincil verileri toplayarak başlamaktadır. İstanbul Teknik Üniversitesi'nin Ayazağa Yerleşkesi SA yönetim (2019) öncesi ve (2022) sonrasında birincil ve ikincil verileri toplamak üzere saha çalışmaları için (i) akademik (ii) idari (iii) konut/yurt ve (iv) kafeterya olmak üzere dört farklı gruba ayrılmıştır. Önce ilk saha çalışması yapılmış (atık karakterizasyonu ve geri dönüşüm potansiyelinin belirlenmesi) ve ardından yeni konteynerler yerleştirilmiştir. Öğrencilere ve kampüs personeline SA yönetimi uygulamaları kapsamında hem yüz yüze hem de online seminerlerle eğitimler verilmiştir. Daha sonra ikinci saha çalışması tamamlanmıştır. Vaka çalışmasının sonuçlarına göre, bölgelerde atık üretim oranı 0,045-0,190 kg/kişi/gün arasında değişmektedir ancak ikinci saha çalışmasında bu oran 0,011-0,117 kg/kişi/gün'e düşmüştür. İlk saha çalışmasında potansiyel geri dönüşüm oranı %76,3 iken ikinci

çalışmada bu oran %68,2 olmuştur. Ayrıca MAA sonuçları, DDT sapmasının %29,8 ile ~%99-100 arasında değiştiğini göstermektedir. DDT'lere giden atık miktarı çok azalsa da yakma tesisinden çıkan kalıntıların veya küllerin bir kısmı yine de DDT'de bertaraf edilecektir. Ayrıca, atıkları düzenli depolama alanından başka yöne yönlendirmek her zaman döngüsellğe neden olmamaktadır veya SA hedeflerine yönelik sürdürülebilir tüketim ve halk davranışlarını doğrudan ele almamaktadır. Çalışmada önerilen atık çerçevesi, kampüs tabanlı ve hedef odaklı atık yönetim çalışmalarındaki potansiyel zorluklara önceden cevap vermeyi amaçlamaktadır. Diğer kurumlardan ve kampüslerinden gelecekte yapılacak vaka çalışmaları, bu metodolojinin doğrulanmasına ve geliştirilmesine yardımcı olacaktır. Ayazağa atık karakterizasyon sonuçlarına göre, SA yönetim sistemi kurma çalışmaları da atık oluşumunda azalma ve geri dönüşüm performansında artış sağlamıştır. Ancak, halk bilinçlendirme faaliyetlerini değerlendirmek için daha fazla çalışmaya ihtiyaç vardır.

Çerçeve ve vaka çalışmalarına ek olarak, İTÜ Ayazağa kampüsü temel alınarak daha döngüsel ve karbondan arındırılmış bir ekonomi sağlamak için yılda 577 ton kaynağında ayrılmış gıda atığını fonksiyonel birim kabul eden bir vaka örneği geliştirilmiştir. YDD sürecinde EASETECH programı kullanılmıştır. Anaerobik çürütme, kompostlaştırma, yakma ve düzenli depolama olmak üzere dört senaryo değerlendirilmiştir. Bunlardan yakma, en yüksek CO₂ eşdeğeri tasarrufuna sahipken (-192 kg CO₂-eş/ton gıda atığı), kaynakların korunmasından ve döngüsellğinden uzak kalmıştır. Yakmanın tersine, anaerobik çürütme en yüksek döngüsellği ve en düşük toksisiteyi göstermiştir. Bu bulgulara dayanarak, ileriki araştırmalar için anaerobik çürütme senaryosu seçilmiştir. Anaerobik çürütme sisteminin iş modeli için gelirler, belediye ücretleri ve işletme maliyetleri dahil olmak üzere tüm ekonomik işlemler değerlendirilmiştir. Anaerobik çürütmeyi temel alan yeni ekonomik modelin döngüsel ekonomi stratejileriyle uyumlu olması ve önemli bir sosyal sonuç olarak paydaş işbirliğini teşvik etmesi beklenmektedir.

Tez çalışması, İstanbul'un DDT'ye sıfır atık gönderimine dair hedeflerinin, 2050 yılına kadar karbon emisyonları üzerindeki etkisini hesaplamıştır. İstanbul'daki atık yönetiminin karbon emisyonları üzerine etkisini analiz etmek için MAA ve YDD'yi birleştiren yeni bir yaklaşım kullanılmıştır. Referans akış, 2020, 2035 ve 2050'de sırasıyla 7,4, 9,2 ve 11,2 milyon ton olarak öngörülen İstanbul'daki yıllık atık üretimi olarak kabul edilmiştir. Gelecekteki atık yönetim stratejileri arasında, gıda ve ambalaj atıklarının düzenli depolamaya yönlendirilmesi, depo gazı yakalamasının artırılması, geri dönüşüm oranlarının artırılması, anaerobik çürütme ve yakma tesislerinin kurulması yer almaktadır. Temel senaryo, 2020'den 2050'ye kadar düzenli depolama saptırmasında bir azalma (%15,5'ten %11,1'e) ve karbon emisyonlarında bir artışa (2,06 milyon ton CO₂/FB'dan 3,68 milyon ton CO₂/FB'ya) işaret etmektedir (FB: fonksiyonel birim). Net sıfır hedefi için İstanbul Büyükşehir Belediyesi tarafından geliştirilen iyileştirilmiş senaryo düzenli depolamaya gönderilen atıkların (DDT'den sapma %15,5 ila %78,0) ve karbon emisyonlarının azalmasına (2,06 milyon ton CO₂/FB ila -1,38 milyon ton CO₂/FB), neden olarak karbon emisyonları açısından net tasarruf sağlamaktadır. Ancak, MAA hesapları iyileştirilmiş senaryonun, sırasıyla 2035 ve 2050 yılları için konulan %60 ve %100 DDT'den atık saptırma hedeflerini karşılayamayacağını göstermektedir. Tüm gelecek senaryolarının net maliyeti (€/ton), 2035 (130€/ton) ve 2050 (160€/ton) olağan (temel) senaryolarından düşük net maliyete sahiptir.

Doktora çalışmasında çevresel ve ekonomik değerlendirmelere ek olarak Çok Kriterli Karar Verme (ÇKKV) yönetimini içeren bir metodolojik çerçeve de önerilmiştir. Çünkü en uygun atık yönetim stratejilerini belirlemek için her düzeydeki paydaşın koordinasyonu gerekir. Atık yönetim sistemleri uzun ömürlü altyapı sistemlerinden oluşur; karar vericilerin atık bileşimini ve politika gerekliliklerini değiştirmeyi ve gelecekteki katı atıkların uygun maliyetli ve sürdürülebilir yönetimi için enerji sistemini değiştirmeyi proaktif bir şekilde planlamaları gerekir. Önerilen çerçeve içindeki göstergeler: anlamlı, içerik açısından açık, ölçeğe uygun, sağlam ve tekrarlanabilir, karşılaştırılabilir ve anlaşılardır. Çalışmaya dahil edilen çevresel göstergeler, YDD çalışmalarında tipik olarak dikkate alınan göstergelerdir. ÇKKV çalışması için; döngüsellik, iklim değişikliği, partikül madde, karasal ötrofikasyon ve denizlerdeki ötrofikasyon çevresel alt göstergeler olarak seçilmiştir. Ekonomik sürdürülebilirlik değerlendirmesi için sermaye/yatırım maliyetleri, işletme maliyeti, vergiler ve gelirler dikkate alınmıştır. Sosyal göstergeler; ortaklık potansiyeli, iş yaratma, kentsel alan gereksinimi, halkın kabul edebilirliği ve halk sağlığıdır. Teknik göstergeler; esneklik, güvenilirlik ve nitelikli çalışan olmak üzere üç kategoride sınıflandırılmıştır. İstanbul'daki en uygun gıda yönetimi teknolojisini seçmek için önerilen çerçeve (AHP ve TOPSIS yöntemlerini) kullanılarak çeşitli atık yönetimi seçenekleri değerlendirilmiştir. Sonuç olarak, KAA'ların organik kısmı için, çevresel, ekonomik, teknik ve sosyal açıdan en iyi seçenek olduğundan İstanbul'da anaerobik çürütme tesisi kurulması önerilmiştir.

Sonuç olarak, İstanbul'daki KKA yönetiminin düzenli depolama ve düşük geri dönüşüm uygulamalarından oluşan modelden, yüksek geri dönüşüm, enerji geri kazanımlı yakma ve anaerobik çürütücü tesisinin kombinasyonunu içeren modele dönüşmesi önerilmektedir. Tüm bu gelişmeler, İstanbul'un döngüsel kentsel metabolizmaya dönüşmesine katkı sağlayacaktır. Geri dönüşüm oranlarını iyileştirmenin ve KAA yönetim emisyonlarını azaltmanın yanı sıra, atıktan elde edilen ikincil ürünlerin kalitesine de dikkat edilmelidir. Artan geri dönüşüm oranı, sistemde geri dönen ikincil malzeme miktarını artırarak döngüsel ekonomiye katkıda bulunacaktır. Fakat geri dönüşüm malzemelerinin miktarının ve kalitesinin iyileştirilmesi, halkın katılımına bağlıdır ve halkın farkındalığını artırmak, bu iyileştirmenin anahtarı olacaktır. Bu çalışma, atık yönetim emisyonları ile hedefe dayalı atık yönetim stratejileri arasında nicel ilişkiler kurarak, sürdürülebilir atık yönetim ve emisyonların ayrıştırılması için bilinçli karar vermenin önemini vurgulayarak mevcut literatüre katkıda bulunmaktadır. Tez, atık yönetim sisteminin mevcut bilgilerinin bir haritasını sunmuş, SA yönetim girişimlerinin başarısı için gerekli olan uygulamaları detaylandırmış ve emisyonların azaltılabilmesi için pratik ve esnek bir bağlam sunmuştur.



1. INTRODUCTION

1.1 Background

There is a growing concern about natural resources and waste management (WM) practices in the world. Increasing population and urbanization are leading to increase in municipal waste generation. Material consumption will be doubled by 2060 and waste generation is expected to increase by 70% by 2050 (OECD, 2018; EC, 2020). The traditional, linear “take, make, use, dispose” WM model is potentially reaching its limits and the availability of raw materials and energy cannot be guaranteed any more (BSI, 2017). According to World Bank data, domestic waste production in the world is approximately 1.3 billion tons/year and it is expected to increase to 2.2 billion tons/year by 2025 (World Bank, 2010). In order to meet the demands and face the challenges proposed by society and government, concrete actions need to be developed for a more efficient production, with less generation of feed and less extraction of virgin raw material from nature. This approach supports several Sustainable Development Goals (SDGs) such as Goal 7, ensure access to affordable reliable, sustainable and modern energy; Goal 8, promote inclusive and sustainable economic growth; Goal 9 promote resilient infrastructure; Goal 11 ensure sustainable consumption and production patterns; Goal 13 actions to combat climate change (UN, 2015).

Urbanization is increasing day by day, due to the developing technology and modern lifestyle. 68% of the world's population will live in cities by the year 2050 (UN, 2018). “Cities occupy just 3% of the Earth’s land but account for 60-80% of energy consumption and at least 70 % of carbon emissions” (UN, 2018). Cities throughout the world are transforming, in order to adapt to climate change. Furthermore, urban living conditions are now coming under pressure as climate change becomes more and more evident. Intergovernmental Panel on Climate Change (IPCC) stated that limiting global warming to 1.5 °C would require rapid, far reaching and unprecedented changes in all aspects of society (IPCC, 2018). In order to thrive and survive, cities have to

respond to these challenges and many more. Adaptation plans to counter climate change vary globally across cities, and only a fraction of cities have a plan, with even fewer in the implementation stage. As the knowledge economy is emerging, in which universities are now considered an essential element of regional development. The university has expected to take a proactive role in the innovation value chain. Universities can be seen as “mini cities” due to their size and the impact of campus activities on the environment and society in their pursuit of achieving sustainability (Alshuwaikhat and Abubakar,2008; Lauder et al., 2015).

Sustainable actions are possible when performed along with practices based on circular economy (CE) in the different sectors of the economy. However, sustainable and efficient WM is still not widespread in the World (Zaman, 2014) and WM problems has not been fully solved (Adeniran et. al. ,2017; Zaman, 2014). Waste problems are growing and problem solving become harder for local administrations (Zaman, 2014). Therefore, the efficient implementation of the zero waste (ZW) management philosophy, which includes waste prevention, efficient use of resources, minimization of waste generation, separate collection and recovery, is important for environmental and economic aspects. ZW has become an ambitious goal over the past decade. ZW refers to the systematic design and management of products and processes to avoid and dispose of waste and materials, and retain and retrieve all resources from waste streams (Zaman and Lehmann 2011). However, there is no consensus on ZW definition and ZW concepts (Zaman and Lehmann 2011). The business and institutional communities are key to this transition and an increasing number of companies are setting waste reduction targets, such as "zero waste to landfill" (Veleva et al, 2017).

1.2 Literature Review Statement of Topic (Problem statement)

Governments and local authorities create their own strategy for reaching ZW goal since there is no one single rule. The goal in legislations could cover ZW to landfill and ZW to incineration plant or both them. However, ultimate goal is ZW production in a first place. Turkiye has been establishing landfills rapidly and the number of landfills has been increased from 15 to 87 between 2003 and 2017. Turkiye generates 32.3 million-ton waste in a year (TSI, 2021). Despite the increased number of the landfills 17 % of waste was still sent to open dumps while only 13.2% was recycled in

2020 (TSI, 2021). This is the highest landfilling rate relative to EU countries (Figure 1.1). The Zero Waste Regulation Legislation was published in 2019. According to the regulation not only municipalities but also private and public institutions are responsible for reaching ZW goal to landfills. Beştepe Presidential Complex in Ankara is the first Zero Waste Management pilot area in Türkiye. Other public institutions and universities have started ZW projects since then.

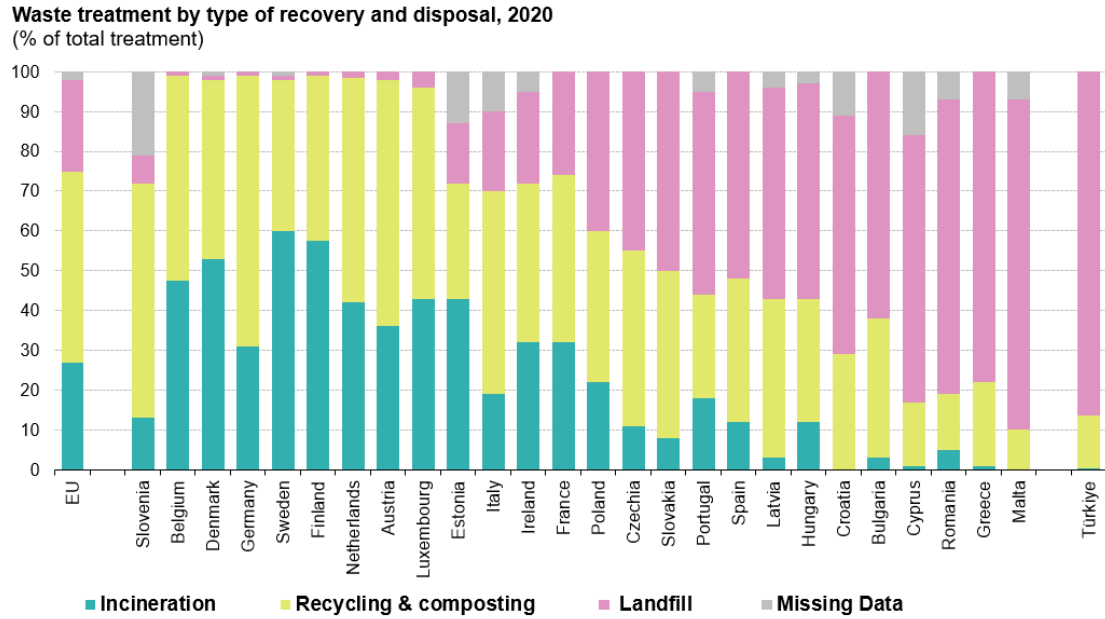


Figure 1.1 : Waste Treatment Types in Europe and Türkiye

To solve complex systems like WM, future pathways should be carefully planned considering the future availability and characteristics of waste streams as well as changes in the background system (i.e., the surrounding economic sectors) to avoid potentially bad decisions and to guarantee the overall best environmental outcome. Environmental impacts of WM could be quantified with a life cycle thinking perspective through life cycle assessment (LCA). The LCA is a standardized method which evaluates environmental impacts related to the use of resources and emissions into the environment from all processes involved in the study (ISO, 2006a; 2006b). *“Material Flow Analysis (MFAs) help to understand the stocks and flows of a WM systems and are a natural starting point for an environmental assessment. MFA and LCA have been applied in parallel for waste management decision support, providing information on the environmental performance of different mass flow scenarios. The combination of regional MFA and LCA allows for assessing environmental impacts*

while ensuring consistency between variables such as MFA and LCA process model transfer coefficients, capacity restrictions of waste-treatment infrastructure, and waste resource availability” (Haupt et al., 2018a).

From the assessed literature, the following research gaps have been identified:

- ❖ There is a lack of studies coupling mass flow analysis, as well as environmental and economic assessments, to evaluate multi-stakeholder approach to WM.
- ❖ There is a lack of studies to identify WM data quality, system gaps and make recommendations to relevant stakeholders.
- ❖ The application of environmental and economic assessments on different scales of ZW management.
- ❖ Providing an improved framework for goal and scope definitions in environmental and economic assessments evaluating ZW management.
- ❖ The environmental and economic assessments of shifting from linear to circular systems can also be quantified in case of business model changes.

1.3 Aim and Scope of the Thesis

Realizing the gaps in the literature (see Section 1.2) is inherently intended to cause large-scale effects on the WM sector that have still to be determined. The goal of this thesis is to assess the environmental and economic results of WM pathways within the context of an increasingly applied ZW approach. The PhD study will aim to answer following questions and the research questions established are addressed through case studies of ITU Ayazağa Campus (campus scale) and Istanbul (urban scale);

1. What are the data and information gaps and how can these gaps be filled to strengthen the WM system?
2. Whether the ZW to landfill goal is possible and is landfill diversion always the best option in terms of environmental impacts and circularity?
3. Which environmental changes are expected with the ZW management?
4. What criteria can assess eco-efficiency of the ZW management system?

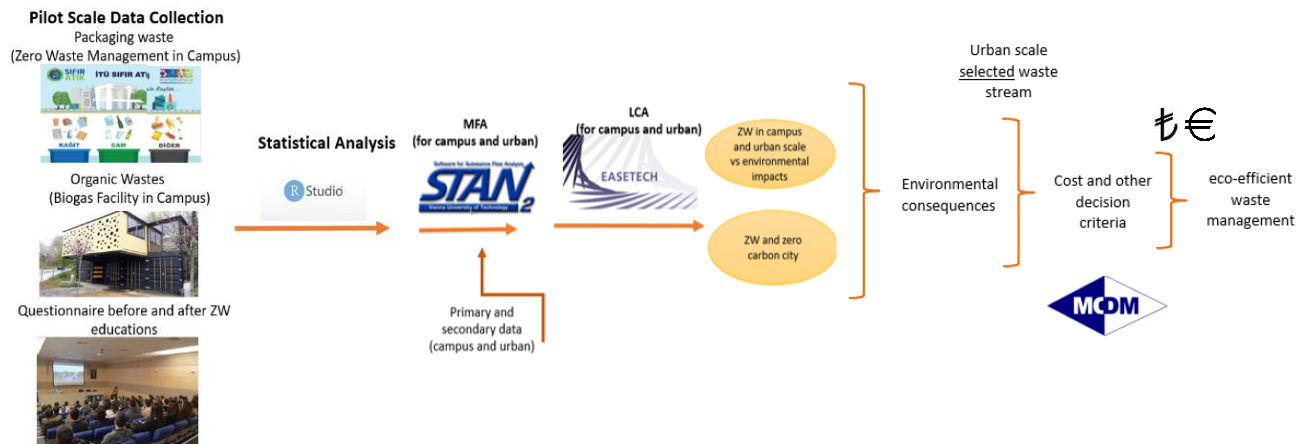


Figure 1.2 : Thesis Methodology

This thesis develops new adapted tools capable of tackling the complexity of WM and anticipating consequences at the system level. First, data quality analysis (DQA) was conducted to all data inventory to address Q1. Secondly, a MFA and LCA framework was developed for institutions to answer Q2 and Q3. Finally, the eco-efficiency tool was developed for a comprehensive assessment of economic, social and technical environmental indicators using the multi-criteria decision-making methodology (MCDM) to respond to Q4 (Figure 1.2).

1.4 Thesis Outline

The thesis is divided into the following chapters:

Chapter-2: discusses the results of the field studies and questionnaires in terms of waste characterization and recycling potential on the university campus.

Chapter-3: describes the “zero waste” MFA-LCA framework strategies for institutions.

Chapter-4: presents a business model for food waste management at ITU for decarbonising university campuses.

Chapter-5: presents the results on the decoupling analysis of WM emissions in Istanbul.

Chapter-6: describes the eco-effinct WM strategy by using Multi Criteria Decision Making methods.

Chapter-7: This section gives a brief summary and critique of the findings and offers future perspectives.

Chapters 2, 3, 4 and 5 representing articles I-II-III and IV respectively (Figure 1.3).

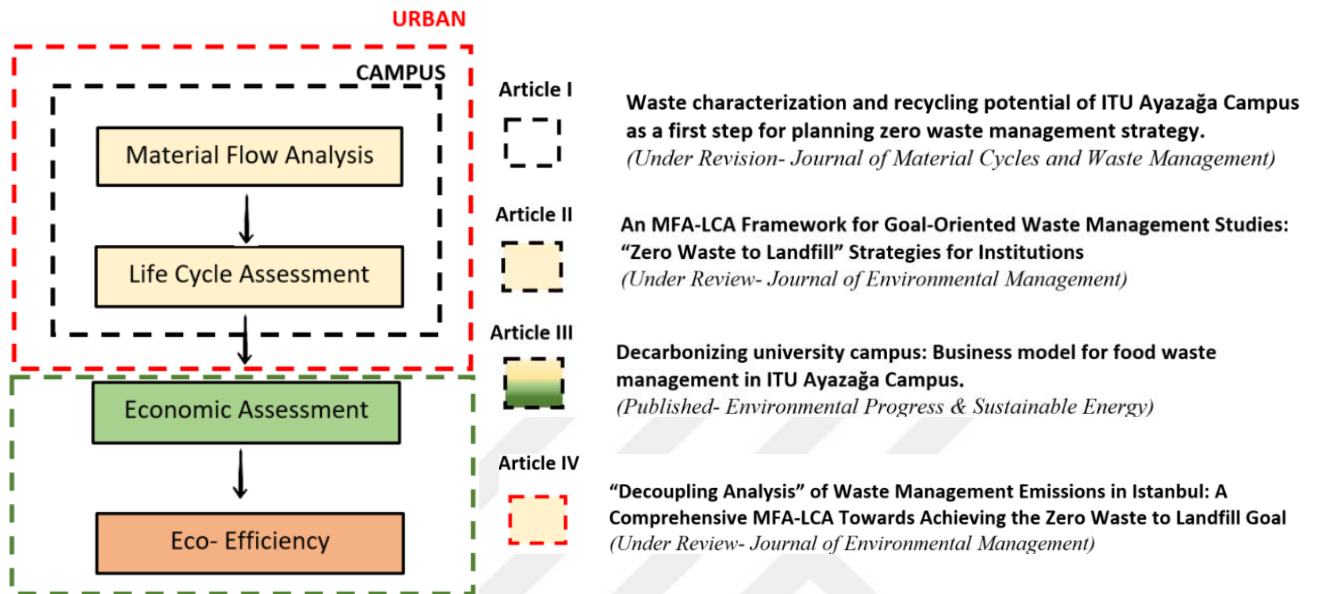


Figure 1.3 : Thesis articles

2. WASTE CHARACTERIZATION AND RECYCLING POTENTIAL IN A UNIVERSITY CAMPUS: ITU AYAZAĞA CAMPUS ZERO WASTE MANAGEMENT PRACTICES

2.1 Introduction

Today, the growing population and the production of solid waste have led to the rapid consumption of scarce resources (Fan et. al., 2021; Haksevenler Gürsoy et. al., 2022). Solid waste management (WM) problems are not fully resolved (Zaman, 2014; Adeniran et al., 2017). This ongoing problem has led to the development of a new concept called zero waste (ZW) (Veleva et al., 2017). The ZW concept aims to reduce the consumption of natural resources and materials. In order to achieve this goal, recycling must be maximized and the amount of waste sent to final disposal must be reduced to a minimum as much as possible in line with the life-cycle approach, starting from the design phase, including the disposal phase (Zaman, 2015; 2017). In addition, ZM aims to prevent possible negative impacts of solid waste (Özbay and Gokceviz, 2022) by preventing and reducing the solid waste generation and increasing the recycling rates (Fagnani and Guimarães, 2017). Local authorities are trying to adopt ZW strategies to reduce waste generation in order to protect the natural resources. Universities can be considered “mini cities” due to their size and the impact of campus activities on the environment and society in their pursuit of sustainability (Alshuwaikhat and Abubakar, 2008; Lauder et. al., 2015). WM studies on university campuses provide preliminary data on a pilot-scale and guide full-scale projects. They also have an impact on society by adopting new strategies, behavioral changes (Moqbel, 2018) and public awareness (Maçın et. al. 2020; Dilkes-Hoffman et. al. 2019) as universities are considered to be of high reputation (Armijo de Vega et al., 2008).

ZW studies in universities around the world started in the late 90s (Mason et al., 2003). and with green campus activities, it has begun to attract more attention in the developing world in recent years (Özverel, 2014; Okutan, 2017). Studies on university campuses deserve more attention since significant amount of solid waste generated due to their large population and variety of activities taking place on the campus.

Waste characterization is the first step in pursuing a comprehensive and sustainable WM strategy (Armijo de Vega et al., 2008). Furthermore, the characteristic changes of waste vary by location, population, climate and culture (Tchobanoglous et. al, 1996; Smyth et. al. 2010; Ayeleru et. al., 2018; Mbama et. al., 2022). University's waste characterization is also a relatively new topic compared to other characterization studies. Previous studies have shown that university waste characterization differs from municipal waste characterizations in terms of ratio of recyclables (Armijo de Vega et al., 2008; Smyth et. al. 2010; Mbama et. al., 2022; Gallardo et. al., 2016).

There has been increasing interest in WM campus characterization studies in Türkiye in the past decade (Banar and Özkan, 2005; Irbaş et. al., 2012; Işıldar and Öztürk, 2015; Yalvaç et al., 2014; Demirarslan et. al., 2017). However, most of the previous studies have focused on recyclable waste, particularly on the amount and economic value of packaging waste (Işıldar and Öztürk, 2015; Özer et. al., 2015). Rather than examine all subcategories of waste composition and comprehensive evaluation with different university campus studies in terms of waste generation rates. Furthermore, higher education institutions (HEIs) are responsible for achieving the ZW landfill target in Türkiye, in compliance with the Zero Waste Regulation (MEU, 2019). Few studies have been published recently on ZW and university campuses (Gündüz, 2021; Akın, 2020; Kavak, 2020; Bahçelioglu et. al., 2020; Gül, 2019). Some of these studies are comprehensive that include both waste characterization and recycling potential for planning a new WM strategy. For instance, Middle East Technical University (METU) has focused on, waste management strategies, compost potential and attitudes and awareness on solid waste management in the campus (Bahçelioglu et. al., 2020). Marmara University has extensively examined the applicability of separate waste collection on a university campus from both technical and social perspectives (Haksevenler Gürsoy et. al., 2022). Another study was conducted at Namık Kemal University to determine the effectiveness of recycling containers in meeting the management objective of ZW (Gül, 2019). However, these studies do not have suggestions for improving ZW strategy by comparing WM activities before and after ZW according to the best of our knowledge.

Although many WM studies have been carried out in universities, most recent studies only evaluate the current situation and do not replicate the study in following years to monitor the impact of the new strategy (Bahçelioglu et. al., 2020; Adeniran et al., 2017;

Armijo de Vega et al., 2008). The effectiveness of a recycling program at a university in a developing country has been investigated before (Mbama et. al., 2022). However, as previously suggested by Cole et. al. (2014), it is important to assess whether the ZW has been successfully implemented makes a difference in a positive manner. Thus, the aim of this study is to determine the waste characterization and recycling potential of the ITU Ayazağa Campus for developing a new ZW strategy and assessing its performance. In this context, the paper addresses following questions; (i) What is the per capita waste generation, waste characterization, recycling performance, impurities in containers and landfill diversion in ITU Ayazağa Campus before ZW in 2019? (ii) How do these values change in 2022 after ZW management activities? (iii) What are the improvements and strategies for WM that are applicable to achieving the long-term goal of the ZW? Therefore, this paper begins by determining WM status before the ZW in Ayazağa Campus. Second, it continues with evaluation of field research to examine waste characterization and recycling potential before the ZW management. Following the initial field study, new containers were placed within the scope of ZW management. Students and campus personnel were trained on ZW management practices within the face-to-face and online (during Covid-19 restrictions) seminars. The final phase of the study, the second fieldwork, was completed and the results were used to provide suggestions for achieving the ZW goal in long-term.

2.2 Materials and method

2.2.1 Study area

Waste characterization studies were conducted on ITU Ayazağa Campus in 2019 and 2022. The campus is located in Maslak- Sarıyer district, which is one of the largest business and commercial center of Istanbul, Türkiye. The campus covers an area of 154 hectares. There are 35,527 and 37,215 registered students, and 3,707 and 3,304 staff on campus in the academic years of 2018/2019 and 2021/2022, respectively. In addition to the Rectorate and the administrative units, there are 8 faculties and 4 institutions in Ayazağa Campus. Central Library, Central Dining Hall, dormitories and Techno-parks are located on this campus. Figure 2.1. shows the map of Ayazağa Campus provided by Quantum Geographic Information System (QGIS).

2.2.2 Waste management in the campus

ITU has been conducting separate collection of recyclable wastes since 2013. In 2019, packaging wastes were collected separately in blue containers, while other wastes were collected in grey containers at the Ayazağa Campus. These wastes are collected daily by the contracted companies of the municipality. Medical wastes are collected every week, while hazardous waste collection frequency changes according to the waste generation. Following the adoption of ZW management, new containers were placed across the campus. Green glass waste containers were also placed to the first floor of each building.

2.2.3 Pilot areas and field studies

Waste characterization preliminary studies began in January 2019. Following the permission given by the rectorate for the study, a contact list of responsible staff on waste management was received. Before starting the study, interviews were held with building managers and cleaning service staff. The campus was divided into four main categories: which are (i) academic (ii) administrative (iii) residential and (iv) cafeteria/canteen. All waste generation locations were determined and categorized according to this classification, and four buildings were selected from each group as a pilot area. (i) Faculty of Civil Engineering was selected as an academic unit while (ii) Student Affairs Building was selected as an administrative unit, (iii) Girls' Dormitory was selected as a residential/dormitory unit and (iv) Central Dining Hall was selected as a cafeteria. The study areas were shown in Figure 2.1. This study was planned and conducted only for non-hazardous waste.

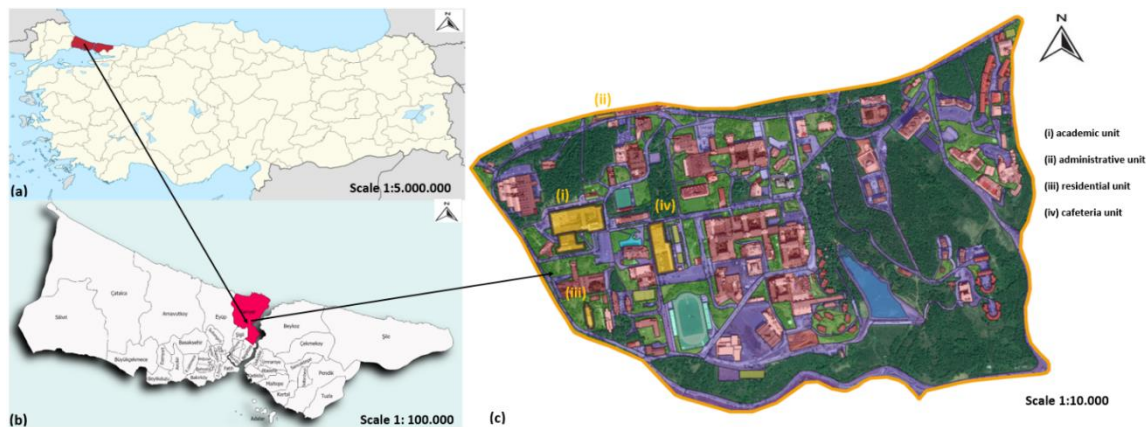


Figure 2.1 : Map of the (a) Türkiye (b) Istanbul and (c) ITU Ayazağa Campus.

2.2.4 Waste sampling

Most of the previous studies, especially those using the American Society for Testing and Materials (ASTM) method, use statistical methods (Gebreyessus et. al., 2019; Gallardo et. al., 2016) to determine sampling size (Adeniran et al., 2017; Taghizadeh et al., 2012). Others use different techniques to decide sampling size (Armijo de Vega et al., 2008; Smyth et. al. 2010). Various approaches exist to obtain quantitative WM data such as going through previous WM records, personal communication with the WM staff and exploring data from other institutions (Smyth et. al. 2010). However, it was decided that the most reliable method to analyze both the amount of waste and the composition of waste is to separate all waste from the pilot areas.

Sampling was conducted in April 2019 (first fieldwork) and May 2022 (second fieldwork) for one week, Monday to Friday. Waste samples were sorted by volunteer undergraduate and graduate students with the help of cleaning staff. During sampling; labeled baskets, personal protective equipment (workwear, glasses, gloves), plastic bags and a digital scale were used. The selected blue and grey containers were marked with project name “Zero Waste Management in ITU” and related information was given to the pilot area building managers and cleaning staff. In addition, the same information was given to surrounding buildings managers to reduce possible interferences. Arrangements were made with waste collection truck drivers and their supervisors. Wastes were collected under in a controlled manner between 7:00-9:00. In 2019, there were only grey containers in the dining-hall, and there was no source separation for plastic caps in a separate container. Packaging waste was separated in the kitchen during the food preparation process. Therefore, that area was considered the blue container of the dining room. After the first preliminary studies, the ZW management system on the campus has been initiated; the number of bins has been increased, labeled and size standardized in 2020 and 2021. In addition, face-to-face and online trainings were given to the residents of the pilot area (students and staff).

2.2.5 Waste amount, waste characterization and landfill diversion

Waste characterization categories were prepared by adapting (Adeniran et al., 2017; Armijo de Vega et al., 2008) studies and shown in Figure 2.2.

Organic waste (raw) • Fruit • Leaves, grass • Vegetables	Organic waste (cooked) • Food residuals	Glass • Green bottles • Amber bottles • Other glasses • Clean glasses	Cardboard-paper • White paper, exam papers • Books • Newspaper, magazines, • Other cardboard	Aluminum • Aluminum cans	Other metals • Tin • Mixed metals • Non-aluminium materials	Textile • Clothing • Shoes	
Other Plastic • Foam • Food containers (yogurt, etc.) • Mixed plastics	PET • PET bottles	Plastic Bags • Cargo and groceries bags • All kind of nylons	Tetra-park • Juice and milk boxes	e-waste* • Cable • Headphones • Phone accessories • other e-waste	Other** • Sanitary waste • Dirty, hair and miscellaneous materials • Masks, gloves	Inert Waste • Ceramic • Gravel, rocks, sand	Toilet Paper • Paper from bathrooms, offices, hallways

* In order to fully determine the waste and recycling potential from the pilot areas, small sized e-waste was included in the characterization. Computer parts like monitors, keyboards, or large wastes like printers were excluded from the study's scope. In addition, the regulation in Türkiye categorizes e-waste as special waste.

**The mask was categorized under the other category due to hygiene concerns

Figure 2.2 : Waste categories in the study

Wet weight of samples was recorded in excel sheets using the classification in Table 2.1. The per capita waste amount and percentage calculations were made in Microsoft Excel by using Equation (1.1) and Equation (1.2).

$$\text{per capita waste (kg/capita/day)} = \frac{\text{average daily waste produced in pilot area (ton/day)}}{\text{population of the pilot area (capita)}} \quad (1.1)$$

$$\text{waste percentage (characterization) (\%)} = \frac{\text{sub-category waste (ton)}}{\text{total waste produced (ton)}} \times 100 \quad (1.2)$$

Waste categories and recycling potentials are shown in Table 2.1 (Adeniran et al., 2017; Armijo de Vega et al., 2008). Organic wastes, e-waste and textile waste were considered potentially recyclables since they have the potential to be recycled on the campus. Organic waste was divided into two parts as cooked and raw to understand how much food was wasted in cafeterias.

Table 2.1 : Waste categories and their recycling potential

	1 recyclable	2 potentially recyclable	3 non- recyclable
Glass (Clear and colorful)	✓		
Aluminum Cans	✓		
Other Metals	✓		
PET Bottles	✓		
Plastic Bags	✓		
Other Plastics	✓		
E-Waste		✓	
Cardboard-paper	✓		
Organic Waste (Raw)		✓	
Organic Waste (Cooked)		✓	
Textile		✓	
Inert Wastes			✓
Toilet paper/Paper Towel			✓
Tetra pack	✓		
Other			✓

The waste stored in blue containers sent to Material Recycling Facility (MRF). The remaining part sent to a landfill. Waste sent to MRF contained non-recyclable waste, while waste sent to the landfill contained some recyclable waste due to impurities. Normally, material flow analysis should be carried out using the market substitution ratio to determine the actual diversion from landfills. However, the actual diversion (%) was calculated using recyclables in the blue container in this study by using Equation (1.3).

$$\text{actual landfill diversion (\%)} = \frac{\text{recyclable waste in blue containers (ton)}}{\text{total waste produced (ton)}} \quad (1.3)$$

The potential landfill diversion (%) was calculated by using all recyclables and potentially recyclables produced on the campus by using Equation (1.4).

$$\text{potential landfill diversion (\%)} = \frac{\text{recyclable waste produced (ton)} + \text{potentially recyclable waste produced (ton)}}{\text{total waste produced (ton)}} \quad (1.4)$$

2.3 Results and discussions

2.3.1 Waste amount

The waste generation rate in the pilot areas varied between 0.045-0.190 kg/cap/day, while it decreased to 0.011-0.117 kg/cap/day in the second field study (Table 2.2.). Academic unit has lowest per capita results in both years (0.045 and 0.011 kg/cap/day for 2019 and 2022, respectively). Because the majority of the unit's population is students who only use the unit for lecture hours. The remaining small portion of the population was composed of academic and administrative staff who use the unit during the working hours. The administrative unit has the opposite situation. Hence, it has the higher per capita waste generation (0.140 and 0.054 kg/cap/day for 2019 and 2022, respectively) compared to academic unit. In fact, a previous study conducted at the Ayazağa Campus (Maçin et. al., 2020) found high awareness of the administrative staff about WM. However, the waste generated by students awaiting their paperwork may be also added to the calculation of the administrative unit waste amount.

According to the results, per capita waste amount decreased more than 50% in 2022 in pilot areas. The pandemic could be the cause of this decrease. The effects of Covid-19 were not the same everywhere because of the geographic location, way of life and the extent of the impact of the disease on daily activities. A study by Koncagül (2021) found that municipal solid waste production per capita declined by 50% in Yozgat (City of Turkiye). Simililary, in China municipal solid waste generation in cities has decreased 30% (Yousefi et. al., 2021). Besides the pandemic, this also may be related to increased awareness of waste generation.

Table 2.2 : Waste Generation Rate (kg/capita/day)

University/Country	University Waste Generation Rate (kg/capita/day) ^c	Country	Source
Anadolu University/ Turkiye	0.088-0.340 ^a	1.12	Banar and Özkan (2005)
University of Baja California/ Mexico	0.456 ^b	1.04	Armijo De Vega et al. (2008)
Universidad Autónoma Metropolitana/Mexico	0.110 ^c	1.04	Espinosa et al. (2008)
University of Northern British Columbia/Canada	0.059 ^b	2.13	Smyth et al. (2010)
Erciyes University/ Turkiye	0.018	1.12	Irbaş et al. (2012)
Gazi University/ Turkiye	0.205-0.308	1.12	Işildar and Öztürk (2015)
The Universitat Jaume I/Spain	0.089	1.28	Gallardo et al.(2016)
Covenant University/Nigeria	0.304-0.488 ^a	0.49	Coker et al. (2016)
Boğaziçi University/ Turkiye	0.210	1.12	Okutan (2017)
Kotebe Metropolitan University/Ethiopia	0.093	0.30	Gebreeyessus (2019)
Middle East Technical University/ Turkiye	0.380-0.450 ^a	1.12	Bahçelioğlu et al. (2020)
Istanbul Technical University/ Turkiye	0.045	1.12	This Study (2019) Academic Unit
	0.566		This Study (2019) Administrative
	0.140		This Study (2019) Dormitory
	0.190		This Study (2019) Cafeteria
	0.943		This Study (2019) Total
	0.011		This Study (2022) Academic Unit
	0.054		This Study (2022) Administrative
	0.117		This Study (2022) Dormitory
	0.083		This Study (2022) Cafeteria
	0.429		This Study (2022) Total

^a minimum and maximum values

^b calculated by the authors of this study

^c the data does not include food wastes from the cafeteria

Waste production rates of units in 2019 and 2022 fall within the literature range in Table 2.2. But this could also related to a broad range of values in the literature. However, total per capita waste amount of first audit result, 0.943 (kg/capita/day), is higher than all given literature in this study. Second audit result is close to the upper range of the literature. It is also higher than all of the other studies in Turkiye except study in METU, 0.450 (kg/capita/day) (Bahçelioğlu et al., 2020). ITU and METU have largest campuses in the country. Therefore, other potential campus users, such as Techno-Park employees, could be a reason for high waste production.

All the findings of the university campus waste management studies show that per capita waste production on campus is lower than their national levels. The wide differences in the studies may not be directly explained by the properties of the study

area or waste production versus income relationship as concluded in the previous studies for municipal waste (Bandara et. al., 2007; Hoornweg and Bhada-Tata, 2012). For instance, Canada has the highest GDP per capita (The World Data Bank, 2019) compared to other countries given in Table 2.2. However, University of Northern British Columbia (Canada) had 0.059 kg/capita/day waste generation rate which is lower than those in most of the studies. While Ethiopia has the lowest GDP per capita among the countries in Table 2.2, Kotebe Metropolitan University (Ethiopia) has higher per capita waste generation (0.093 kg/capita/day) compared to other studies however, the waste generation rate is lower, 0.059 kg/capita/day, than those in most of the studies (Smyth et. al., 2010). This demonstrates that waste generation in the university have a different tendency than municipal waste. The second audit results are supporting this hypothesis. All these results indicate that a variety of studies should be conducted before desiding WM strategy in the university campus.

2.3.2 Waste characterization

2.3.2.1 Academic unit

In both 2019 and 2022, the largest waste categories were cardboard-paper (28.7%, 23.4%) and toilet paper (15.2%, 19.1%) (Figure 2.3). These were followed by organic waste (7.0%, 13.7%), other (19.4%, 11.0%) and glass (5.0%, 10.3%). Remains, mostly held by other plastics (8.0 %, 5.1 %) and plastic bags (2.9%, 4.2%). Except for Fridays in both 2019 and 2022, the amounts of cardboard-paper were close to each other throughout the week.

2.3.2.2 Administrative unit

Cardboard-pPaper (42.8%), organic waste (cooked) (36.8%), and toilet paper (42.8%) haved the highest waste amounts in this unit in 2019. Organic waste (cooked) generally consists leftovers from takeaway orders. In 2019, glass percentage was 1.5% (Figure 2.3). In 2022, cardboard-paper (28.7%), organic waste (cooked) (11.5%), toilet paper (5.4%) and glass (24.1%) percentages changed significantly. The reason of the significant variation could be the lack of containers for glass waste in the building in 2019. Glass waste accumulated in new green containers during the ZW could explain the high second audit values. During studies, major changes in waste components were not observed throughout the week. On the other hand, post Covid-19 pandemic's impact on waste distribution is inevitable. For example, plastic production has been

found to be much higher during Covid-19 in the USA (Aurpa, 2021). Because online education and working from home have prompted people to order online (Aurpa, 2021). Similar conditions may also explain the shift in plastic waste for the Ayazağa campus.

2.3.2.3 Dormitory unit

Girls' Dormitory wastes contained 14.2% and 10.8% cardboard-paper in 2019 and 2022, respectively (Figure 2.3). Pizza boxes, personal care cream boxes and cardboards from architecture students' old model assignments are the main contributors to cardboard waste. The dormitory waste had 8% and 10.3% of glass in 2019 and 2022, respectively. Most of the glass waste originated from mineral water and glasses of different beverage types. Girls' Dormitory had high amount of organic matter on Monday due to home cooking activities in shared kitchen at weekends. Other waste mostly consists of hygiene products waste, hair and miscellaneous materials. The largest changes observed in organic and other waste categories in 2022. According to Yousefi (2021), organic waste was one of the waste categories most affected by the Covid-19 pandemic, and the findings of the second audit were consistent with this study. (Yousefi et. al., 2021).

2.3.2.4 Cafeteria unit

The amount and composition of packaging waste change according to the menu of the day in the dining-hall. For instance, if there is a bagel,pretzel, etc. on the menu more metal wastes are generated (from oil, cheese, etc.), or if there is yogurt on the menu, more plastic waste is generated since it is served in plastic dish, therefore the composition changes during week. Raw organic waste (74.9% and 64.4%) and cooked organic waste (13.7% and 13.2%) had the majority of waste in 2019 and 2022, respectively (Figure 2.3). The remaining fraction is mostly cardboard-paper and other plastics.

The daily variations of waste characterizations are given in the appendix. The results indicated that PET bottles, other plastics, cardboard-paper, and tetra paks had higher variations than other recyclables. Inert waste and organic waste (cooked and raw) had a higher variance in the categories of non-recyclable and potential recyclable, respectively. E-waste was only generated on Monday during the second audit while it

was generated all throughout the first audit except Tuesday and this leads to significant variations in results.

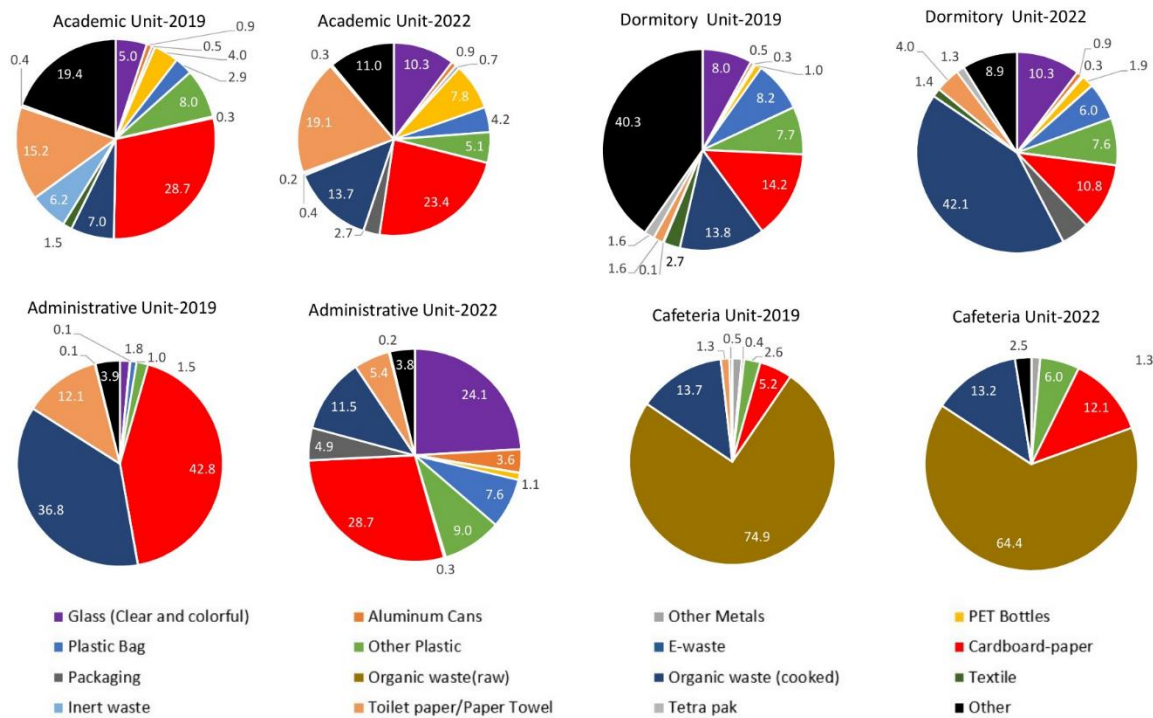


Figure 2.3 : Waste characterization in 2019 and 2022 (%).

2.3.3 Recycling potentials, impurities in the containers and landfill diversion

The recycling potentials (Figure 2.4) of wastes in blue, grey and green (in 2022) containers were calculated according to the classification in Table 2.1. Overall, potentially recyclables were 76.3% in the first field study, and then decreased to 68.2% in the second field study. Recyclables increased from 16.3% to 26.1% in the same period. In addition to recycling potential, impurities in containers were presented in this section. Green containers, blue containers and grey containers are used for glass waste, recyclable waste and other waste, respectively. Remain waste categories in the containers are considered an impurity in this study. To be more precise, recyclable waste such as plastic in a grey container is a class of non-target waste and is therefore considered an impurity in the calculations.

2.3.3.1 Academic unit

Non-recyclable waste ratio decreased from 40.8% to 30.6% due to decrease in other wastes in academic unit while potential recyclables increased from 8.8% to 18.3% (Figure 2.4). Not only the potential quantity of recyclable waste, but also the recycling

performance has been enhanced in this unit. In particular, recycling waste in blue containers increased from 67.3% to 82% in 2022, 35% of waste was recyclable in grey containers in the second audit and this was 42.2% in 2019 (Figure 2.5). A negligible amount of waste was collected in the green containers.

2.3.3.2 Administrative unit

Potentially recyclable materials were reduced by 36.8% to 19.1% due to lower organic waste generation. Meanwhile, the rate of recyclable waste increased from 47.3% to 71.5% due to the increase of glass waste amount (Figure 2.4). Purity of the waste in blue container is decreased to 76% in 2022. The average purity of green containers was 91%, which is offering a promising performance about other unit's source separation and public awareness for the future (Figure 2.5).

2.3.3.3 Dormitory unit

The dormitory unit tends to have similar results with administrative unit in 2022 except potentially recyclable wastes in grey containers (Figure 2.4). Potential recyclable materials increased from 16.4% to 49.5% due to higher organic waste generation. Non-recyclable waste ratio declined due to reductions in other waste in 2022 (Figure 2.4). The non-recyclable waste in academic and dormitory units was higher than the administrative unit because of paper towels and miscellaneous materials. Recycling performance has been improved in this unit. The blue container had 41.1% and 86% recyclables in 2019 and 2022, respectively. Grey containers had 42% recyclable content in 2019 and have been reduced to 33% in 2022 after adopting ZW activities on campus. Green containers have 94% purity which has the highest ratio within the second audit (Figure 2.5).

2.3.3.4 Cafeteria unit

Organic (raw) waste (74.9% to 64.4%) declined and potential recyclables ratio reduced 88.7% to 79.2% between first and second audits (Figure 2.4). Grey containers have 97% potential recyclables in both 2019 and 2022 (Figure 2.5). Blue containers have high purity due to the well-trained kitchen staff in the dining-hall. A negligible quantity of waste has been collected from green containers.

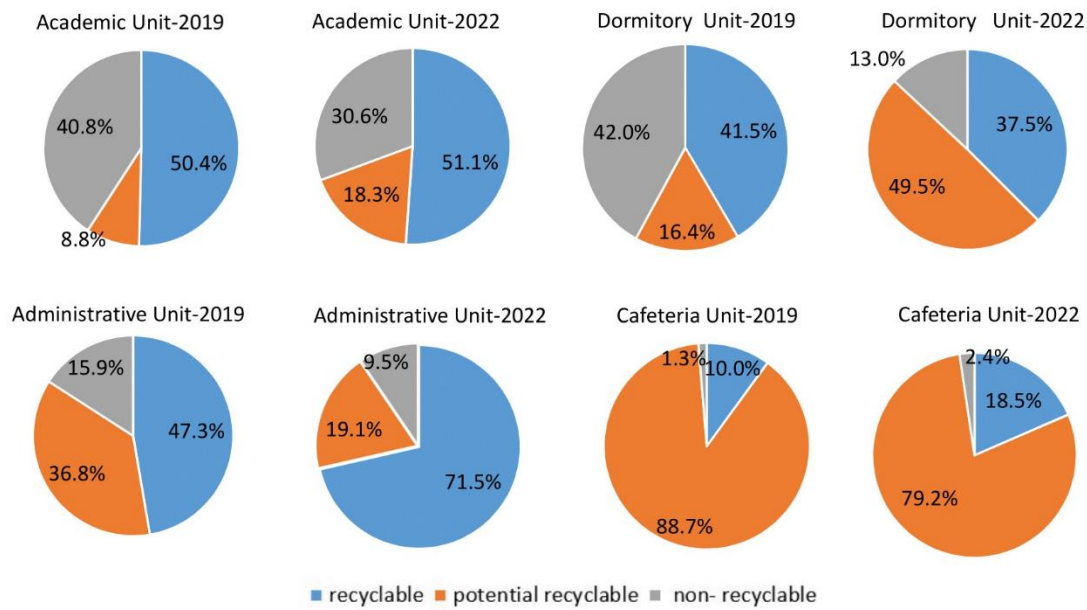
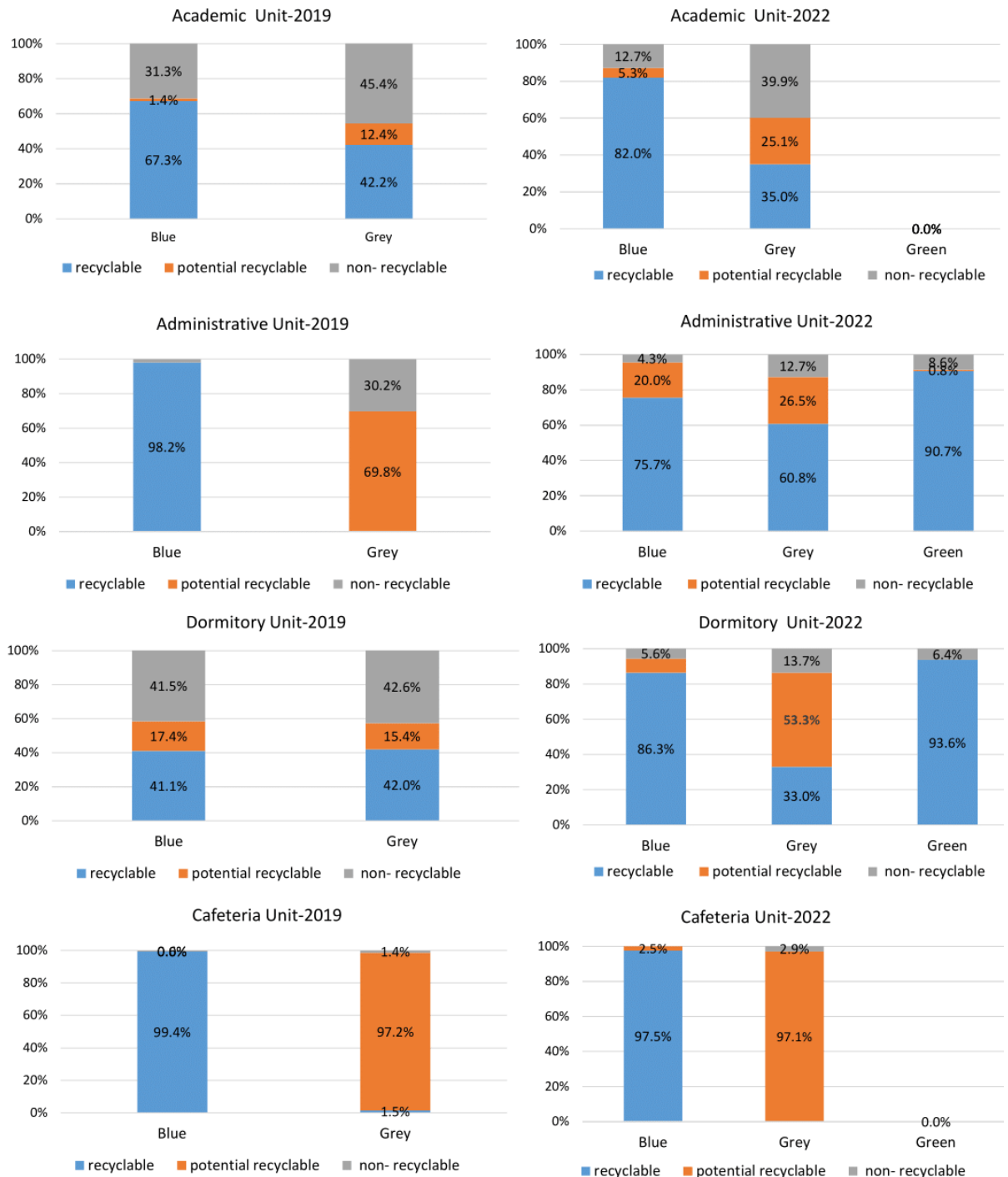


Figure 2.4 : Recycling potentials of produced wastes in 2019 and 2022

In conclusion, since there was no significant difference between the waste composition of the blue and grey containers in 2019, it is possible to say that the university's waste disposal policy was not adhered to by staff and students. A study conducted by Mbama (2022) also found high impurities with in the containers.



* Glass containers in academic and cafeteria units were assumed to be negligible because the waste in these containers was below scale accuracy.

Figure 2.5 : Waste recycling potential and impurities in containers in 2019 and 2022.

Calculations based on Equations (3) and (4) showed that the actual diversion of the Ayazağa Campus from landfill increased from 16.3% to 21.3% and the potential diversion increased from 82.6% to 85.1% in 2019 and 2022, respectively. Consequently, the additional diversion potential from landfills was 63.8% (difference

between potential and actual) if impurities in containers were avoided and potential recyclables were managed properly. Previous study in University of Lagos (Nigeria) showed only 1% of waste was diverted from landfills in spite of its recycling policy and infrastructure due to a lack of commitment to the recycling policy (Mbama et. al., 2022). There were also findings from past study that promote recycling and increase landfill diversion by 62.6 (Bashir et. al., 2020).

Waste characterization and their recycling potential could change in different regions (Tchobanoglous et. al., 1996) due to geographical and cultural differences (Mbama et. al., 2022). Previous WM studies conducted at universities (Gallardo et. al., 2016; Yalvaç et. al., 2014; Irbaş et. al., 2012) gave all values regarding waste composition; however, some have not grouped them in accordance with their recycling potential. Therefore, the authors of this study completed the calculations of potentially recyclable wastes from previous studies and the results were presented in Table 2.3. Sampling periods range from 1 week to 1 year. Time constraints, financial resources (Smyth et. al., 2010) accessibility of the study area (campus), number of representative samples and other local conditions have an effect on the study period. The results of this study are in the range of the literature. Recyclable waste is mainly found in academic and administrative units of ITU, as in other universities.

The rate of potentially recyclable materials in the literature is usually high because of organic waste (Gallardo et. al., 2016; Taghizadeh et. al., 2012). This is especially evident in the study by Gebreeyessus et al. (2019), where the amount of organic waste led to 93% of potentially recyclable waste. Results of this study (ITU) are consistent with literature by potential recyclables. The highest percentage belongs to cafeteria due to organic waste. In the case of non-recyclable waste, some studies have shown that its level is less than 1 % (Gebreeyessus et. al., 2019; Yalvaç et. al., 2014). Depending on the campus study unit, large-scale findings can be obtained.

Similar to the amount of waste, a relationship could not be established between a country's GDP and recyclable waste ratio of university campuses. For instance, while recyclable waste has a high rate of municipal waste in developed countries, according to Table 2.3, Turkish universities tend produce more recyclable waste. before taking action on WM.

Table 2.3 : Recycling potential of previous university studies

University/Country	Recyclable (w/w)%	Potential Recyclable (w/w)%	Non-Recyclable (w/w)%	Sampling Duration	Source
University of Baja California/ Mexico	32.0	34.0	34.0	12 days	Armijo De Vega et al. (2008)
University of Northern British Columbia/Canada	49.3	21.6	28.4	10 days ^a	Smyth et al.(2010)
University of Tabriz/Iran	59.8	28.8	11.4	1 year ^b	Taghizadeh et al.(2012)- Academic
University of Tabriz/Iran	58.4	34.2	7.4	1 year ^b	Taghizadeh et al.(2012)- Administrative
University of Tabriz/Iran	21.0	76.1	2.9	1 year ^b	Taghizadeh et al.(2012)- Dormitory
University of Tabriz/Iran	28.0	70.2	1.8	1 year ^b	Taghizadeh et al.(2012)- Cafeteria
The Universitat Jaume /Spain	17.9	66.7	15.5	1 year ^c	Gallardo et al. (2016) Canteen
The Universitat Jaume /Spain	37.1	17.1	45.9	1 year ^c	Gallardo et al.(2016)Classrooms, offices,laboratories (cleaning service)
University of Lagos/ Nigeria	33.3	41.7	25.0	11 months ^d	Adeniran et al. (2017)
University of Lagos/ Nigeria	55	35	10	1 month	Mbama et. al. (2022)- Administrative
University of Lagos/ Nigeria	35	45	20	1 month	Mbama et. al. (2022) - Commercial
University of Lagos/ Nigeria	51	36	13	1 month	Mbama et. al. (2022)- Dormitory
Islamic University of Indonesia/ Indonesia	10.3	28.4	61.3	8 days	Kasam et al. (2018)
Kotebe Metropolitan University/Ethiopia	6.4	93.4	0.3	12 days	Gebreeyessus et al. (2019)
Erciyes University/ Turkiye	73.0	-	-	1year ^b	Irbaş et al. (2012) ^e
Mersin University/ Turkiye	33.7	64.1	2.15	3 months ^f	Yalvaç et al. (2014)- Cafeteria
Mersin University/ Turkiye	96.6	1.1	2.3	3 months ^f	Yalvaç et al. (2014)- Academic
Mersin University/ Turkiye	99.4	-	0.6	3 months ^f	Yalvaç et al. (2014)- Library
Anadolu University/ Turkiye	58.0	42.0	-	1 year	Banar and Özkan (2005)
Gazi University/ Turkiye	83.2	12.8	4	1 month ^f	Işildar and Öztürk (2015) Academic
Gazi University/ Turkiye	34.4	54.4	11.3	1 month ^f	Işildar and Öztürk (2015) Cafeteria
Gazi University/ Turkiye	80.5	13.5	6	1 month ^f	Işildar and Öztürk (2015) Administrative
Artvin Çoruh University/ Turkiye	18.2	43.9	37.9	5 days	Demirarslan et. al (2017)
Bogaziçi University/ Turkiye	80.0	5.0	15.0	1 month ^g	Okutan (2017) Academic

^a 5 days for each semester.

^b 1-week sample from each month.

^c 7 days in April, 17 days in May and 15 days in October.

^d Sampling was carried out thrice every month.

^e Irbaş et al. (2012) give their recyclable ratios however they don't give any additional information about the remaining 27 %.

^f 1-day sample from each month.

^g 1 sample from each week.

Even when universities in the same country, as in Türkiye (Yalvaç et. al., 2017; Özverel, 2014) are compared, a broad range of results have been achieved on campus, unlike municipal solid waste results (Hoornweg and Bhada-Tata, 2012). This situation demonstrates the importance of each university carrying out its own study before taking action on waste management.

2.3.4 Suggestions towards to zero waste goal on the campus

The ZW goes beyond maximizing recycling and focuses on the waste hierarchy, targeting to recover all resources (Cole et al., 2016). The study results show that significant percentage of recyclable and potentially recyclable waste were produced in campus. Also, there is support from campus residents to attend recycling activities. Despite the fact that many studies have been conducted on Ayazağa Campus related to, much remains to be done. Throughout the study, it became clear that targeting specific waste categories and strategies contributes to achieve the ZW goal on campus.

Public participation and awareness are one of the most important parts of sustainable ZW management (Zaman, 2015; Dilkes-Hoffman et. al., 2019; Bahçelioglu et. al., 2020). Academia (universities, high schools etc.) have an advantage over other institutions since they have great opportunities to pass the information on waste prevention and waste reduction practices to the younger generations. Almost all ZW activities in public and private institutions in Türkiye continue with a top-down approach. Bahçelioglu et al. (2020) states that only 33% of students consider themselves primarily responsible at WM in their university. Therefore, a survey was conducted to understand public attitudes of ITU Ayazağa residents. The survey was attended by 1994 respondents and according to results 90% participants are willing to contribute in new ZW management activities even if they bring about a change in their daily habits, such as walking longer distances by throwing waste into containers (Maçin et. al., 2020). Subsequently, there is support from the public, but in order to implement this support in practice, education and awareness must continue in the campus.

Waste reduction is another significant step of the ZW approach. This step reduces the adverse impact on the environment by reducing waste generation. The university can have an influence on producers by supporting the green purchasing concept (Leal Filho et. al., 2019). Purchasing products from greener companies will trigger manufacturers

to embrace sustainability. It will also help to create collaborative consumption behavior in the university (Zaman, 2017). ITU needs to extend outsourced services and the supply chain policies for green purchasing which has a minimum impact on the environment. In addition, responsible consumption and production patterns are specified in Goal 12 of the Sustainable Development Goals by 2030. Legal grounds in Türkiye require the cheapest product to be selected when making purchases at universities, which leaves environmental concerns behind. ITU could prioritize local sources to make green purchases less costly in terms of environmental burden, money and time.

Reusing of the products is maximize the use of materials. For instance; placing water dispensers in buildings with water purification system especially in dining-hall can cause a reduction in plastic usage. Previous surveys in Saudi Arabia showed that 84% of the participants are willing to use tap water with a water purification system instead of plastic bottles (Saleem et al., 2019). Moreover, thermos (coffee mug) distribution during student registration can help to decrease single-use of plastics (Taghizadeh et. al., 2012). The ITU dinning hall throws away leftovers and plastic cups to the same container and approximately 320.000 plastic cups are diverted to landfills annually. Rough estimates indicate that almost 1 million pet bottles are used annually in a faculty of Ayazağa Campus. According to the survey (Maçın et. al., 2020) 68% of residents in the ITU will drink tap water if water fountains will be placed in the buildings. While 20% are hesitant to drink from water fountains and 11% will not drink at all (Maçın et. al., 2020). Therefore, it is necessary to set up a water purification system to reduce plastic bottles. The ITU's official store sells these products, but incentivizing or free distribution may be preferable.

Source separation is one of the most critical stages for recycling. Previous study in the campus clarified that, residents need more recycling points and 87% of them believe that ZW labels and posters will increase the success of ZW management (Maçın et. al., 2020). Therefore, almost 4000 new containers with labels were placed to increase source separation. Positive results were particularly obtained in academic and residential units in the second audit. Besides, there would have to be drop-off centres for the remaining potential recyclables (e-waste, textile etc). In addition, recycling of plastic bottles can turn into profit (Armijo de Vega et. al., 2008). Previous study

conducted for ITU Ayazağa Campus show that 21,908 \$/year profit was possible with packaging waste recycling (Maçın et. al., 2022). The deposit system may provide a further incentive for recycling.

Cardboard-paper is a significant waste component in ITU similar to the other universities. An electronic communication system called “Papyrus” was developed by ITU for internal correspondence between university units, used to reduce paper use. Almost 185,000 pages used in Papyrus instead of printed papers in 2019. Electronic submission of homework and student projects are another solution for waste prevention and reduction. The E-learning system called Ninova is used by ITU students for project assignments; however, it should be extended to all courses. Double-sided printing on campuses is another successful strategy to reduce paper consumption (Smyth et. al., 2020).

Establishing policies to create clean waste management streams (e.g., anaerobic digestion, composting) is a significant step in ZW (Zero Waste International Alliance, 2022) Organic waste is a major subcategory of campus waste in ITU similar to previous university studies (Taghizadeh et. al., 2012; Yalvaç et. al., 2014; Gebreeyessus et. al., 2019). Reducing organic waste on university campuses is attainable with awareness campaigns. In a previous study at Rhodes University (South Africa), clarified that students’ knowledge of how much food wasted in the dining-halls would have an influence in changing their behavior (Painter et. al., 2016). Composting and anaerobic digestion are the most preferred organic waste management options on university campuses for using recycling and recovery potential. Several universities in the United States already used anaerobic digestion for organic WM. For instance, Duke University plans to use biogas as an alternative source of fossil fuels as part of meeting the climate neutrality target in 2024. Currently they do not have a biogas facility but they are guiding and supporting nearby biogas facilities in return they are sending organic waste to these facilities (Duke University Office of Sustainability, 2016) Michigan State University installed a biogas facility in 2013 with a capacity of 17,000 tonne/year for co-digestion of farm waste and food waste (Michigan State University, 2013), the 20% of the produced electricity is used by the facility itself and the remaining 80% is used on campus. Adeniran et. al. (2017) suggests that University of Lagos (Nigeria) may need to consider the possibility of

managing its food waste through a synergy with present sewage system to optimize methane gas generation for energy generation. Loyola University diverted 200 tonne/year (almost 20% of their total waste) organic waste from landfill with campus collective program (Loyola University, 2020), in Texas Dallas University, totally 230 tonne waste has been composted since 2012 (University of Texas at Dallas, 2019) and 630 tonne/year was waste sent to local composting facility by Brown University in 2022 (Brown University, 2023). The universities in Türkiye have started on campus compost activities especially after Zero Waste Regulation. For instance, Bahcelioglu et. al. (2020), study showed that the organic waste obtained from METU campus (organic food waste and yard waste) were suitable for composting. Niğde Ömer Halis University, Pamukkale University and Yıldız Technical University have attempted to apply in-vessel composting process on campus (PAU, 2019; YTU, 2019) and Ondokuz Mayıs University (OMU, 2019) has started windrow composting. However, they have not officially shared their study's results yet. In addition to the composting there are some feasibility studies for anaerobic digestion. ITU already has a biogas plant in Ayazağa Campus called Sustainable Energy Production Facility (SEPF) which has a capacity of 1.6 tons/day. The biogas plant could produce 30 kWh of energy per day. Electricity and heat from SEPF is expected to be used on campus. SEPF will accept the waste from the dining hall and other cafeterias. However, the plant is in preliminary operation and is not being used to full capacity. Therefore, organic wastes were considered potentially recyclable in Table 2.1 since they were landfilled during this study.

The management of residues such as inert waste should be disposed of and controlled without putting the environment or human health at risk. Non-recyclable wastes such as toilet paper/paper towel and sanitary wastes are also produced in the campus. These wastes could be reduced by using hand dryers. However, environmental impacts should be assessed using the life-cycle impact assessment prior to taking measures. Systems and policies that promote waste or threaten the environment and human health are unsustainable for the ZW's goal (Zero Waste International Alliance, 2022). In addition to the new WM plan, regular monitoring and reporting is crucial as well.

2.3.5 Limitations of the study and future recommendations

The researchers of this study have considerable control over the field contrary to some (Smyth et. al., 2010) of the previous studies. Although a detailed assessment has been conducted, there are still some limitations in this study. Hazardous waste has been excluded, and this missing data may prevent a holistic perspective of WM on campus. Another parameter is the sampling period. While some studies were conducted for only one semester (Armijo de Vega et. al., 2008; Demirarslan et. al., 2017), others were conducted for the fall and spring terms (Gallardo et. al., 2016; Adeniran et. al., 2017; Smyth et. al., 2020). However, no significant difference between fall and spring semesters in the waste composition was found in some studies (Smyth et. al., 2010; Gallardo et. al., 2016). In any case, it can be suggested to conduct waste characterization studies in different seasons to observe possible fluctuations. Also, the surrounding buildings may mislead the results, especially in the first field study. Because the cleaning staff brought blue, grey, and green containers separately to the sorting area only in the second field study. According to Painter et al. (2016), external effects are inevitable during waste management field studies. The field work in this study should be extended to other areas of the campus in order to get a better insight. Because this study does not reflect all of the Ayazağa campus in terms of waste quantity, characterization and recycling potential. Additional field studies will be conducted in the scope of ZW management at Ayazağa campus to determine the average waste generation.

This study would provide a basis for future WM studies at the university. Examination of parameters like the C/N ratio and pH in the future are significant for managing organic waste (Lin et. al., 2018). The material flow analysis of the WM scenarios is critical for calculating the actual diversion of landfills. The ZW is difficult to achieve in the absence of clear management policies and requires long-term initiatives (Cole et. al., 2016). Inadequate implementation of policies results in low levels of recovery. The results of this study will assist in the development of more strategies to reduce the presence of non- targeted wastes in the containers (Mbama et. al., 2022). The new ZW management strategy sometimes appears with economic challenges due to the need for initial investment (Veleva et. al., 2017). The economic consequences of new WM strategy investments and green purchasing activities should be examined. Besides

all this, Covid-19 effects will most likely fade away in the near future. Therefore, it is preferable to perform a third audit to see the impact of ZW activities on landfill diversion without a pandemic.

2.4 Conclusion

Waste characterization and recycling potential studies are crucial for understanding WM performance and planning new strategies. Based on the results, the waste characterization is similar to 2019 audit, but the ratio of recyclable waste increased in 2022 after implementing ZW practices in Ayazağa Campus. Furthermore, the second field study in 2022 showed that there was a reduction in waste amount. These results revealed that recyclable and potentially recyclable wastes were still sent to landfill. The waste diversion from landfills could be increased by waste separation at the source, especially for potential recyclables. The priority of the ZW is the prevention and reduction of waste and only the application of recycling does not provide a suitable framework for a ZW or resource strategy in terms of absolute material reduction. Students and staff should be continually educated about the significance of ZW management. An efficient and sustainable ZW management goal can be part of campus life by considering all environmental, economic and social aspects.



3. AN MFA-LCA FRAMEWORK FOR GOAL-ORIENTED WASTE MANAGEMENT STUDIES: “ZERO WASTE TO LANDFILL” STRATEGIES FOR INSTITUTIONS

3.1 Introduction

The traditional linear waste management (WM) model “take, make, use, dispose” is potentially reaching its limits and the availability of raw materials and energy cannot be guaranteed any more (BSI, 2017). Recently "circular gap" reports have started to appear that investigate the overall circularity in society and how closely we have closed our material consumption. The results have found that we are far from closing the gap due to increasing consumption. For instance the circularity for the Netherlands have been found to be 24% (Colloricchio, 2023) and the circularity of Denmark and Sweden as low as 4% and 3% (Keys et al., 2023). In response, Zero Waste (ZW) has emerged as an ambitious goal over the past decade. ZW entails a systematic approach to designings and managing products and processes, aiming to prevent waste generation, responsibly dispose of materials, and maximize resource recovery from waste streams (Zaman and Lehmann 2011). However, there remains a lack of consensus on the exact definition and concepts associated with ZW (Zaman and Lehmann 2011). The transition to ZW heavily relies on the participation of business and institutional communities, with an increasing number of companies setting waste reduction targets such as "zero waste to landfill" (Veleva et al, 2017). “Institutions” in this context encompass universities, government bodies, hospitals, and administrative divisions of medium and large corporations. Additionally, the term “campus” refers to the specific area within universities, hospitals or other institutions, as defined within the scope of this study.

A subgroup of ZW research that have come the furthers, is ZW to landfill, where there are successful projects in the literature, particularly in cities in China (Zhao et al., 2021; Ma et al., 2023; Yang et. al., 2023) and Europe namely; Netherlands, Sweden, Denmark and Belgium (Zaman and Lehmann 2011), medium-large size companies (Song et al., 2015; Veleva et al,

2017; Anuardo et al., 2022) and universities in the USA (Anuardo et al., 2022). However, achieving complex goals like ZW requires detailed planning, robust steps, and stakeholder cooperation (Moqbel, 2018). Comprehensive WM planning should not only consider a single quantitative or qualitative measure at the campus, but also cover the entire life cycle of waste after it leaves the campus (Turner et al., 2016), as moving up in the waste hierarchy does not mean that it mitigates environmental impact fully. While campus-specific studies have gained attention in recent years, they have focused on identifying quantitative (de Vega et al., 2008; Smyth et al., 2010) or qualitative (Yuan et al., 2013) measures, rather than comprehensive data or a framework for deciding on a WM strategy. Quantitative studies have been conducted on waste characterization, recycling potential (de Vega et al., 2008; Smyth et al., 2010; Adeniran et al., 2017), and ZW index (Kasam et al., 2018). Some studies are more comprehensive (de Vega et al., 2008; Smyth et al., 2010; Moqbel, 2018), while others are more focused on characterization results (Gebreeyessus et al., 2019). Although, framework studies on ZW management exist in the literature, they are often related to construction and demolition waste (Lu et al., 2021), ZW manufacturing (Kerdlap et al., 2019), waste minimization (Song et al., 2015), or ZW society (Zaman, 2017).

Material flow analysis (MFA) has been increasingly used to identify waste flow from generation until the end of life, particularly for goal-oriented WM strategies (Stanisavljevic & Brunner, 2014; Wang et al., 2020; Kasavan et al., 2021). Although MFA has been used in recent years to determine waste flows, it is unable to determine environmental impacts (Allesch & Brunner, 2017; González-García & Dias, 2019). Life cycle assessment (LCA) helps to overcome this limitation and provides a clear understanding of environmental benefits and burdens (Thushari et al., 2020). Therefore, MFA followed by the LCA methodology has been used to provide fundamental information about the environmental performance of a WM system (Turner et al., 2016). While previous MFA-LCA studies have considered waste as a main actor, they were generally conducted at an urban (Turner et al., 2016; Istrate et al., 2021) or country scale (Wäger et al., 2011; Schmidt et al., 2020). The MFA-LCA methodology has been proposed as a significant tool for institutions and campus-wide studies aiming to achieve the goal of circular economy (CE) and sustainability (Pauliuk, 2018). We are still a long way from implementing widely accepted approaches to CE evaluation. Proposing a framework at this stage will not solve all the problems. However, it will contribute to a better evaluation and

comprehension of CE at the institution (organizational) level. (Droege et. al., 2021). CE practices are being supported by various methods at different levels and scales (Mendoza, et al, 2019a). Achieving sustainable CE approach and WM requires research, stakeholder engagement, accurate data, and detailed planning (Zaccariello et al., 2015; Papargyropoulou et al., 2016), which can be challenging to implement at a large scale. However, studies at smaller scales, such as hospitals (Koncagül et al., 2022) companies (Veleva et al, 2017), university campuses; can serve as testbeds for WM systems and sustainability projects, offering insights for scaling up (Gu et al., 2019; Stephan et al., 2020). Due to an overall lack of awareness, inadequate policies, economic limitations and lack of decision-support frameworks, the CE concept has not yet been fully incorporated into current institution operations on campuses (Mendoza, et al, 2019b).

To date, campus scale MFA-LCA studies have primarily been conducted to determine the environmental impacts of materials and energy flows (Lopes Silva et al., 2014; Stephan et al., 2020). Although there are some preliminary examples (Graham Sustainability Institute, 2011) to determine the environmental effects of WM for institutions and their campuses, there is still no clear approach for a framework guiding how to use MFA and LCA methods. The aim of this study is to provide a comprehensive framework for institutions to develop new WM strategies that consider the entire life cycle of the WM system and its environmental impacts. The research involved five main steps: (i) defining the objective, (ii) defining the framework, (iii) presenting the framework's results through a case study, (iv) discussing the framework's findings, and (v) drawing conclusions. While the MFA-LCA method has been utilized previously for different aspects of WM (Le Hesran et al., 2020), this framework proposes a stepwise approach that emphasizes the significance of environmental impacts of various WM scenarios, based on qualitative and quantitative measures. Therefore, this study addresses several research gaps, including (i) a lack of a systematic MFA-LCA approach to guide institutions in developing a goal-oriented WM strategy, (ii) the feasibility of achieving ZW to landfill goal at institutions,

and (iii) whether landfill diversion is always the best option in terms of several environmental impact categories and circularity

3.2 Methods

This section presents a framework for assessing the environmental impacts of goal oriented WM system. While previous MFA-LCA studies have relied primarily on quantitative data, incorporating qualitative data is crucial for effective WM planning. Therefore, this framework employs both types of data.

3.2.1 MFA-LCA framework for the institutions

The framework comprises five main modules, namely goal and scope, inventory collection (including MFA), impact assessment, data quality, and interpretation which are addressing key steps in LCA. It consists of eight steps, which are described in detail in Figure 3.1. The suggested framework is based on ISO 14040-44 standards (ISO, 2006a), which provide a systematic approach to LCA. Moreover, the data quality assessment (DQA) part of the framework is inspired by (JRC, 2011) and (Henriksen et al., 2021). The proposed framework is developed in response to previous challenges faced by authors and other researchers in the field

(Perlman, 2020). It aims to improve the existing MFA-LCA studies by integrating both quantitative and qualitative data for WM scenario planning.

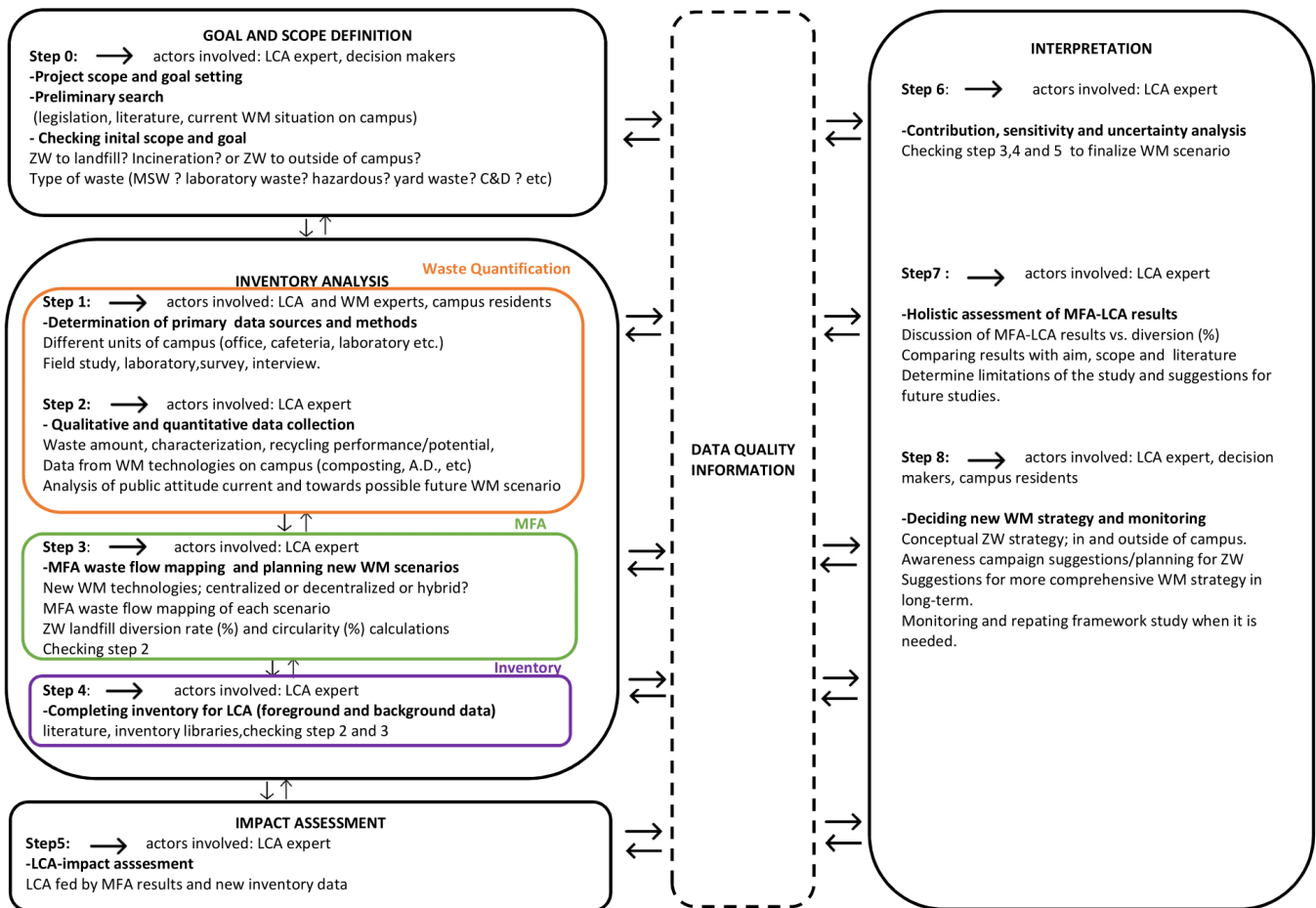


Figure 3.1 : MFA-LCA waste management framework for institutions. The four larger solid boxes follow the general ISO 14040 phases. To this has been added a Data quality box, and a number of steps to guide the user through each phase.

3.2.2 Framework steps

The proposed framework is based on the ISO14040 standard (ISO, 2006a; 2006b), which consists of four main steps: goal and scope, inventory, impact assessment, and interpretation. In the framework, Step 0 represents the goal and scope step, which involves a detailed

examination of the current WM structure before deciding on the goal and scope. In this step, waste types such as municipal, laboratory, yard, and C&D are investigated under the scope.

Step 1 to Step 4 cover the inventory analysis step in the ISO standard. Step 1 involves the determination of waste sources, including waste amount from different units, such as cafeteria, faculties, garden, etc., to identify hotspots of the campus, while waste composition is significant to determine the heterogeneity of waste flows in the LCA system. This step has a significant effect on the decision-making process by revealing where the majority of waste is coming from, and adjustments in Step 0 may be needed.

Step 2 describes the details of the data collection process. Data collection in the framework involves both qualitative and quantitative measures. Primary data collection through quantitative studies, such as waste characterization or operation of decentralized facilities at the campus, allows for quantitative data collection, while qualitative studies, such as questionnaires to campus residents, provide valuable information with respect to understanding public attitudes toward managing ZW and predicting future WM scenarios. In addition, interviews with internal and external stakeholders will help to understand the WM structure at the university. Quantitative and qualitative data collection might be carried out simultaneously, and prioritizing quantitative studies will help to decide details of qualitative data collection, especially in terms of public behavior on source separation.

After collecting primary data from the campus in Step 2, missing data for mapping waste flows from generation till the disposal will be provided from local waste authorities, municipality, and private WM companies. In Step 3, MFA will be used to map waste flow of current and future WM scenarios. Also, the need for new WM facilities in and/or outside of campus will be determined in this step. Landfill diversion rate of each scenario will be calculated to compare LCA results and diversion rates in the following steps.

Missing data of foreground and background systems will be completed in Step 4 from literature and life cycle inventory (LCI) libraries such as Ecoinvent. DQA is done after collecting the

whole inventory for LCA. Even though the data collection process is over in Step 4, data quality influences each step as it is seen in Figure 3.1.

Step 5 is impact assessment step as described the ISO standard. Environmental impact of selected categories is calculated by using LCA software (e.g.; EASETECH, Open LCA, GaBi, SimaPro etc.) or excel. Steps 6 to 8 cover the interpretation of the results according to the ISO standard. Step 6 involves contribution, sensitivity, and uncertainty analysis, with sensitivity analysis following the method by Clavreul et al. (2012) and DQA effects being analyzed using the method by Andreasi Bassi et al. (2017). Step 7 involves a holistic assessment of the LCA results, including a discussion of the results versus the diversion rate, limitations, and future suggestions. Finally, Step 8 involves the decision-making process for a conceptual ZW strategy, considering social aspects, economic possibilities, and legal basis. Monitoring of the new WM strategy is recommended to follow the WM performance and determine when the framework study should be repeated.

Before starting the study, the actors involved in each step should be determined. The LCA expert represents the main researcher in the study and can also be a WM expert, in addition the WM expert represents people from the municipality and private WM companies. Campus residents represent students, academic and administrative staff in the campus, and decision makers represent the university management. LCA experts are involved in all steps of the framework, while campus residents have a significant role in the qualitative data collection step. The decision makers have a role in the determination of the goal and scope and in finalizing the framework study. The framework does not have strict instructions for additional questionnaires or open discussions with stakeholders, but they may be conducted if needed.

3.3 Case study

In this section, we will discuss the results of the framework-case study. The study focused on ZW management at Ayazağa Campus of Istanbul Technical University (İTU) in Türkiye, which was chosen due to data accessibility.

3.3.1 STEP 0: Goal and scope of the case study

The current legislation (which the aim of ZW to landfill) and WM situation were evaluated, and preliminary unstructured interviews were conducted to understand the WM and data handling structures at the campus. The geographic scope of the study was the Ayazağa campus in Sarıyer-Istanbul, and the study boundaries included all WM activities, from waste generation to disposal, as well as reprocessing and utilization activities of recycled and recovered materials. The functional unit (FU) of the study was all municipal waste, including yard wastes from the campus, which amounted to 1990 tonne/year in 2019. Hazardous waste, laboratory waste and C&D waste were excluded from the FU.

3.3.2 STEP 1: Determination of primary data sources and methods

The quantitative method included a field study, which involved waste characterization, impurities in containers, recycling potential, and operation of the decentralized WM facility (anaerobic digester) in the campus. The qualitative methods included semi-structured interviews with WM experts and an online questionnaire to campus residents. The campus was divided into four main units: academic, administrative, cafeteria, and residential. Yard waste from gardens and waste containers on campus streets for municipal solid waste were also included in the study.

3.3.3 STEP 2: Data collection

Field studies were conducted to collect data for the ZW goal, which was chosen in Step 1. Questionnaires were used to gauge public attitudes towards ZW, while semi-structured interviews were conducted with cleaning staff on campus and experts from private companies contracted by the municipality to understand the current WM system. Additionally, literature sources were consulted to complete the inventory.

3.3.4 Waste management in the campus before the ZW

At Ayazağa Campus, single stream recyclables were collected separately in blue bins, while residuals were stored in grey bins. The recyclables were sent directly to the Material Recycling Facility (MRF), while the others were sent to the transfer station (TS). From the TS, waste was transported to the Kısırmandıra Facility (KF) and/or Seymen Landfill. The KF is an integrated

WM facility that includes MRF, composting, and Refuse Derived Fuel (RDF) plant units. The mass balance of KF units was determined based on a 12-month observation study (Sezer & Arıkan, 2011).

3.3.5 Waste amount, composition, and impurities in the source

Waste production data was collected from the field study conducted on campus, taking into consideration impurities in the blue and grey containers. In addition to campus waste, glass waste collected from the streets via drop-off method was learned through personal interviews with the waste collection company. The annual yard waste amount was obtained from the Directorate of Construction and Technical Works of the university.

3.3.6 WM inside and outside of the campus

The data for waste collection, transport, Material Recycling Facility (MRF), market substitution of recyclables, composting, use on land, incineration, and landfill were obtained mainly from the municipality and literature sources, while data for the anaerobic digester (AD) was provided by the campus. However, due to Covid-19 restrictions on campus, only preliminary data was available for the AD plant. It's also important to note that the incineration plant in Istanbul was still in the preliminary stages at the time of the study.

3.3.7 STEP 3: MFA waste flow mapping and planning new WM scenarios

MFA consists of three main elements: system, process, and flow (Brunner and Rechberger, 2004). The MFA shows flows within and between processes in a defined system. STAN 2.6 was used as the MFA software (Cencic, 2016), and the output calculations were based on the mass-conversion principle. The landfill and use on land processes were considered as stocks of the system, while materials entering (Input) and leaving (Export) the system for further processes outside the system were represented as (I) and (E) (Figure 3.2).

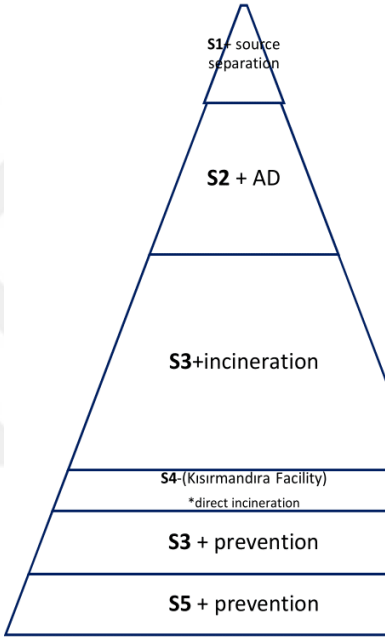
Landfill Diversion in the case study was defined as the percentage of waste generated that is not disposed of in a landfill. Circularity was determined by the proportion of recycled and recovered material in the WM system relative to the total waste amount. For packaging waste,

circularity was calculated based on the substitution of materials from MRFs. In the case of biodegradable waste, there were two destinations: AD and composting.

The field study data and questionnaire results were used while creating the current and future scenario for the MFA study. The waste compositions were categorized as others, glass, packaging -paper-cardboard, and food waste, based on the waste collection, transport, and disposal methods in the campus. The waste from the campus was sent to either the Seymen Landfill or KF. The WM scenarios were divided into two main parts, before and after ZW management. The waste impurities in the recyclable and residual waste bins were included in the calculations.

Currently, the diversion rate of campus waste from the Seymen Landfill to the KF is uncertain. To understand the effects of direct landfilling, four different scenarios were created. The waste sent to Seymen Landfill was set as 75%, 50%, 25%, and 0% of grey bins in the a, b, c, d scenarios, respectively, while the remainder was sent to the KF. The change in mis-sorted (impurities) waste fraction after the ZW awareness campaign was calculated using (Maçın et al., 2020) questionnaire results. The future scenarios were scenario 2, 3, 4, 5, 6 and 7. The effects of AD on the campus were included in scenarios 3, 4, 5, 6 and 7. The prevention of PET bottle, plastic bag, other plastic, paper, and cardboard waste components were included in Scenario 6 and 7 based on questionnaire results. Scenarios 4, 5 and 7 were designed to assess the incineration facility's effects on landfill diversion. Details of WM scenarios were explained in Table 3.1

Table 3.1: WM scenarios of the case study

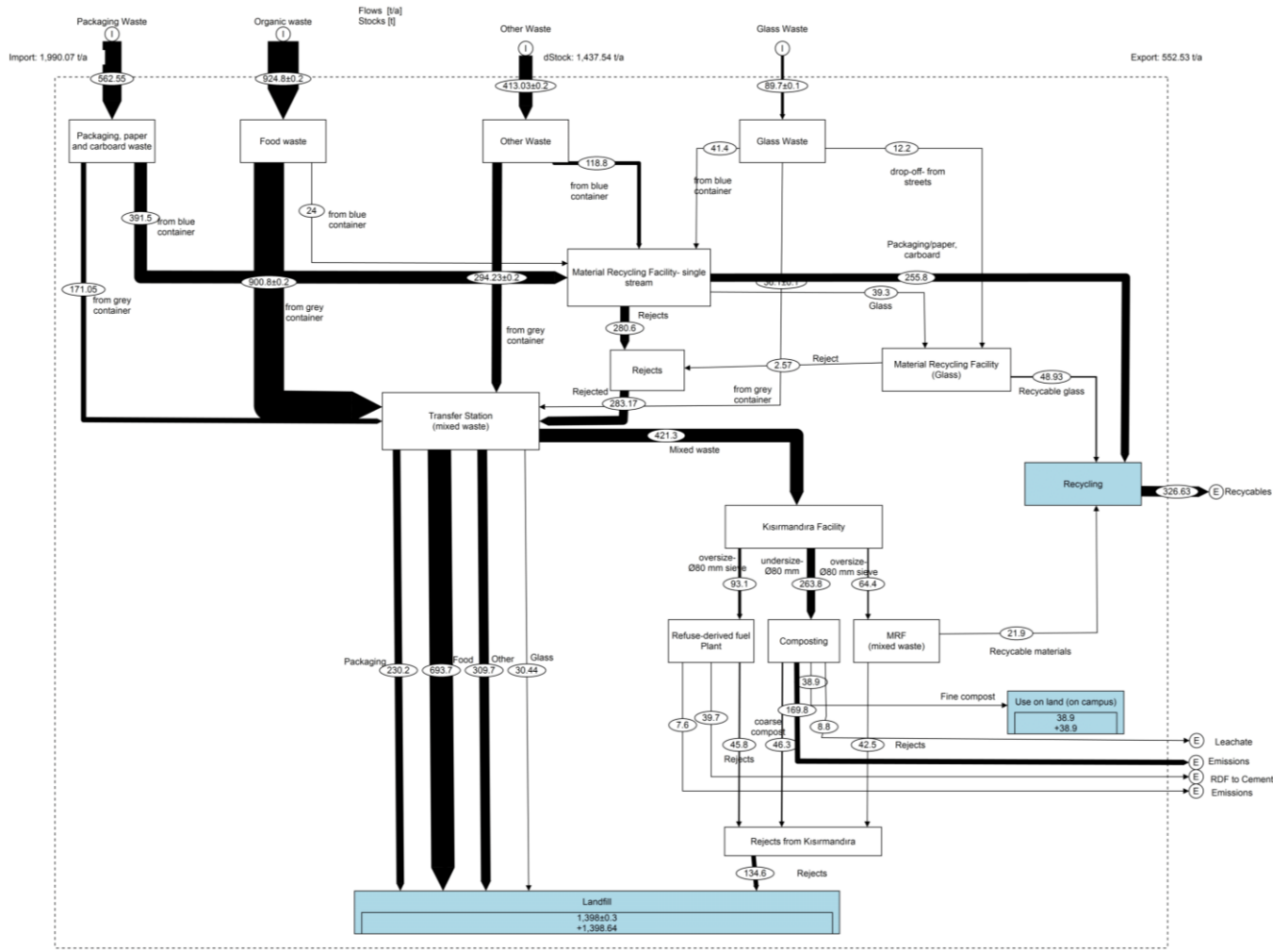
Before Zero Waste Management		
Scenario	Description	Key aspects and changes
S1-a	The scenario includes mis-sorted waste (impurities) in the blue and grey bins. Blue containers and drop-off glass waste sent to MRFs while grey containers sent to transfer station (TS). After TS, 25% of the grey bins sent to KF while remain sent to landfill	The landfill diversion rate after TS is still not certain. KF and amount of waste sent to this facility will effect overall landfill diversion.
S1-b	Source separation same with S1-a, but 50% of the grey bins sent to KF while remain sent to landfill	
S1-c	Source separation same with S1-a, but 75% of the grey bins sent to KF while remain sent to landfill	
S1-d	Source separation same with S1-a, but 100% of the grey bins sent to KF.	
Zero Waste Management		
Scenario	Description	Key aspects and changes
S2	Same as S1-d, but mis-sorted waste fractions in the bins were decreased according to questionnaire results.	
S3-a	Source separation same with S2, Food waste from the dining-hall and cafeteria units sent to AD in the campus. 25% of the grey bins sent to KF while remainder is sent to landfill.	
S3-b	Same as S3-a, but 50% of the grey bins is sent to KF while remainder is sent to landfill	
S3-c	Same as with S3-a, but 75 % of the grey bins sent to KF while remainder is sent to landfill.	
S3-d	Same as S3-a, but 100% of the grey bins sent to KF while remainder is sent to landfill	
S4-a	Same as S3-d, but 100% of the grey bins sent to KF while all rejected streams is sent to incineration	
S4-b	Same as S3-d, but all of the waste in grey bins and rejected streams from other facilities is sent to incineration. (KF was neglected in the calculations)	
S4-c	Same as S1-d, but mis-sorted waste fractions in the bins were decreased according to questionnaire results.	
S4-d	Source separation same with S2, Food waste from the dining-hall and cafeteria units sent to AD in the campus. 25% of the grey bins sent to KF while remainder is sent to landfill.	
S5	Same as S4-a, but 100% of the grey bins is sent to incineration.	
S6***	Same as S3-a but pet bottle, plastic bag, other plastic, paper and cardboard waste reduction were also included.	
S7***	Same as 5, but 100% but pet bottle, plastic bag, other plastic, paper and cardboard waste reduction were also included.	

* MRF= Material recovery facility, TS= Transfer station, WTE= waste-to-energy

** KF (Kısırmandıra facility); mixed MRF, mixed waste composting facility

*** We exempted from the zero burden assumption, in cases where we included waste prevention, but it was modelled simplistic here as just the materials in the products. (see appendix Ecoinvent data for waste prevention).

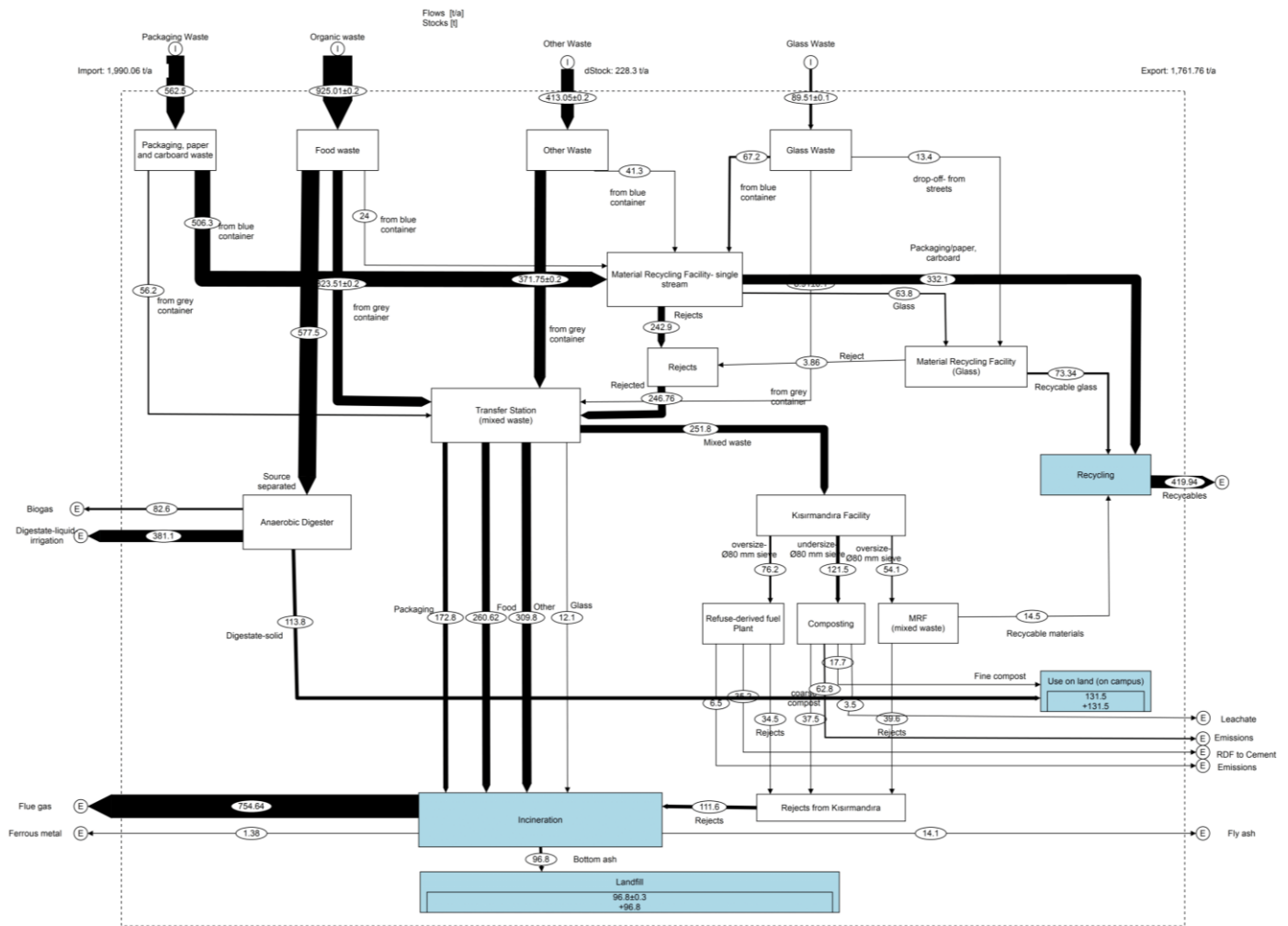
The MFA examples before and after ZW strategy of the case study (scenario-1a and scenario-4a) are illustrated in Figure 3.2 and Figure 3.3 while information about the results of other scenarios can be found in the Appendix.



*footnote: Landfill Diversion: $100 - [\text{Stock (Landfill)} \div \text{Flow (Import)}] \times 100$ (ISIE-SEM, 2021).

Figure 3.2 : MFA illustration of the case study before ZW improvements – scenario-1a

The waste flow and system boundary details are illustrated in Appendix. The case study's system boundary includes the substitution of virgin material with secondary materials from MRFs, the use of compost and digestate on land to substitute chemical fertilizer, and the use of RDF product in a cement factory to substitute petroleum coke. The study also calculated flue gas emissions from waste incineration, environmental savings from energy recovery as electricity and heat, and the disposal of fly ash and bottom ash from the incineration plant. The WM scenarios were based on questionnaire results, current, and under construction facilities in Istanbul, while studies using the framework for other universities also can include hypothetical scenarios.



*footnote: Landfill Diversion: $100 - [\text{Stock (Landfill)} \div \text{Flow (Import)}] \times 100$ (ISIE-SEM, 2021).

Figure 3.3 : MFA illustration of the case study after ZW improvements - scenario-4a

3.3.8 STEP 4: Background data-completing inventory

The missing data in the background systems were obtained from Ecoinvent 3.7.1 (Ecoinvent, 2020), and the electricity grid of Istanbul was assumed to be the same as Türkiye's electricity mix. in Ecoinvent (Ecoinvent, 2020; EPDK, 2020).

Table 3.2 : Summary of the WM facilities inventory for the case study

MRF	Range	Value	Source
Market substitution factor (plastic)	0.67-0.90	0.85	(ELPCA, 2012; Fernández-Braña et al., 2020)
Market substitution factor (paper)	0.47-0.90	0.85	(ELPCA, 2012; Fernández-Braña et al., 2020)
Market substitution factor (cardboard)	1	1	(ELPCA, 2012)
Market substitution factor (glass)	0.83-0.84	0.84	(ELPCA, 2012; Fernández-Braña et al., 2020)
Market substitution factor (Non-Fe metals)	0.65-100	0.85	(ELPCA, 2012; Fernández-Braña et al., 2020)
Market substitution factor (Fe metals)	0.51-1	0.7	(Tagliaferri et al., 2016)
Market substitution factor (Aluminium)	1	1	(ELPCA, 2012)
Market substitution factor (PET)	0.81	0.81	(ELPCA, 2012)
Electricity consumption-mixed waste (kWh/t)	10.0-14.3	14.3	(Kurdoğlu, 2013; Yıldız-Geyhan, 2016)
Electricity consumption-source separated(kWh/t)	10.0-14.3	10.0	(Kurdoğlu, 2013; Yıldız-Geyhan, 2016)
RDF			
Electricity consumption (kWh/t)	40-79	40	(Burnley et al., 2011; Montejó et al. 2013; ;Turner et al., 2016; Guven et al., 2019; Abeliotis et al., 2020)
Diesel consumption- trommel screen (80 mm) (L/t)	2.31	2.31	(Grzesik & Malinowski, 2016)
Transfer Station			
Electricity consumption (kWh/t waste)	-	4	(Turner et al., 2016)
Diesel consumption (L/t waste)	0.42-1	0.42	(Turner et al., 2016; Istrate et al., 2021)
Collection and Transportation Trucks			
Diesel consumption (L/t)-collection	5.69	5.69	(Yaman, 2011)
Diesel consumption (L/t-km)-transportation	0.018	0.018	(Yaman, 2011; Sezer, 2019)
Anaerobic Digestion			
Leaking gas (%)	0.4-14.9	2	(Scheutz and Fredenslund, 2019)
Electricity consumption (kWh/tonne waste)	49-101	49	(Boldrin et al., 2011; Ardolino et al., 2018; Guven et al., 2019)
Methane content	50-70	65	(Akça et al, 2020)
Wheel loader diesel consumption (L /tonne)	1	1	(Lipasto, 2012)
Composting			
Electricity consumption (kWh/tonne waste)	53	53	(Boldrin et al., 2011)
Wheel loader diesel consumption (L /tonne)	1	1	(Lipasto, 2012)
Leachate			
Leachate treatment (kWh/kg COD)	1.5	1.5	(Turkish water institute, 2019)
Landfill			
Electricity consumption (kWh/tonne waste)	5-8	5	(Manfredi et.al., 2009)
Diesel consumption (kg /tonne)	1-3	1	(Manfredi et.al., 2009)
Energy recovery – CH ₄ in landfill gas (%)	45-60	55	(Keskin,2021)
Incineration			
Water consumption (kg/tonne waste)	397	397	(ELPCA, ,2012)
Energy recovery – electricity (%)	-	25	(IMM, 2012)
Energy recovery – hheat (%)		69	(IMM, 2012)

The marginal electric mix was calculated using values for fossil and renewable electricity sources from literature instead of directly taking the Turkish mix in Ecoinvent (Ecoinvent, 2020; EPDK, 2020). The forecasting of changes in the mixture from fossil fuel to renewables for sensitivity analysis was calculated using SHURA 2020. A summary of the MFA and LCA inventory of WM facilities is given in Table 3.2.

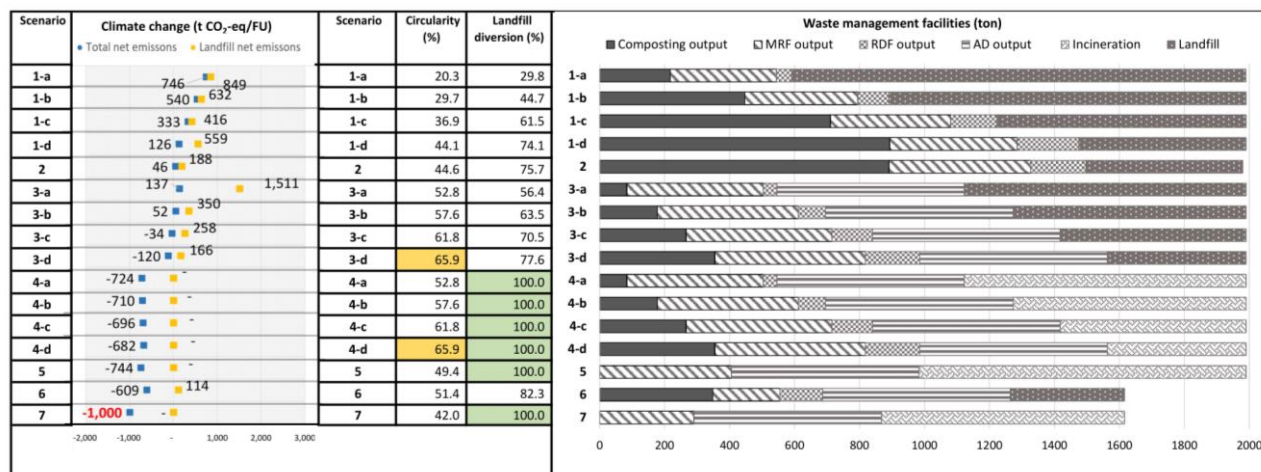
The DQA step was based on the method by Weidema and Wesnæs (1996) and followed the Henriksen et al. (2021) framework. The DQA level was classified according to EC (JRC, 2011). DQA was conducted not only for the foreground data, but also for the electricity and heat background data. While high-quality data is typically expected in campus-scale studies, the DQA results showed that most of the data had basic quality due to the lack of a data recording system at the university and the unclear data sources for WM facilities outside the campus. The inventory mostly consisted of basic quality data, followed by high-quality and estimation data. The results were explained in detail in Appendix.

3.3.9 STEP 5: Life Cycle Impact Assessment (LCIA)

The LCA study was conducted using the EASETECH software, which was specifically developed for solving complex WM problems with heterogeneous flows (Clavreul et al., 2014), following the ISO 14040 (ISO, 2006a) and ISO 14044 standards (ISO, 2006b). System expansion was used to handle multi-functional processes, and the zero-burden assumption (Ekvall et al. 2007) was made for the system boundary, which means that the environmental burdens of producing the materials/products which becomes waste is not included. The LCA was carried out as an attributional LCA, and the Ecoinvent 3.7.1 (allocation at the point of substitution) database was used for the background process. A 100-year time period was considered to calculate the long-term effects of the WM system, while the 20-year effects were included in Appendix for sensitivity analysis.

The ILCD Handbook recommended method was used for the LCA (JRC, 2011; Hauschild et al., 2013), as detailed in Appendix. In addition to the LCA, the circularity of different scenarios was calculated to examine the relationship between landfill diversion, circularity, and environmental impacts. The results showed that landfill diversion did not always improve

circularity. The climate change impact, circularity and landfill diversion results are illustrated in Figure 3.4, while the results for other impact categories are provided in Appendix.



*footnote: Waste management facilities (ton) were based on output of facilities to prevent double counting. Real substitution of packaging, cardboard, paper, food, and yard waste were calculated by considering impurities. WM facility performance, substitution ratios and market conditions were considered during circularity calculations. Maximum circularity was showed in yellow, and landfill diversion was shown in green.** Landfill diversion (%) represents direct landfill.

Figure 3.4 : Circularity (%) vs landfill diversion (%) and Climate change emissions (kg CO₂-eq/FU)

3.3.10 STEP 6: Contribution, sensitivity, and uncertainty analysis

Climate change impact

Contribution analysis results revealed that the majority of emissions were coming from landfilling, while savings were observed from material recycling. The global warming impact of the scenarios varied between 746 kgCO₂-eq (scenario 1-a) and -1000 kgCO₂-eq (scenario 7) FU waste. Surprisingly, composting had an environmental burden instead of saving, which was due to mixed waste acceptance from the transfer station and lower global warming savings from fertilizer substitution compared to the electricity consumption in the composting plant. The impact of composting was evident when comparing scenario 4-a and scenario 4-d. In terms of incineration plant, energy recovery decrease with waste diversion for recycling due to the changing waste input characteristic. The AD facility had net savings due to its acceptance of source separated food waste and additional savings from biogas utilization, which were greater than the release from the AD facility. Furthermore, comparing scenario 1-d vs. scenario 2 and scenario 3-d vs. scenario 6 demonstrated the importance of source separation and waste prevention, respectively. In scenario 7 incineration and MRF savings were less than scenario

4a-4d due packaging waste prevention (Figure 3.5.). Even though scenarios 6 and 7 have the same amount of packaging waste prevention, scenario 7 has more net savings due to incineration savings. Comparing Scenarios 4 and 6 demonstrates the significance of determining “which” waste is prevented. The higher amount packaging waste going to the single MRF facility in Scenario 4 leads to higher net carbon saving (Figure. 3.5.) and circularity (Figure.3.4) values. However, the advantages provided by the substitution of fossil resources will change in the medium term (see Figure 3.6.).

An uncertainty analysis was performed by using Monte Carlo simulation with 10,000 random samplings to evaluate especially the relevant parameters leading to the net saving in the red range in Figure 3.5. The substitution ratio of glass, PET, aluminum, cardboard, paper, and plastic in MRFs, the biochemical methane potential of AD input, electricity, and heat recovery of the incineration plant, and gas collection and CHP efficiencies in the landfill are the most relevant parameters in the climate change category. The results of the uncertainty analysis show high variance, mostly due to limited data about gas collection efficiency. Therefore, the highest uncertainty was observed in scenario 1-a, which has the highest landfilled waste amount.

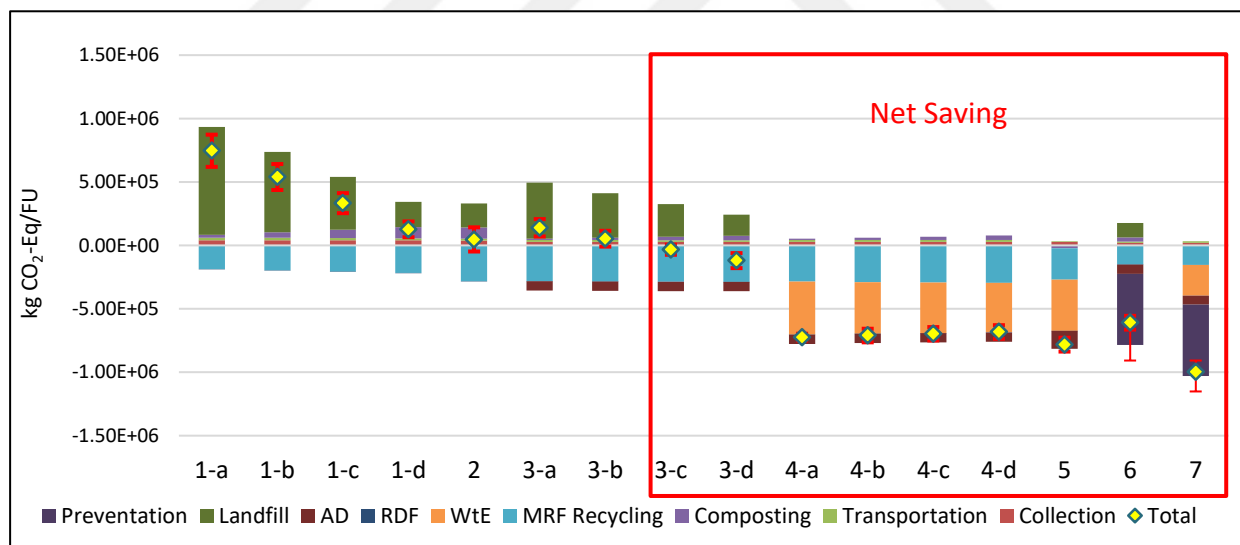


Figure 3.5 : Contribution analysis and uncertainty of net emissions (climate change category)

This section presents sensitivity analyses for two investigations. The first considers changes in the electricity mix by the year 2030, based on a technical report published for Türkiye's

optimum electricity generation capacity mix (SHURA, 2020). This scenario is based on balanced implementation of different policy objectives such as carbon emission reduction, use of domestic resources and cheap electricity. Under the scenario, fossil source (coal and natural gas) use will decrease from 57.7% to 41.7% while the electricity coefficient changes from 0.55 kg CO₂ eq to 0.34 kg eq CO₂ by 2030. The second scenario is based on a change in waste characterization, using the waste characterization of Istanbul (Ak and Sezer, 2020) instead of the campus waste characterization (Figure 3.6.).

The first sensitivity analysis shows that the net landfill burdens are higher in 2030 due to landfill gas substituting less fossil fuels. However, the original scenario has higher net burdens compared to the 2030 electricity mix in some cases, such as scenario 1-c, 1-d, and 2, due to increased MRF savings and reduced CO₂ emissions from electricity consumption during composting in 2030. In contrast, the original scenario has more net savings in scenario 3 and scenario 4 due to energy recovery in AD and incineration. Scenario 6 has more savings compared to scenario 4 and 5, the reason for this is that replacing fossil-based electricity with renewable electricity generation will eventually eliminate the perceived GHG benefits associated with waste-to-energy combustion. This suggests that using WtE as an alternative technology instead of a KF for campus waste landfill diversion has more savings in the climate change category only while the alternative is a fossil based electricity.

The second sensitivity analysis shows that urban waste characterization has different results compared to campus waste. In scenario 6, for instance, plastic bag reduction at the source results in more net savings. Additionally, urban waste characterization scenario has less net burdens in scenario 1-a to 1-d compared to the other two scenarios due to the lower amount of biogenic Carbon in landfilled waste. However, in scenario 5, the urban waste characterization scenario has fewer savings due to the high amount of organic input to the incineration plant.

Other impact categories

The focus of the case study has been exemplified with climate change impacts. But we also assessed the other impact categories in ILCD recommended LCIA method (Hauschild et al., 2013).

Fertilizer application on soil is found to have negative consequences in terms of eutrophication and acidification, with terrestrial acidification generally causing net impacts, except for scenario 1-d that benefits from recycling. Incineration is generally contributing with savings in all scenarios, primarily because it replaces the heavily fossil-fuel-based Turkish electricity mix.

Similarly, the AD process demonstrates savings in most impact categories due to energy recovery, while composting burdens the environment due to the release of ammonia during compost maturation. Marine eutrophication impacts are primarily attributed to composting and AD due to the spreading of the degraded material on farmland, as well as direct impacts from the composting process. While recycling and waste prevention lead to net savings in this category. The freshwater eutrophication impact is similarly a high net impact from AD due to digestate use as fertilizer with runoff and leaching causing impacts. Therefore scenarios 1 and 2 stand out with net savings, as they exclude AD from their practices.

Terrestrial eutrophication burdens are mainly due to collection, transportation, and electricity consumption, with additional contributions from engines in AD and electricity consumption in MRFs. Scenarios 4-5 and 6-7 exhibit net savings thanks to waste prevention and savings in incineration facilities respectively, while scenario 1-3 result in net burdens.

MRF recycling, particularly in the case of plastics (PE), and waste prevention, result in substantial impacts within the human toxicity cancer category, primarily due to the additional transportation involved. Conversely, the ecotoxicity and human toxicity non-cancer categories bear burdens arising from collection and transportation; however, in all scenarios, they exhibit net negative impacts, thanks to the savings realized through electricity substitution in the case of ecotoxicity, and recycling and composting practices in the context of human toxicity non-cancer.

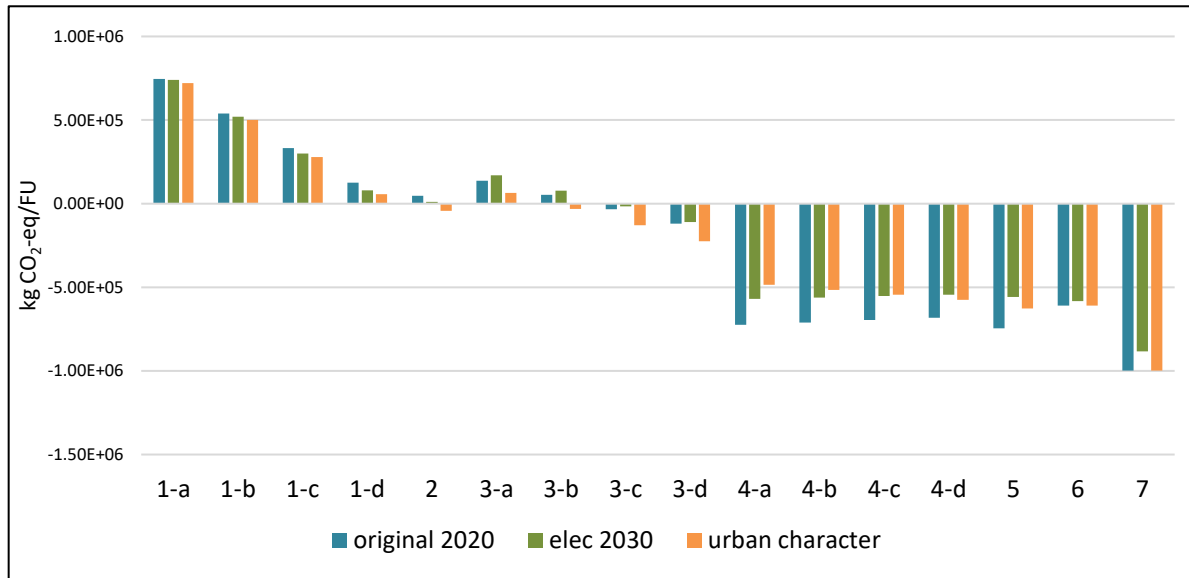


Figure 3.6 : Electricity mix change in 2030 and urban waste characterization sensitivity.

3.3.11 STEP 7: Holistic assessment of MFA-LCA results

3.3.11.1 Comparing results with aim, scope and literature

Based on the results, it is feasible to achieve ZW to landfill on a campus scale by implementing various WM strategies, such as waste prevention (S6-7), source separation (S2-3-4-5-6-7), AD (S3-4-5-6-7), and incineration (S-4-5-7). However, certain types of waste, such as bottom ash from incineration plants or others, may still require disposal in a landfill (Vaverková, 2019).

3.3.11.2 Discussion of MFA-LCA results vs. diversion (%)

The results of the case study indicate that most of the WM data have basic data quality, primarily because of the lack of a data recording system for WM facilities outside of the campus. As a result, much of the data is unclear, derived from secondary sources. The MFA results demonstrate that some of the residuals or ash from the incineration plant will still need to be disposed of in landfill. Figure. 3.4 illustrates that landfill diversion ranges between 29.8% and 100%. While there is an increase in net environmental savings and circularity with diversion rates, there is no direct correlation. Scenario 6 and 7 are less circular by definition since circularity encompasses all types of recycled and down-cycled end-of-life waste. The technology used to substitute landfill has an impact on net impacts and circularity (Andreasi Bassi et al., 2017). Even though landfill diversion increases in scenario 4-a compared to

scenario 3-a, thanks to the incineration plant, the circularity ratio remains the same. Energy recovery occurs, but material recovery is not observed due to the nature of incineration. Slorach (2019) pointed out that circularity does not always mean environmental sustainability. Their study showed that composting was the worst option in terms of LCA. Nonetheless, composting is preferable to incineration and landfill within the waste hierarchy (Slorach et al., 2020). Details of environmental impacts, landfill diversion, and the circularity relationship are significant topics to be addressed in future studies.

3.3.11.3 Determine limitations of the study and suggestions for future studies.

This case study represents the first attempt to apply the framework, but it has some limitations. Firstly, some aspects of the inventory are limited to the preliminary stage for the AD and the design parameters of the incineration plant. Thus, the study could be repeated once both facilities are fully in operation. Secondly, the study relies on data of basic quality due to the lack of institutional data recording and WM structure. Thirdly, the landfill diversion after TS for Ayazağa Campus is still unclear, leading to a wide range of results between sub-scenarios a to d. Therefore, a new WM data recording system should also include transportation and destination data. Furthermore, future WM scenarios in the case study are based on questionnaire results. Other studies could focus on WM strategies that comply with circular economy targets and climate action plans while applying the framework.

3.3.12 STEP 8: Deciding new WM strategy and monitoring

3.3.12.1 Conceptual ZW strategy; inside and outside of campus.

It is suggested that an awareness campaign be planned for ZW, and that engagement with both the municipality and university management is necessary for decision making. It is important to engage in waste mapping with MFA in STEP 3 to comply with initial discussions in STEP 0, in order to avoid potential disagreements about WM options. The combination of top-down (campus management) and bottom-up (students, academics, and non-university staff) approaches to the framework serves to implement WM both inside and outside of the campus (Zen et al., 2016).

3.3.12.2 Suggestions for more comprehensive WM strategy in long-term

ZW is a holistic approach that has relations with various terms such as CE, industrial ecology, and LCA (Zaman, 2015). Universities and other institutions can be seen as “mini cities” due to their size and impact on the environment and society, and therefore environmental, economic, and social aspects should be considered in the planning of ZW management on campus. Reporting annual developments to all stakeholders and conducting motivation and awareness activities through social media are also significant. The ITU’s administration is willing to participate in ZW activities, and their willingness could be supported by other stakeholders (Zen et al., 2016) such as students, academic and administrative staff. The success of waste prevention and reduction activities is dependent on public attitude (Zaman, 2015). Additionally, almost all ZW activities in public and private institutions in Türkiye have been carried out using a top-down approach. Hence, awareness campaigns and a new WM strategy are needed for public participation (Zaman, 2017).

3.3.12.3 Monitoring and repeating framework

This study will be a starting point for future sustainability studies in the university. The ultimate goal is to prevent waste production in the first place. However, if this is not possible, ZW to landfill is possible with the knowledge of waste amount and taking necessary actions to diminish it. WM monitoring should be done regularly using quantitative measures such as waste amount and recycling performance, as well as qualitative studies to follow the effect of new WM on the public and their demands. Regular monitoring will show WM performance and when the framework study should be repeated. The WM monitoring system should include reviewing waste characterization, developing and reporting, implementing sustainability practices, and strengthening WM actor relations (Schoeman et al., 2021).

3.4 Discussion of the framework

3.4.1 Data collection

The data structure for the WM system was developed from scratch using a framework. However, there is no standard method available to characterize flows in universities (Perlman, 2020). Hence, detailed field studies, questionnaires, and interviews were conducted to determine the inputs for the MFA and LCA in the framework. Nevertheless, it

may still be challenging to obtain high data quality assurance results, especially for WM activities outside of the campus. Despite this, the framework can help establish an institutional data structure baseline for institutions.

3.4.2 Tools

A combination of MFA and LCA was chosen for this framework, but other methods were also considered. Input-Output Analysis (IO) requires detailed data and is mostly used for larger scale studies such as country-level analyses (He et al., 2020). Substance Flow Analysis (SFA) has been used in previous studies, particularly for nutrient flows (Vujovic et al., 2020), but MFA is more suitable for primary studies. SFA could be used for further originated campus studies after the main structure of the WM system is decided using the framework. Multi-Criteria Decision Making (MCDM) could also be an alternative (Coban et al., 2018), but it weights the relative importance of criteria based on expert decisions, leading to additional uncertainty and less information in terms of environmental impact categories. MCDM is more appropriate for high data quality studies or could be used after using this framework. Considering study scale, data availability, and quality, MFA and LCA are the most suitable methods for goal-originated campus scale studies and for converting complex waste flows into environmental impacts.

3.4.3 Legal structure

The main goal of the framework is based on regulation, and future WM scenarios are directly affected by the legal structure. The ZW concept has different approaches in the literature. Legal structure is also important to understand the responsibilities of stakeholders in deciding WM strategies. Municipalities play significant roles in WM activities in some countries, while states are responsible for WM in others. The legal structure affects stakeholder engagement and institution management in making final decisions in Step 8.

3.4.4 Advantages

The framework is designed for ZW but is also flexible for other goals. It is based on well-known methods and helps structure a WM data baseline from scratch, even if the institution has no data related to WM. This framework will link bottom-up and top-down approaches in evaluating the environmental impacts of WM systems and address systemic challenges

related to achieving the ZW objective (Iacovidou et al., 2017). The nature of the developed framework, which combines WM policy/regulation and WM strategy in the institutions, will lead to reaching the ZW goal (Al-Aomar et. al, 2022).

3.4.5 Shortfalls

The case study revealed that framework application is possible on a campus scale, but it also has possible shortfalls. The framework is based on a detailed field study, and data collection is time-consuming, especially for data outside of the campus. The awareness of the importance of sustainability as a driver for change is necessary for institutions to implement initiatives for sustainable WM, but there is a lack of standards. However, frameworks may help deal with this obstacle and find a suitable WM strategy for a specific area. The framework is more suitable for a top-down approach than a bottom-up one and requires administrative support. It needs stakeholder engagement for decision making, and public participation has a critical impact on the success.

Existing studies on ZW practices, including the framework suggested in this study, have primarily emphasized "zero waste-to-landfill" target, which primarily involves recycling and waste-to-energy facilities as a means to reduce the amount of waste sent to landfills (Veleva et al, 2017). However, these studies have often overlooked environmentally friendly approaches such as sustainable consumption, waste prevention and re-use. (Agyabeng-Mensah et al., 2021). To achieve a comprehensive ZW practice, it may be necessary to implement a holistic transformation of current production, extraction, consumption, and processing systems (Agyabeng-Mensah et al., 2021). Consequently, ZW should place greater emphasis on guiding individuals in altering their lifestyles to support sustainable natural cycles, wherein waste is regarded as a resource that can be utilized by others (Song et al., 2015).

3.5 Conclusion

This study proposes a MFA-LCA framework to address challenges that may arise during a campus-scale WM study. The framework was developed by reviewing established MFA-LCA-related WM, institutions and campus sustainability studies and assumes that primary data collection will be required since no prior data is available. Several key features of the

framework are highlighted, including the identification and classification of waste resources on campus, the combination of qualitative and quantitative data collection, estimation of potential environmental impacts of different WM strategies, and the provision of a baseline for future data collection and monitoring systems. While the framework has the potential to be applied to other goal-oriented studies beyond ZW, limitations exist, especially with regard to economic aspects. To address this, future studies could focus on integrating life cycle cost considerations into the framework. Moreover, it should be noted that qualitative and quantitative data collection processes are time-consuming when compared to other WM studies. Future case studies from other institutions can validate and improve this methodology.

In addition to introducing the framework, a case study is presented in this paper. The results demonstrate that some residuals or ash from the incineration plant will still be disposed in landfill. In addition, landfill diversion does not always promote circularity or directly address sustainable consumption and public attitudes towards ZW goals. Consequently, new WM strategies should prioritize waste prevention and focus on circularity objectives. Furthermore, a comprehensive approach considers which WM technology is utilized to further reduce WM emissions while advancing the ZW objective.



4. DECARBONISING UNIVERSITY CAMPUSES: A BUSINESS MODEL FOR FOOD WASTE MANAGEMENT AT ISTANBUL TECHNICAL UNIVERSITY (ITU) AYAZAĞA CAMPUS, TURKIYE

4.1 Introduction

Globally, 1.32 billion tonne of food waste (FW) are sent to landfills at a cost to the global economy of over USD 900 billion in wasted resources (Mak et al., 2020) while simultaneously creating 3.3 billion tonne of greenhouse gases (Lin et al., 2022). FW management is a significant environmental problem today, due to depleting natural resources (Zaman, 2014) and the use of land resources (OECD, 2019). Increased focus on FW minimization, aligned with structured waste management (WM) approaches, is a necessity when seeking to solve global problems; however, when it comes to applying this strategy in actuality, too many stakeholders are involved. Pilot studies are thus needed before expanding new strategies to large regional systems (Mak et al., 2020).

University campuses could be pioneer hubs for sustainability projects via the living lab concept (Esfilari et al., 2021). Alternative strategies have been developing in the university campuses to recover FW, in order to close nutrient and carbon loops and replace centralized waste management facilities such as landfill and incineration (Al-Aomar et al., 2021; Wang et al., 2021; Vea et al., 2018) which converts FW into value-added products such as organic fertilizers and energy. A sustainable FW management model goes further than emissions by designing circular flows of resources. There will also be adverse effects due to the array of land guidance for biomass production, including further greenhouse gas emissions due to indirect land use restrictions (Tan and Lamers, 2021). This could cause rebound effect as it changes production and consumption demands (Tan and Lamers, 2021). Closing material loops is not enough to guarantee environmental improvement. Hence, a chosen WM strategy is expected to be delivered in the context of more circular and decarbonized economies (Styles et al., 2022).

Moving towards a circular economy (CE) will require a shift throughout the value chain, including new business models and consumer behaviour (Vea et al., 2018). There is also increasing demand for sustainability and decoupling (OECD, 2019). Life cycle assessment

(LCA) is a well-established tool for understanding environmental impacts. However, there have been discussions suggesting that LCAs neglect economic and socio-technical aspects (Goffetti et al., 2022). A sustainable business model (SBM) could close the gap since it pursues a sustainable economy, it, therefore promotes resource protection and societal and environmental benefits instead of fostering economic growth alone (Ribeiro et al., 2018). Circular business models (CBMs) are an extension of SBMs and comply with the decoupling of economic growth with the consumption of resources (Braun et al., 2021). The most commonly used and accepted business model for WM studies is the "Canvas" business model proposed by Osterwalder and Pigneur (2010). Since, Osterwalder and Pigneur's (2010) canvas model is driven by economic value (Joyce and Paquin, 2016), it neglects environmental and social aspects. The transition to a more circular proposition, therefore, requires the expansion of the traditional Canvas business model. A holistic sustainable business model includes a triple bottom line to reconcile traditional economic objectives with social and environmental concerns (Zaman, 2012). Hence, the Triple Layered Business Model Canvas (TLBMC) was suggested by Joyce and Paquin (2016) to explore sustainability-oriented business models by adding two layers to Osterwalder and Pigneur's (2010) original Canvas model. The TLBMC has been used in several studies, albeit they focused on the sector (Ribeiro et al., 2018) or company (Joyce and Paquin, 2016) level. The latter authors contended that some users found the TLBMC a bit daunting, due to the number of inputs required. Another study, conducted by Amoussouhoui et al. (2022), added two segments to the original Canvas model to evaluate financial viability and sustainability, albeit it prioritized technologies rather than conceptual system design. Finally, the Ecocanvas model, provided by Daou et al. (2020), was created for circular business and covers several areas in addition to the economy, such as legal and social.

As part of climate change mitigation, Türkiye aims to develop sustainable food system practices and create an "Action Plan for the Circular Economy," which also includes the objectives of the LCA (MoEUCC, 2022). However, FW management impacts at campus level can inspire and lead to larger projects, in order to help solve global problems by providing preliminary results faster (Mak et al., 2020). The campus and, in some cases, its surroundings thus become a testing ground where solutions to improve the sustainability of succession and academic operations are sought and tested, and integrated into education (Verhoef et al., 2020). Previous studies at universities in Türkiye have focused on different topics, such as reducing FW at source

(Ozcicek-Dolekoglu and Var, 2019), landfill diversion by composting (Bahçelioğlu et al., 2020), the biogas potential of anaerobic degradation (Alibaş et al., 2016; Yiğit, 2018), biogas for heat production (Coşar et al., 2013) and the co-digestion of FW and wine inoculum in AD (Kasarcı, 2012). Furthermore, a number of FW-related life cycle assessment (LCA) studies exist at campus level (Tunalı M., 2012; Genç, 2021) in Türkiye. For instance, an LCA study relating to FW management, conducted at Boğaziçi University (Tunalı, 2012), found a 7.2% impact reduction of climate change across the whole campus, thanks to using biogas instead of coal to generate electricity. Another LCA was carried out for FW generated in an ITU dining hall (Genç, 2021); however, the study assumed a generic functional unit (FU) of 1,000 food services, instead of the actual amount of FW on campus, thereby making it hard to compare it to other studies. Nonetheless, the study found that AD is the most environmentally friendly solution for organic waste management, in comparison to composting and landfill (Genç, 2021). That said, to the best of our knowledge, no study has created a business model by using LCA results in combination with circularity and the cost of existing facilities in and outside the campus.

The aim of this study is to analyze a FW management model that provides a more circular and decarbonized economy at ITU Ayazağa Campus, Türkiye. The goal is to propose a business model that can easily be used by a wider audience. In this regard, this paper addresses the following questions: (i) what is the ranking of decarbonization, toxicity, acidification, and eutrophication scenarios, and how is that ranking going to change in the future? (ii) What scenarios have the highest circularity? (iii) What are the net costs and CO₂ abatement costs of the scenarios? We then propose a business model by using results from the previous three questions.

4.2 Materials and method

4.2.1 Modelling assumptions

The modelling approach in this study was based on ISO 14040 and ISO 14044 standards (ISO, 2006a & 2006b). CO₂ emissions coefficients from the IPCC 2021 report (IPCC, 2021), for GWP 20 and GWP100 years were used for climate impact calculations, while other impact categories were based upon Hauschild et al. (2013). The EASETECH v 3.4.1 (Clavreul et al., 2014) model was used for calculations, while the background data were obtained from Ecoinvent 3.7.1

(Ecoinvent, 2020). To address multi-functionality, and attributional approach was followed, and system expansion was applied. The functional unit (FU) in this study was the annual source-separated and treated FW (577 tonne/year) from the ITU Ayazağa campus. Since the WM project is applicable in the long term, future conditions in the year 2050 were included in the calculations. The average grid mixes in 2020 and 2050 were calculated, as changes in the electricity mix could affect CO₂ savings due to changes in fossil fuel substitution.

4.2.2 System boundaries and scenarios

In this study the circularity of the FW was defined as the distribution of nutrients (N, P) back into the soil during land application. Four scenarios were created (Figure 4.1) to determine the most circular, environmentally and economically feasible FW management method and business model as follows: an AD facility at the campus, composting, incineration and a landfill facility, which are operated by the municipality (outside of the campus). These scenarios also compared centralized and decentralized WM facilities. (Angouria-Tsorochidou et al., 2021). Composting and incineration facilities are located 26 km away from the campus, while the landfill is 112 km away. Hence, emissions that occur during the transportation of waste were included in the calculations, except for the AD scenario, where it was assumed to be negligible. A zero-burden assumption (Ekvall et al., 2007) was made for environmental impacts resulting from FW production and WM facilities. The AD is operated under mesophilic conditions (37 °C), and biogas produced in the facility supplies the heat required by the digester, whilst electricity produced from the biogas reduces the use of fossil fuels used in the production of electricity (Akça et al. 2020). Methane loss from AD is assumed at 2% (Scheutz and Fredenslund, 2019). The digestate is treated in an artificial lagoon. After treatment, the solid part is used as fertilizer in the campus park-garden areas, and the liquid part is used for fertigation. Thus, less inorganic fertilizer and water consumption will be required. The total energy consumption in the digester is 49 kWh/tonne of waste. While 18.3 kWh of this amount is used by pumps and technical equipment, 30.6 kWh of energy is needed to heat the reactor. The composting unit is designed as a tunnel system with an eight-week active composting cycle (Sezer and Arıkan, 2011). The average moisture content of input material is 74% (Akça et al. 2020) while output changes by between 20 and 28% (Gencsoy, 2010). After composting, the materials are screened with a Ø15 mm trommel screen, and undersized material is used in Istanbul Metropolitan Municipality (IMM) parks, while oversized material is sent to a landfill.

Electricity consumption involved in tunnel composting is 53 kWh/ton (Boldrin et al., 2011). Data on the land application of N and P, in terms of circularity, were based on Gençsoy (2010), Yoshida et al. (2014), and Cobo et al. (2018). The N and P circularity indicators have been defined as the quantity of nutrients that are recycled, applied to the soil and absorbed by the soil to the amount of nutrient present in the collected FW (Cobo et al., 2018). The incineration plant's energy requirements are 25% electricity and 69% heat recovery efficiency according to design (IMM, 2012). In the landfill scenario (which is the current FW management policy for the campus), waste is sent to the regional landfill (Seymen). Waste degradation rates at the landfill for each waste category were taken from De la Cruz and Barlaz (2010). Gas collection and gas engines are installed at the landfill, as well as leachate collection and treatment.

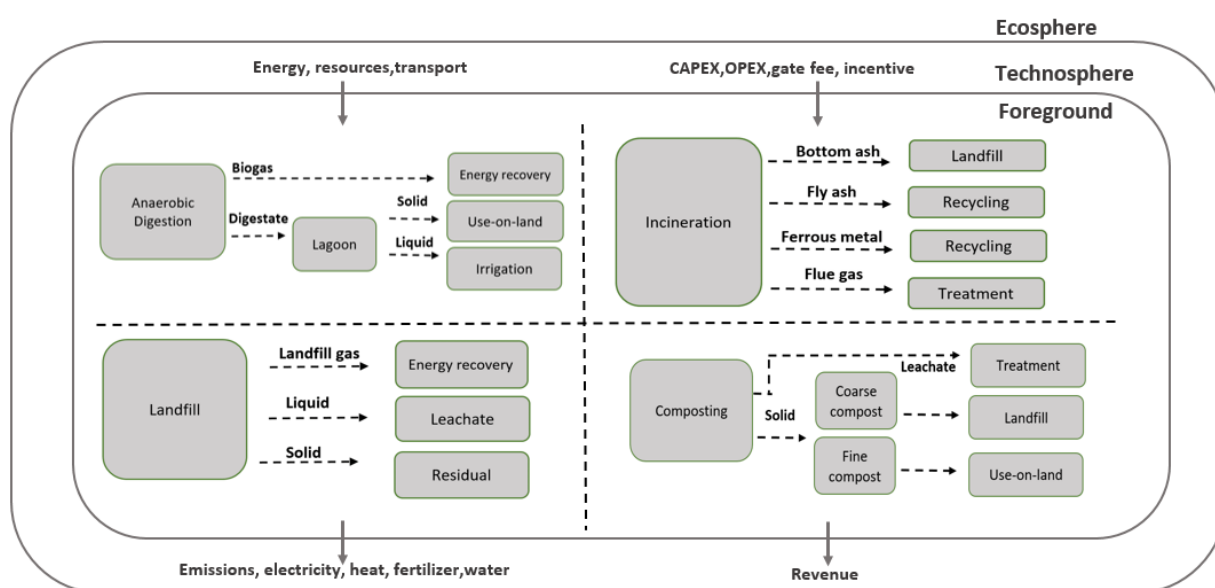


Figure 4.1 : System boundaries of the waste management scenarios

4.2.3 Data inventory

Waste production data were provided by a field study at the campus. Most of the AD data were primary, except for lagoon data and gas leakage inventory, for which literature data were used as a proxy. Operational data for the other three facilities were provided by the literature, and cost data were provided via personal interviews with the municipality, market price investigations, and the literature. A summary of the environmental and economic inventories is given in Table 4.1, whilst a detailed inventory is provided in the Appendix.

Table 4.1 : Summary of the environmental and economic inventories

Anaerobic Digestion	Value	Source
Leaking gas (%)	2	(Scheutz and Fredenslund, 2019)
Electricity consumption (kWh/tonne waste)	49	(Primary data, 2019 ^a ; Boldrin et al., 2011)
Methane content	65	(Primary data, 2019)
Wheel loader diesel consumption (L /tonne)	1	(Lipasto, 2012)
Methane potential (L/kgVS)	370	(Akça et al, 2020)
Composting		
Electricity consumption (kWh/tonne waste)	53	(Boldrin et al., 2011)
Wheel loader diesel consumption (L /tonne)	1	(Lipasto, 2012)
Leachate		
Leachate treatment (kWh/kg COD)	1.5	(Turkish water institute, 2019)
Landfill		
Electricity consumption (kWh/tonne waste)	5	(Manfredi et al., 2009)
Diesel consumption (L /tonne waste)	1	(Manfredi et al., 2009)
Energy recovery – CH ₄ in landfill gas (%)	55	(Keskin, 2021)
Incineration		
Water consumption (kg/tonne waste)	397	(ELPCA, 2012)
Energy recovery – Electricity (%)	25	(IMM ^b , 2012)
Energy recovery – Heat (%)	69	(IMM, 2012)
Energy recovery – Electricity (%) - sensitivity	26	(Bisinella et al., 2021)
Energy recovery – Heat (%) - sensitivity	62	(Bisinella et al., 2021)
Cost Inventory		
Biomass energy-incentive (€/kWh)	3.2	
Thermal waste treatment-incentive (€/kWh)	3.1	(Official Gazette of the Republic of Türkiye, 2021)
Landfill gas-incentive (€/kWh)	1.9	
Minimum wage (€/month)	347	(Official Gazette of the Republic of Türkiye, 2020)
Electricity (€/kwh)	0.12	(Energy Map, 2020)
Natural Gas (€/kwh)	0.04	(IGDAŞ, 2020)
Waste transportation (€/tonne)	9.6	(Primary data, 2020)

^a Primary data: field study was done on Ayazağa campus by the authors for the environmental and personal interviews relating to the cost data. ^b IMM: Istanbul Metropolitan Municipality

4.2.4 Sensitivity scenarios

Sensitivity analyses were conducted for changes such as; biogas upgrading (Faragò et al., 2021) in the anaerobic digester facility (Scheutz and Fredenslund, 2019) and different energy recovery performances at the incineration plant (Bisinella et al., 2021).

4.2.5 Economic calculations

Capital costs, operational costs, and revenues were included in the economic calculations. Inventory data were obtained from the municipality via personal interview and technical reports. The currency of Turkish Lira converted to Euro. A real interest rate of 3% was applied and annualized with a WM facility's lifetime of 30 years (Ozgun et al., 2021). Economic calculations in the case study were evaluated according to a widely established investment valuation method, as in equation (1) (net future value).

$$F = P(1 + i)^n$$

P: present value

F: future value of the given amount of money

i: interest rate per period (i = 3% for this study)

n: year (n = 30 years for this study)

The net cost of each scenario was calculated by subtracting revenues from the sum of investment and operating costs, while the CO₂ abatement cost was calculated by using equation (2) (Faragò et al., 2021). Details of the operational cost items are indicated in the Appendix.

$$CO_2 \text{ abatement cost} = \frac{\text{Net cost (€/year)}}{\text{Avoided emission (ton CO}_2 \text{ avoided/year)}}$$

Currently, Türkiye has a financial incentive to produce energy from waste, in the form of electricity production from biogas, thermal treatment, and landfill gas. However, this incentive is currently only valid until 2025 (Official Gazette of the Republic of Türkiye, 2021), and there is a high chance of policy changes due to economic conditions in the country. A change in the

electricity mix and incentive policy could thereby affect net costs and CO₂ abatement costs. Hence, future cost calculations were considered with and without incentives. In addition, the EU Emissions Trading System (EU ETS) scenario was calculated for the future.

The EU ETS policy addresses climate change and the reduction of greenhouse gas emissions in a cost-effective manner. The ETS is essentially a market-based mechanism for reducing carbon dioxide (CE Delft, 2021). Municipal waste incineration is currently excluded from the EU ETS, but the European Parliament has recently agreed to include municipal incinerators by 2026 (Zero Waste Europe, 2022). The study conducted by CE Delft projects the price of the EU ETS at 90 €/tonne (CE Delft, 2021); hence, this value was used during cost calculations. Even though Türkiye is not a member of the EU, there is a strong relationship between Türkiye and EU countries, so there is a high possibility of applying the ETS in Türkiye in the near future.

4.3 Results and discussion

4.3.1 Environmental impacts

4.3.1.1 Circularity

Circularity was calculated by considering putting nutrients back into fields (and substituting raw materials by using less chemical fertilizer). It was found at its highest in the AD scenario at 42.6%, followed by composting at 36.1%, while for landfill and incineration circularity was zero, as illustrated in Figure 4.2. Details of nutrient (N, P) distribution during land application is shown in the Appendix. While incineration has the greatest carbon savings, there is neither decoupling nor circularity for natural resources.

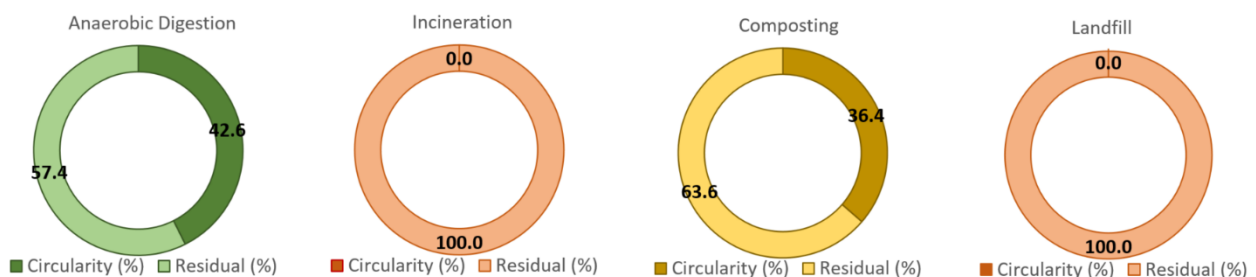


Figure 4.2 : Nutrient circularity (%) of the FW management scenarios

4.3.1.2 Carbon emissions

The net impacts are presented in Figure 4.3. According to the results for 2020, incineration had the most savings (-192 kg CO₂-eq/tonne FW), followed by anaerobic digestion (AD) (-126 kg CO₂-eq/ tonne FW), composting (119 kg CO₂-eq/tonne FW), and the landfill (924 kg CO₂-eq/tonne FW). Figure 4.3 shows the contribution analysis of the scenarios per tonne FW. Most of the burdens were from electricity consumption and input-specific emissions in the incineration scenario, but chemical usage (e.g. sodium hydroxide, ammonia) also has a significant impact (Figure 4.3). Most of the savings were from fossil-based electricity substitution followed by heat substitution. Landfill burdens mostly originated from uncollected landfill gas. For composting, net burdens were observed due to energy consumption in the plant and the different efficiencies of use-on-land applications for chemical and compost fertilizer. The AD scenario produced net savings thanks to electricity substitution, but heat substitution was lower, due to the temperature requirements (mesophilic) of the digester itself. Consequently, landfill remained the worst option, followed by composting, which is a similar outcome to that of Slorach et al. (2019) and Culaba et al. (2023). Net savings were observed for the AD and incineration scenarios; hence, these two options were found preferable for decarbonization of FW management at Ayazağa campus.

In the original scenario, the use of fossil source (coal and natural gas) is expected to decrease from 57.7% to 23.2% by 2050, with a change in electricity coefficient from 0.55 kg CO₂-eq to 0.18 kg CO₂-eq for GWP 100 years. The results indicate that the change in electricity mix has an effect on the overall results, but it does not alter the ranking of scenarios as it illustrated in Figure 4.3. GWP 20 year coefficient results (see Appendix) shows that the impacts of landfill are more dramatic from a shorter time point of view, due to the large releases of methane. Consequently, all decarbonization scenarios (including 20-year and 100-year GWP) were preferable to AD and incineration rather than landfilling (Hodge et al., 2016).

The results of the sensitivity analysis scenario for biogas upgrading showed that, additional savings occurred in 2020 and 2050 for biogas upgrading, but they were negligible since biogas upgrading itself also consumes chemicals (especially amine scrubbing), water, and electricity (Faragò et al., 2021). The results did not show major alterations with the implementation of biogas upgrading, rankings. An additional sensitivity analysis was performed on the change in

energy recovery from the incineration plant. Although heat and electrical recovery is down compared to the original scenario there were no changes in the preferred technologies' rankings. Detailed sensitivity analysis results are given in Appendix.

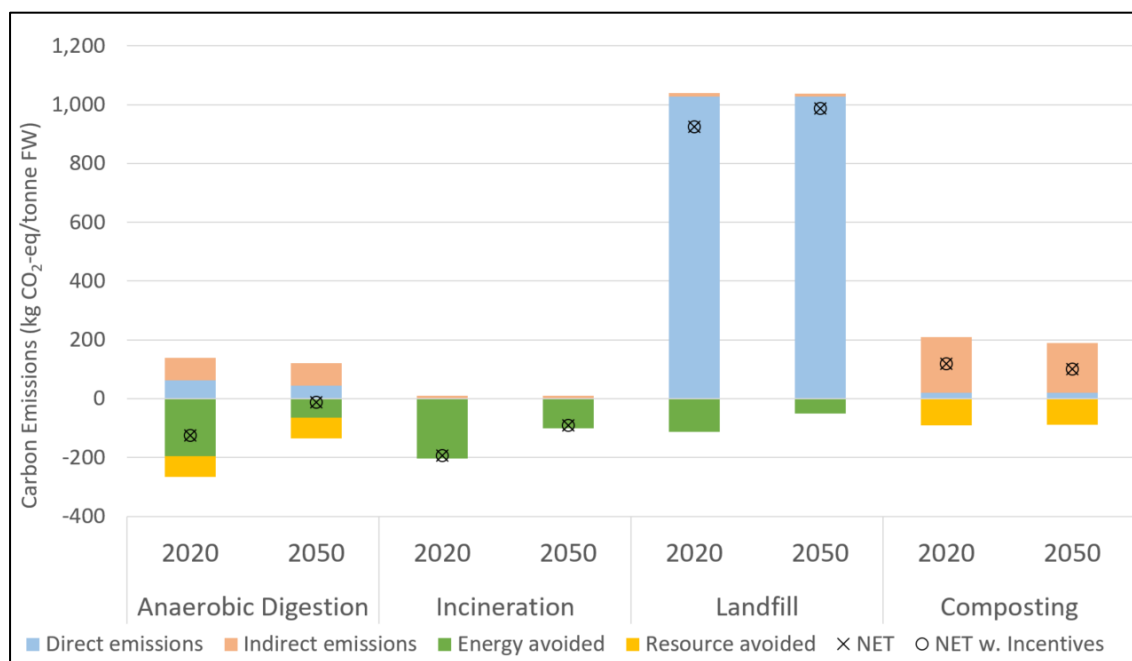


Figure 4.3 : Net carbon emissions of different FW management scenarios (kg CO₂-eq/tonne FW)

4.3.1.3 Other environmental impacts

Fertilizer application on soil has significant burdens in eutrophication and acidification-related impact categories (Hodge et al., 2016). AD and composting have high marine eutrophication due to nitrate emissions into ground water, nitrogen oxide release during combustion processes and high nitrate emissions on digestate application (Figure 4.4).

While, the nutrient circularity (especially N) was very high in the AD which leads to high eutrophication marine effects, incineration had the high freshwater eutrophication because of the phosphates in the fly ash. Meanwhile, credits for mineral fertilizers offset the impact from electricity used for composting, resulting in a net-negative in eutrophication freshwater. AD is the best option, saving, as the credits for electricity and fertilizers are higher than process emissions in terms of eutrophication freshwater. The potential of freshwater eutrophication was the lowest in AD. Furthermore, the eco-toxicity of AD in freshwater was an order of magnitude lower than that of incineration. Terrestrial acidification was high for both composting and AD

due to ammonia released to the air during compost maturation and digestate application, respectively. AD has negative values for human toxicity, while incineration has poor performances in terms of cancerous human toxicity, due to the emissions of bottom ash transport and landfilling. Finally, all impact categories have slightly higher burdens in 2050 compared to 2020 due to the difference in energy mix (Figure 4.4).

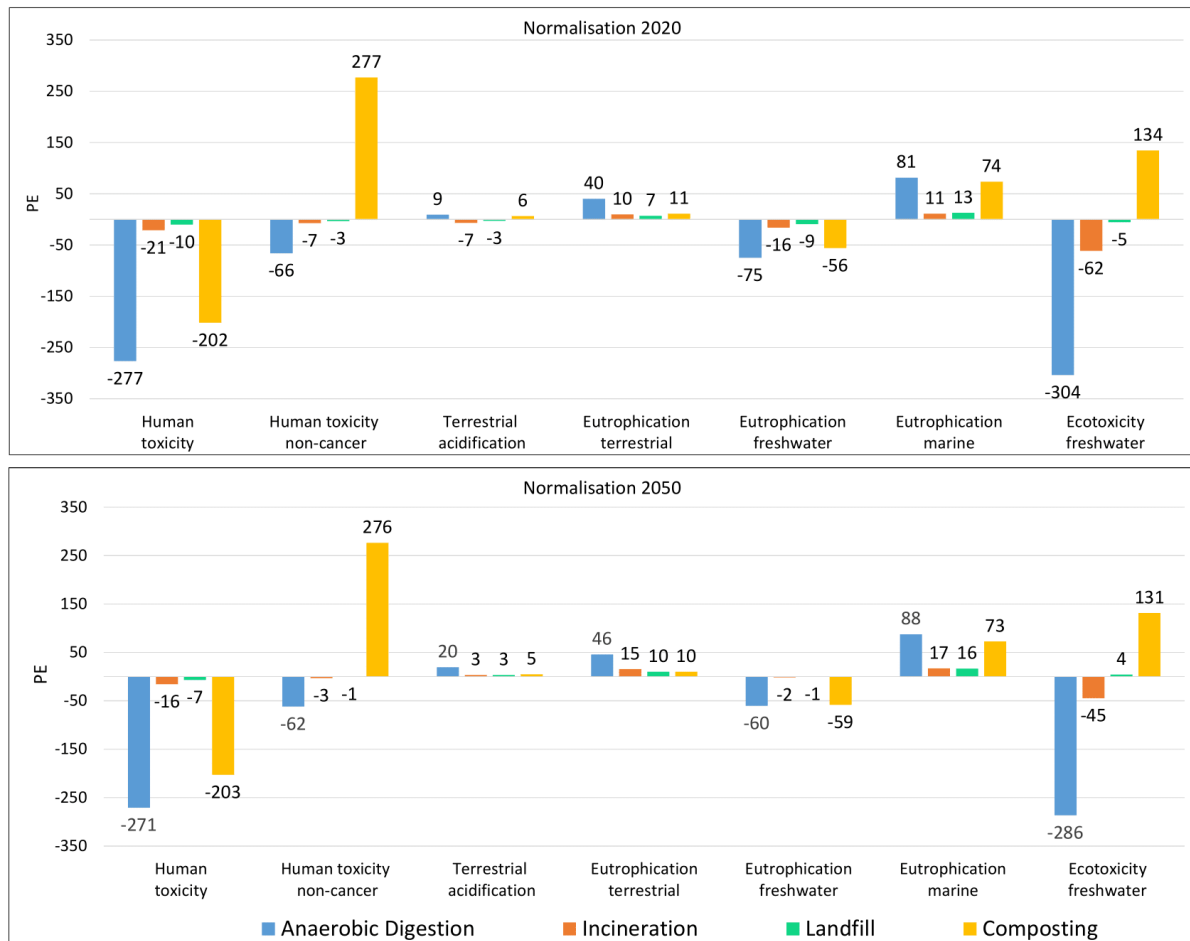
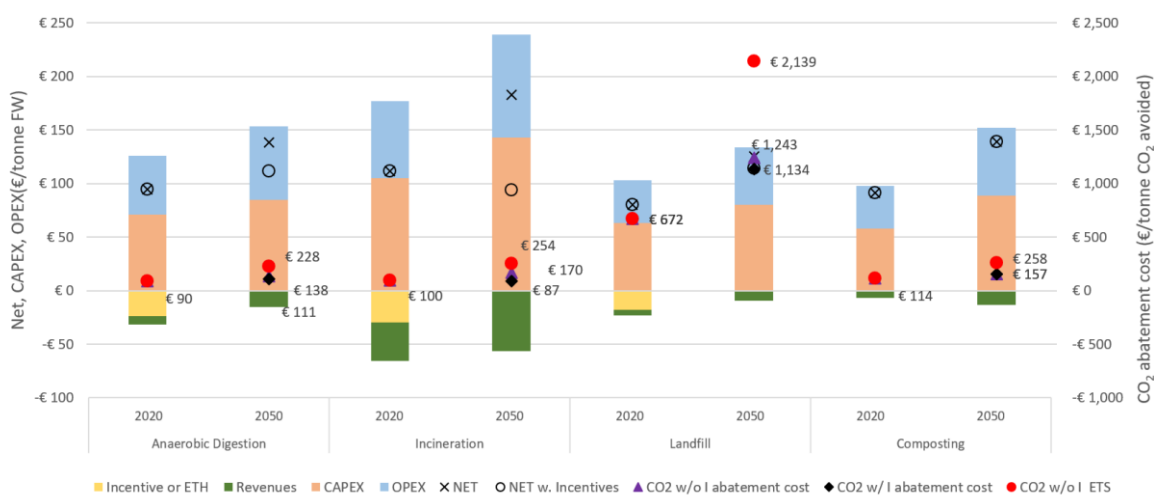


Figure 4.4 : Other impact categories net impacts for (FU)

4.3.1.4 Financial costs

Financial costs in the form of capital, operations, and revenues were calculated and illustrated in Figure 4.5. Electricity substitution was the main source of revenue. Hence, AD and incineration saw savings from the operation due to high financial incentives provided by government, albeit these will most likely fade away in the future. Therefore, net costs were calculated for 2050, both including and excluding incentives. Net value when considering the incentives is indicated in Figure 4.5 with a circle. The landfill had the lowest net costs in 2020.

However, costs associated with reducing CO₂ emissions from landfills were the highest. The CO₂ abatement cost was almost 10 times higher than in other scenarios, due to less CO₂ savings in the landfill. Therefore, landfilling was not preferable. AD had the second highest net cost (94 €) in 2020, due to high OPEX (operating expenses) compared to landfill and composting. However, the costs involved in reducing CO₂ were the lowest in 2020 (€111) and 2050, with no incentive scenario (€138), due to significant CO₂ savings. Incineration had the highest net costs in all scenarios, due to high OPEX and CAPEX (capital expenses). CO₂ abatement costs (87€, 170€, 254€) were lower than landfill (1134€, 1243€, 2139€) and composting (157€, 157€, 258€). Composting, with and without incentive scenarios, had the same results (157€), because there is no energy recovery from composting. Nevertheless, composting still had lower CO₂ abatement costs in all scenarios compared to landfilling, due to CO₂ savings made during the process. The ETS scenario results show that Türkiye would promote the recovery and recycling of organic matter if ETS were included in the waste management system (CE Delft, 2021.) This is especially relevant to the landfill scenario, which has the highest CO₂ abatement costs (2139 €), while composting (258 €) and AD (228 €) have the lowest. The ETS scenario only alters the incineration and composting CO₂ abatement cost ranking. Consequently, the AD scenario has the lowest CO₂ abatement costs in all scenarios, except for the incentive scenario.



* 2050 data for NET, OPEX and CAPEX are represented without incentive results.

* CO₂ abatement costs: the cost of a technology that will reduce greenhouse gas emissions by one tonne, OPEX: operating expenses, CAPEX: capital expenses, ETS: emissions trading system.

Figure 4.5 : Net, capital and operational costs per tonne food waste management (€/tonne FW).

4.3.2 Scenario selection for the business model

When analyzing the carbon emissions, circularity, and cost calculation results, AD offers a promising opportunity to progress in the waste hierarchy by decoupling waste treatment. Small-scale AD plants have been gaining attention in terms of a shift toward a more decentralized system (Thiriet et al., 2020). Decentralized systems have advantages over the traditional centralized system due to reduced transportation needs, increased public participation, and the reinforcement of local nutrient and energy loops (Thiriet et al., 2020). Decreasing fossil fuel is one of the advantages of AD (Slorach et al., 2019). In addition, previous studies have found that AD is the most beneficial solution for managing FW (Levis and Barlaz, 2011; Tian et al., 2021). Similarly, Guven et al. (2019) established that incineration results in more savings due to high energy efficiency assumptions. This findings could be attributed to using the design values of the plant due to a lack of operational data. Previous studies carried out on university campuses in China (Zhou et al., 2021) and the USA (Xu et al., 2022) also found AD as an efficient strategy for FW management, which has advantages over incineration and landfilling such as the valorization of FW, stakeholder partnerships, public participation from the beginning and increasing awareness regarding source separation (Donner et al., 2021). Stakeholders are defined as groups or individuals who may be affected by activities related to BMs (Bertassini et al., 2021), but in circular BM systems, creating value should be a co-operative effort. However, it is important to know that a much more significant benefit would be obtained through FW production prevention in terms of both economic and environmental aspects, as the largest impacts are from the production of food that is wasted – much more than the management itself (Slorach et al., 2019). There is no doubt that CE principles and the waste hierarchy give priority to AD and composting over incineration and landfill (Slorach et al., 2019). Thus, AD was selected for further FW management elaboration with regard to the business plan.

4.3.3 Business model

Technology development is often considered sufficient for real-life applications, but social acceptance and feasible business strategies are also needed (Donner et al., 2020). In addition, it is not possible to define WM under a single stakeholder, as it requires the cooperation of multiple stakeholders. The feedstock business models in this study do not prioritise economic

performance but concentrate rather on the efficient use of resources while maintaining economic viability by reducing energy and material consumption and upgrading generated waste (Donner et al., 2021). Local actors can provide important inputs into system design by responding to needs and challenges (Angouria-Tsorochidou et al., 2021). Hence, it is important to include all stakeholders in the business model design stage. The business model plan for FW management at Ayazağa Campus using AD is compatible with the Osterwalder and Pigneur (2010) and Lagrasta et al. (2021) Canvas model, which is illustrated in Figure 4.2.

The proposed BM will align with the circular economy by slowing down, closing, and narrowing flows while creating value (Pieroni et al., 2021). Ellen Macarthur Foundation (EMF) (2017) promotes the cascading back of nutrients in the soil. However, some elements of the BM, such as key activities, key resources, channels, and value proposition, are not clearly defined by the EMF (Lüdeke-Freund et al., 2019). Biorefineries are widely used for FW management in BMs. The authors of this study decided to use the original Canvas model with the caveat of inserting two additional levels, namely the environmental and social (Lagrasta et al., 2021). This model has common SBM and CBM clusters, namely resource efficiency (narrowing loops), resource longevity (slowing, closing loop) and economic growth, but it does not belong to either. Instead, it is a simplified version of CBMs (Pieroni et al., 2021).

In the model created for ITU, stakeholders, and income/expenditure tools are defined with how FW sent to the AD system will travel within the CE. According to the proposed model, key partners are logistic partners and university administration/residents, in the short term, and municipality and farmers in the medium term. Local willingness to separate waste at source is a significant key resource, which in the case of ITU was found to be receptive to engaging in this scenario. Renewable energy production in the form of biogas, chemical fertilizer substitution from recovered nutrients, and closing the loop for waste are quantitative value propositions in the BM, whereas raised awareness about waste to energy and nutrient circulation concepts are qualitative value propositions. Customers on the small scale are techno-park (university science and business parks within the campus, set up to enhance research and development and collaboration with industries), campus residents (students, academic staff, and administrative staff), and university administration in the initial stage. Potential customers are the district municipality and local farmers in the medium term. Sources of income are fertilizers and electricity. In addition, if a special agreement is made with the district municipality, an

additional source of income may be created. Operating costs are the salary of the technician working at the facility, the transportation of waste from the cafeteria to the facility, and other costs. The cooperation of internal and external stakeholders, living lab experience, stakeholder engagement, and increased awareness are important social outcomes. The prevention of FW and the implementation of a practical solution for managing it accordingly represent an important social value of the proposed BM (Ribeiro et al., 2018). The suggested BM also engages the community in managing FW, and feedback is provided during deliverance by personal contact (Ribeiro et al., 2018).

Key Partners -Logistic partners -University administration/ residents (students, staff, etc.) -Municipality -Farmers	Key Activities -Manage resource flow -Biogas production -Fertilizer production -Education of local partners	Value Propositions -Biogas, renewable energy production (avoiding fossil fuel during energy consumption) -Fertiliser (organic) -Closing loop for waste. -Raised awareness on waste to energy concept	Customer Relationships -Pilot cases -Education about CE -Visiting possible local partners	Customer Segments <u>Small Scale</u> -Technopark -Campus residents -University administration <u>Medium-Scale</u> Municipality -Farmers (both scales) or gardening
	Key Resources -Local willingness to separate waste at source. -Human resources -Know how -Employees		Channels -Infrastructure for electricity -Local sales	
Cost Structure -O&M technical -Salary -Logistics -Capital costs for infrastructure			Revenue Streams -Fertilizer sale -Electricity sale -Agreement with district municipality	
Social & Environmental Requirements -Experiment (living lab) -Stakeholder engagement -Monitoring system -Increased awareness			Social & Environmental Benefit -Being a pioneer for the CE in the region -Nutrient recovery -Avoiding landfill and methane release into the atmosphere	

Figure 4.6 : Business model for FW management on Ayazağa campus, according to the Osterwalder and Pigneur (2010) Canvas model and Lagrasta et al. (2021)

The public could rely on renewable biological sources while pursuing economic growth through initiatives consistent with CBMs (Reim et al., 2019). Nevertheless, each study comes with its own limitations. Firstly, Zaman (2012) clarified that the major issues challenging the success of the waste management business are finding potential partners and creating public-private partnerships. In addition, a lack of regulatory policies and standards that support new business

models and competition with fossil fuels also act as barriers (Lüdeke-Freund et al., 2019; Donner et al., 2021). Secondly, it is not realistic to expect feedstock without impurities (Slorach et al., 2020). A previous questionnaire study conducted on Ayazağa campus shows a public willingness to participate in zero waste (to landfill) management (Maçın et al. 2020) and source separation of waste. However, quantitative social aspects specific to FW should be measured in the future. Finally, rebound effect is a significant problem for the long term applications. A properly implemented BM can avoid the rebound effect by aiming to produce the minimum amount of waste and maximum practical benefit from products with biomass cascading, waste hierarchy and efficiency when using biomass or renewable resources (Salemdeeb et al. 2017). However, due to the unforeseeable nature of highly complex systems, such as a circular economy market system, it is likely impossible to find meaningful conditions that are both necessary and appropriate (Zink and Geyer, 2017). While we acknowledge the limitations of the study, this does not negate the value of these results, highlighting the important relationships between decarbonisation and, circularity in multiple FW management scenarios.

The decarbonizing ranking results revealed that three scenarios other than the landfill scenario achieve the zero-waste-to-landfill target, albeit the results are very different in terms of circularity, carbon emissions, and costs. This indicates that a variety of aspects should be considered before selecting technologies for goal-originated WM systems and BM planning. This study compared a decentralized pilot-scale AD facility with full-scale centralized incineration, composting, and landfill facilities. The effect of a centralized AD plant is a significant topic for larger-scale studies. Transport-related emissions in AD (decentralized) have been omitted in this study due to lack of knowledge of where AD sites will be placed, there is therefore not enough information to assess whether omitting this transport has any impact on the conclusion. Previous studies have shown that waste collection fuel consumption may range from 1.4 L to 10.1 L of diesel/tonne of waste. A collection of source separated FW (feedstock) and transportation of digestate will lead to additional efforts, which results in increased fuel consumption and emissions. Care should therefore be taken to find an appropriate balance between the scale of the facility and the collection distance while selecting the site for centralized AD facilities. The BM should be verified for larger studies as it was previously the case for similar models such as Ecocanvas and TLBMC. This research is a non-profit academic

study: how it will be utilized by businesses and possible partnerships with various stakeholders should be considered in the future.

4.4 Conclusions

The new business model in this study seeks to create a platform for sharing ideas while promoting best practices relating to the environmental and economic aspects of WM. The emission calculations show that incineration offers the greatest carbon savings, followed by AD, but it provides neither decoupling nor circularity for natural resources. Also, it has high toxicity compared to the AD. Additionally, current and future cost calculations show that incineration has the highest uncertainty in terms of net costs, while AD has the highest circularity and lower uncertainty for the future. The business model is designed in tandem with the living lab concept, in which the end-user is actively engaged in the planning and finalisation of the system. While, the quantitative outcome is renewable energy production in terms of biogas, chemical fertilizer substitution, and closing of the nutrient loops, a qualitative outcome is seen in raised awareness on waste to energy concepts. On the other hand, rebound effect production trends and consumer demands are significant indicators to be conquered.



5. “DECOUPLING ANALYSIS” OF WASTE MANAGEMENT EMISSIONS IN ISTANBUL: A COMPREHENSIVE MFA-LCA TOWARDS ACHIEVING THE ZERO WASTE TO LANDFILL GOAL

5.1 Introduction

Solid waste management (SWM) is a significant contributor to global greenhouse gas (GHG) emissions, accounting for approximately 5% of global releases (Turner et al., 2015; Zhao et al., 2021). The majority of these emissions are attributed to landfills (Turner et al., 2016; Zaman, 2016). Despite efforts to move up the waste hierarchy, many countries, including Bulgaria (64%), Spain (57%), Hungary (51%), Portugal (49%), U.S.A., (52.6%), Brazil (59.1%), Mexico (65%), China (79%) and Türkiye (90%) , still rely heavily on landfilling (Vaverková, 2019; TSI, 2021). To address this issue, the concept of zero waste (ZW) to landfill has gained popularity worldwide, including in the USA and Türkiye (Castigliero et al., 2021; MoEU, 2019). The initial objective of ZW strategies is to reduce waste generation through responsible production and consumption, while maximizing reuse and recycling efforts (Zaman, 2015). However, ZW to landfill strategies are also widely applied, with the primary goal of minimizing the amount of waste sent to landfills (Castigliero et al., 2021). Despite the widespread adoption of ZW strategies, measuring their performance remains a challenge (Song et al., 2015). Merely relying on landfill diversion as an indicator is insufficient to assess the effectiveness of ZW goals and achieve true decoupling from resources (Zaman, 2014).

Decoupling, originally introduced by von Weizsäcker (1989) and further developed by the OECD to analyze the relationship between economic development and environmental impacts (OECD, 2002). Decoupling occurs when economic growth outpaces environmental emissions by original definition (Muangthai et al., 2014). Decoupling indicators are used to measure this phenomenon, typically with an environmental pressure variable as the numerator and an economic variable as the denominator (driving force), although alternative variables can also serve as the

denominator (OECD, 2002). The issue of decoupling environmental pressure (such as carbon emissions) from economic growth has endured over time (Wu et al., 2023) various topics including waste management.

Decoupling studies have predominantly focused on landfill diversion, with limited analysis of waste management related emissions. However, there are existing decoupling studies that analyze the carbon footprint of SWM (Cifrian et al., 2012) and the effectiveness of ZW strategies on global warming potential in urban settings (Castigliengo et al., 2021). While previous SWM decoupling studies have predominantly focused on Europe, there is no consensus on the results of the environmental implications. However, some instances of relative decoupling (occurs when environmental pressures increase at a slower rate than the economic activity driving them) have been observed, particularly in terms of landfill diversion, attributed to regulations and policies such as landfill bans and taxes. Additionally, cases of absolute decoupling (there should be both sustained economic growth and a reduction in environmental pressures) have been observed with reductions in the amount of biodegradable waste being sent to landfills (Wu et al., 2023; Mazzanti et al., 2012). Although relative decoupling of waste generation and absolute decoupling of landfilled waste have been evidenced in OECD countries, absolute decoupling in terms of waste generation remains elusive. The EU policy framework has proven effective in achieving waste diversion from landfills but does not appear to incentivize waste prevention. Furthermore, specific waste quantity reductions required to achieve objectives have not yet been specified at the EU level.

Despite Türkiye's efforts and investments from climate finance, the country still lags behind in meeting reduction targets. Only a few studies have been conducted in Türkiye on topics such as decoupling (Karakaya et al., 2019), CO₂ emissions decomposition from energy use (Ipek Tunç et al., 2009), and the environmental Kuznets curve (Akbostancı et al., 2009). Istanbul, as the most developed city in Türkiye in terms of population, technology, and GDP, is at the forefront of new strategies and technology investments in the waste management sector. While previous SWM studies have been conducted in Istanbul, they have mostly focused on specific waste types or employed other methods such as multiple-criteria decision making (MCDM) or mathematical modeling. Only one study has been found that

explores the climate change effects and SWM relations of Istanbul for the future (IMM, 2015).

Accurate and reliable forecasting of waste amounts is crucial for developing sustainable SWM strategies, (Zaman & Lehmann, 2013), as it aids decision-makers in understanding potential savings and burdens associated with future SWM strategies (Zaman & Lehmann, 2013). Previous waste amount forecasting studies have been conducted for the ZW goal (Ayeleru et al., 2018) to assess substitution potential, but detailed calculations of possible environmental impacts were lacking. Material flow analysis (MFA) results can help prevent overcapacity of SWM facilities, while life cycle assessment (LCA), a well-known tool for SWM, has not been extensively utilized in waste management decoupling studies to the best of our knowledge. Only one LCA study has been found, which assessed the decoupling of economic growth and environmental impacts of consumption in EU-28 countries (Sanyé-Mengual et al., 2019). No study has been found that employs LCA to assess the environmental impacts of SWM strategies and their effect on waste management emissions decoupling using historical and forecast data for a specific city. To effectively pursue comprehensive targets like ZW, an examination of historical data and future SWM scenarios based on forecasted data is essential. Assessing future SWM strategies using forecasted data and tools like LCA enables policymakers to develop new goal-oriented strategies. LCA provides insights into the environmental consequences of SWM technology changes, from traditional landfill methods to cleaner waste-to-energy (WtE) technologies, across various impact categories. Today, decision makers have become aware of the significance of waste management and decoupling concepts due to the emergence of global problems such as climate change.

The assessment of future SWM strategies using forecasted data and tools like LCA plays a significant role in helping policymakers develop new goal-oriented strategies. Therefore, this study introduces a novel approach for SWM decoupling analysis, utilizing LCA as a tool to understand environmental impacts. Furthermore, it endeavors to overcome the obstacle of data limitation through the use of MFA, which is one of the main challenges for waste management studies in developing countries. The study aims to analyze the environmental impact and decoupling of the SWM

sector in Istanbul, as well as the effect of new SWM strategies on decoupling towards the ZW goal in the city, using forecasted data until 2050. In pursuit of this objective, the paper addresses the following questions: (i) Whether landfill diversion (ZW strategies) is driving relative or absolute decoupling and whether the current policy targets are sufficient for achieving decoupling in Istanbul, (ii) What are the decoupling trends of other impact categories in Istanbul until 2050, and (iii) How will the usage of WtE technologies (incineration and anaerobic digestion) and renewable energy sources in the country impact waste management emissions?

5.2 Methodology

5.2.1 Decoupling approach

Decoupling shows the dependent relationship of various variables, therefore, environmental decoupling can be assessed beyond economic aspects (GDP). Türkiye's economy is experiencing strong fluctuations, and GDP is difficult to predict. The study focuses on "emissions decoupling," which refers to the separation of emissions from waste management growth (Figure. 5.1). Previously literature has often reported a strong relationship between municipal waste and economic growth (e.g., Kuznets (Kuznets, 1955) and Environmental Kuznets Curve). However, our study did not demonstrate this relationship. In Türkiye, estimating waste amounts based on population projections and developing a policy and investments accordingly is the common approach.

It is expected that the newly established processing facilities will remain operational for the next 30 years (Zhao et al., 2022). Therefore, a combination of MFA and LCA methods was employed to conduct a future decoupling analysis of waste management in Istanbul. The future decoupling analysis involved comparing the total waste generated and the associated W waste management emissions. Decoupling is observed when waste quantity increases while waste emissions decrease

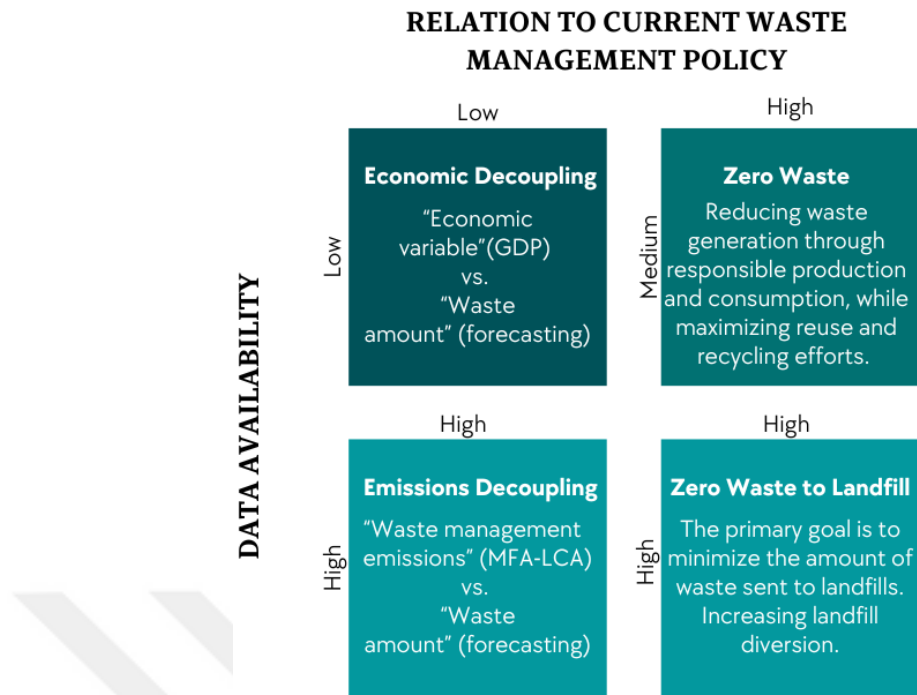


Figure 5.1 : Decoupling and zero waste concepts in the study

Several driving factors were considered for waste generation, including trends in per capita waste, waste characterization, and population changes (Izquierdo-Horna et al., 2022). The amount of future waste was estimated by multiplying the projected population of Istanbul by the waste generation per capita. To determine the future population, five different population calculations were utilized done (Figure. 5.2) for winter and summer conditions and the one with the highest R2 results was selected. Additionally, waste characterization changes were taken into account based on previous studies conducted in Istanbul (Yildiz et al., 2012; Ak and Sezer, 2020), as waste characterization significantly impacts overall LCA results (Bisinella et al., 2017). The Appendix provided detailed information on waste generation and composition among 2020-2050 years. Separate calculations were performed for the Asian and European sides of Istanbul, considering slight differences in SWM activities on the two continents. It is important to note that Istanbul represents the combined results of both sides in this article.

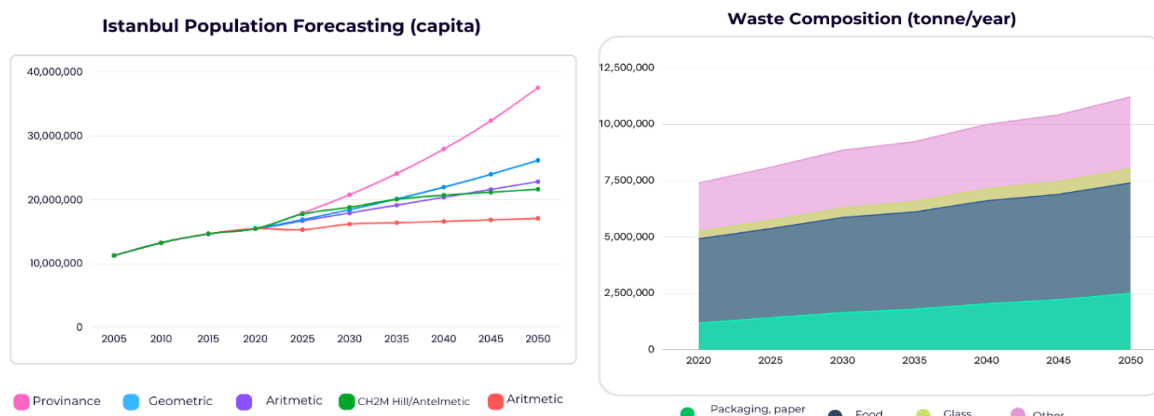


Figure 5.2 : Population and waste composition forecasting of Istanbul

5.2.2 Scenarios

The study examines four main scenarios representing the current waste management (WM) situation in Istanbul and the future plans of the municipality. It includes all WM facilities operated by Istanbul Metropolitan Municipality (IMM), as well as small to medium-sized materials recovery facilities (MRFs) operated by the private sector. The scenarios encompass a Business as Usual (BAU) scenario for Istanbul, as well as future scenarios for 2035 and 2050. The key differences between the BAU and future WM strategies lie in the increased MRF capacity and the adoption of WtE technologies, specifically anaerobic digestion (AD) and incineration, in Istanbul.

In the scenarios, the AD facilities are fed by source-separated food waste, while the input for incineration is mixed MSW obtained from transfer stations. The study anticipates a gradual increase in the collection of source-separated municipal packaging waste by the municipality and authorized institutions, while a decrease is expected in the activities of scavengers. Impurities were taken into consideration during the Material Flow Analysis (MFA) and Life Cycle Assessment (LCA) studies.

Further details of the scenarios are explained below, and Table 5.1 provides information on the capacity of the WtE technologies and key aspects of each scenario.

Business as usual scenario (year 2020, 2035 and 2050) [BAU]: The BAU scenarios analyze waste management (WM) performance in Istanbul for 2020,

2035, and 2050. The only difference among these years is the amount of waste generated. Currently, around two-thirds of the waste in Istanbul originates from the European side, while the remaining third comes from the Asian side (Korkut et al., 2018). Waste collection is the responsibility of district municipalities, using containers with capacities of 400 L and 800 L (Korkut et al., 2018). Collection of household waste in Istanbul is highly efficient (Korkut et al., 2018), assuming all generated waste is collected. Most waste is sent to transfer stations, with a small portion directly going to landfills. There are four transfer stations on both the Asian and European sides. Waste transportation and disposal are managed by the IMM, while the operations of WM facilities are conducted by the Istanbul Environmental Management Industry and Trade Co. (ISTAC Co.), a semi-private company established by IMM (Kanat, 2010; Korkut et al., 2018).

As of 2020, the majority of waste is still sent to landfills, while the remaining waste is directed to the Kisirmandira Facility (KF) on the European side and the MBT facility on the Asian side. KF includes materials recovery and composting facilities. Compost usage in Istanbul is currently limited to IMM parks, as the compost market is underdeveloped. Additionally, part of the KF's output is utilized in an RDF facility to replace petrocake. A large-scale MBT facility employing a bio-drying process followed by RDF production operates on the Asian side, managed by IMM (Güven et al., 2019). Two main types of MBT are used in the WM sector: mechanical biological pretreatment (MBP) and mechanical biological stabilization (MBS) (Montejo et al., 2013). The Komurcuoda MBT facility is categorized as MBS, prioritizing RDF production and material recovery (Öztürk et al., 2015). Packaging, cardboard-paper, and glass packaging waste are collected separately (Yıldız-Geyhan et al., 2016) and sent to a single-stream MRF. The recovery efficiencies of both single-stream and mixed waste MRFs are below 100%, with the rejected stream being sent to landfills. In Istanbul, scavengers play a significant role in collecting recyclable packaging waste (Yıldız-Geyhan et al., 2016). Despite their decreasing numbers, scavengers are assumed to still be present in the system in 2035 and 2050, based on another study suggesting societal acceptance and their contributions to recycling (Yıldız-Geyhan et al., 2016). However, obtaining quantitative data within this illegal system is challenging (Yıldız-Geyhan et al., 2017), and the waste amounts collected by scavengers are therefore based on survey

results (Tutar M., 2020; Güğüş E.A., 2019; Ukşal F., 2019; Deniz G.C., 2010; Fakioğlu E., 2010).

Prospective plan (year 2035 and 2050) [Prosp]: Building on the BAU scenario, this scenario incorporates the future plans of IMM to achieve climate target by 2050 (IMM, 2021). The primary objective of this plan is to divert 50% and 100% of total waste from landfills. By 2050, it aims to achieve 50% source separation of FW and 100% recycling of cardboard, paper and packaging waste. Additionally, the plan targets to collect 75% and 85% of landfill gas by 2035 and 2050 respectively. However, achieving the IMM's ambitious targets for 2050 are challenging, leading to development of alternative scenarios.

Alternative 1: Prospective plan + Incineration capacity (2035 and 2050 [Prosp +Inc]: This scenario was designed to assess the impact of increased capacity. In addition to the [Prosp] scenario, the total incineration capacities in Istanbul increase to 7250 tonne/day and 12550 tonne/day, for 2035 and 2050 respectively, according to this scenario.

Alternative 2: Prospective plan + Incineration capacity –AD capacity (year 2035 and 2050) [Prosp+Inc-AD]: This scenario aims for a more realistic approach by assuming reduced source separation of food waste compared to the [Prosp +Inc] scenario. In Istanbul, the total anaerobic digestion (AD) capacities decrease to 2575 tonne/day and 5424 tonne/day in 2035 and 2050, respectively. The remaining food waste is directed to landfill and incineration.

Table 5.1 : Istanbul waste management targets

Capacities	Source separated food (%)	Source separated packaging (%)	Total recovery (%)	Incineration capacity (tonne/day)*	Landfill gas capture (%)	Landfill diversion (%)
<u>[Prosp] 2035</u>	50	60	60	4,250	75	50
<u>[Prosp +Inc] 2035</u>	50	60	60	7,250	75	50
<u>[Prosp +Inc-AD]2035</u>	25	60	60	7,250	75	50
<u>[Prosp] 2050</u>	100	100	100	7,250	85	100
<u>[Prosp +Inc] 2050</u>	100	100	100	12,550	85	100
<u>[Prosp +Inc-AD] 2050</u>	50	100	100	12,550	85	100

5.2.3 Material flow analysis (MFA)

MFA is utilized to determine the future WM facility capacities. MFA encompasses the system, process and flow adhering to the law of mass-conversion (Brunner and Rechberger, 2004). The STAN 2.6 software, developed by Vienna University of Technology (Cencic, 2016), was used to calculate and visualize the WM flow of Istanbul. In the MFA, landfills and use-on-land processes are considered as stocks within the system, while materials entering (Input) and leaving (Export) the system for further external processes denoted as (I) and (E). The recovery calculations in the study focus solely on inputs, and do not include rejections or other outputs from WM facilities. Hence, the recovery rate and real landfill diversion ratios were calculated separately in the result section. Landfill diversion is defined as the fraction of waste generated that is not disposed of in a landfill. Detailed information regarding the MFAs can be found in Appendix.

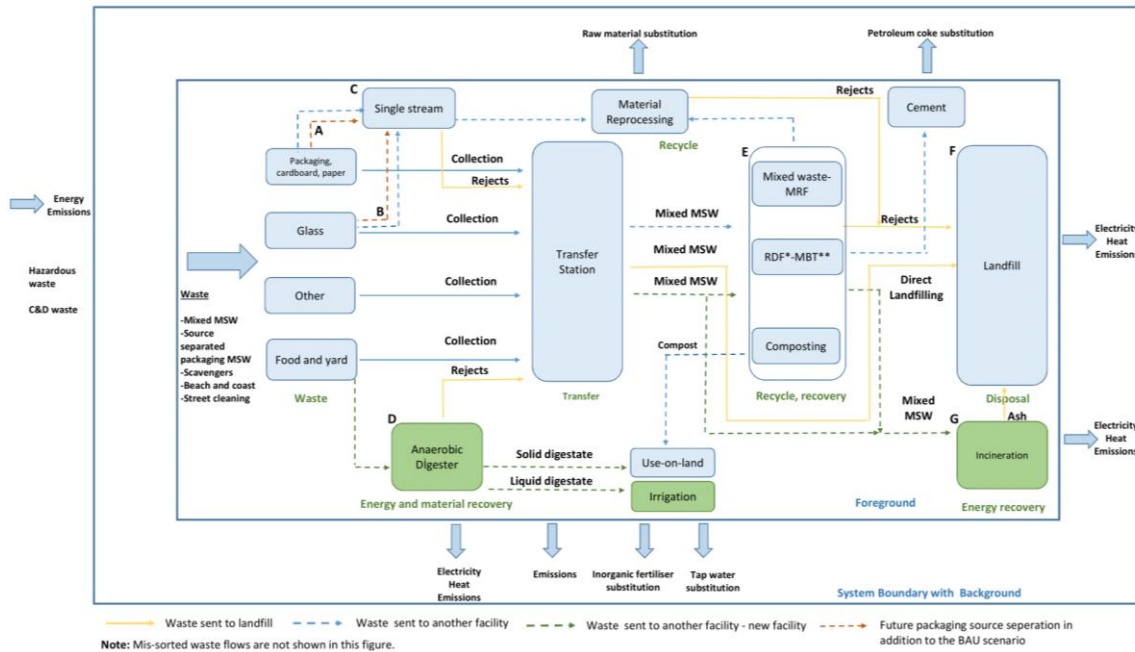
5.2.4 Life cycle assessment (LCA)

5.2.4.1 Aim, scope and boundaries

To quantify the changes in waste management (WM) emissions resulting from zero waste (ZW) strategies, a MFA followed by a LCA was performed. The LCA employed a consequential approach and utilized background processes from the ecoinvent database, specifically the "Substitution, consequential, long-term" dataset. The LCA modeling was performed using EASETECH (Clavreul et al. 2014). The objectives of the LCA study were: (i) to assess the environmental impacts of the WM system in Istanbul, (ii) to forecast how these impacts will change in the future with waste-to-energy (WtE) technological investments and increased capacity of material recovery facilities (MRFs) due to source separation, and (iii) to evaluate WM emissions to determine whether the ZW landfill goal contributes to relative/absolute decoupling.

The geographic scope was set as the city of Istanbul. The functional unit (FU) of the study was defined as the annual waste production in Istanbul, covering mixed MSW (including household and commercial waste), source-separated packaging waste from various sources (residential, authorized institutions and scavengers), street cleaning waste, beach waste, and coast cleaning waste. The total annual

waste amounts for the years 2020, 2035, and 2050 were calculated, excluding construction and demolition (C&D) waste and hazardous waste. The waste flows were categorized into four main waste types: packaging waste (including paper and cardboard), glass, food waste, and other waste types, considering both current and future WM plans in Istanbul. The WM system for the LCA includes waste collection, transportation, MRF operations, recycling, RDF production, and landfilling (see Figure 5.1.).



*MBT in Asia with 2000t/day is operated by IMM while RDF in Europe is accepted as operated by private sector company.
 ** Composting unit is only operated in European side is assumed as used on land while composting unit output in Asia side (MBT facility) is sent to landfill. ***A-B: Future packaging and glass source separation in addition to BAU scenario; C: single stream MRFs capacity, D: Anaerobic digester capacity, E: Integrated WM facility capacity, F: Landfill capacity, G: incineration plant.

Figure 5.3 : Waste management system and LCA boundaries

5.2.4.2 Data sources and modelling assumptions

Standard methods were followed to the LCA; ISO 14040 and 14044 (ISO, 2006a; 2006b). The EASETECH software, specifically designed for complex WM problems was employed. The study adopted the zero-burden assumption disregarding upstream burdens, a consequential approach was applied, and multi-functional processes was addressed through system expansion (ISO, 2006b).

Waste amount, collection and transportation data was primarily sourced from the annual report of ISTAC (ISTAC, 2020) and the IMM database (IMM, 2020).

Additionally, previous WM studies specific to Istanbul were utilized for inventory purposes, including waste characterization (Yildiz et al., 2012; Ak and Sezer, 2020), WM facility operation (Yıldız-Geyhan, 2016; Kurdoğlu, 2013; Sezer and Arikan, 2011; Gencsoy 2010; Kara et al., 2009), collection and transportation (Yaman, 2011; Sezer, 2019), and environmental assessment of WM strategies (Coban et al., 2018; Guven et al., 2019). The Ecoinvent 3.7.1 (substitution, consequential, long-term) database (Ecoinvent, 2020) was used for background processes. Further details of foreground and background LCI data can be found in the Appendix.

For the electricity produced by the WM system, an equivalent amount of electricity generated from Turkish electricity mix was assumed to be substituted. The analysis considered the potential changes in renewable energy sources in the energy mix and their impact on the WM system. Türkiye, like many developing countries, aims to increase the share of renewables in its electricity mix, leading to a decrease in avoided carbon emissions through fossil fuel substitution (Castigliero et al., 2021). Under the original scenario, the use of fossil sources (coal and natural gas) was projected to decrease from 57.7% to 23.2%, resulting in a reduction of the electricity coefficient from 0.55 kg CO₂-eq to 0.05 kg CO₂-eq by 2050 for a global warming potential (GWP) of 100 years.

The heat produced by the AD was assumed to be used by the facility itself to maintain a certain temperature (mesophilic conditions) and the remaining heat was fed into the district heating network. Material recycling calculations were performed based on the performance of the MRF. The real substitution of virgin materials was estimated considering market conditions and literature sources (Fernández-Braña et al., 2020). The long-term effects of the WM system were limited to a 100-year timeframe. The impact categories were based on the ILCD recommended method described by Hauschild et al. (2012) and are provided in the Appendix. Finally, a Data Quality Assessment (DQA) and sensitivity analysis were conducted.

5.2.4.3 Data quality assessment

The DQA method suggested by Weidema and Wesnæs (1996) was employed considering data quality indicators such as technological representativeness, geographical representativeness, time-related representativeness, completeness, and precision uncertainty (EC-JRC, 2011). Based on the EC-JRC (2011) guidelines, data quality results (DQR) were classified (calculated by using Equation 5.1), into three categories: "high quality" (DQR < 1.6), "basic quality" (DQR between 1.6 and 3), and "estimate" (DQR > 3).

$$\text{DQR} = \frac{\sum \text{data quality indicators} + \text{weakest data quality indicator} \times 4}{n \text{ of data quality indicators} + 1} \quad (5.1)$$

5.2.4.4 Perturbation analyses and scenario analysis

Contribution analysis and data quality assessment (DQA) findings were applied to identify parameters for perturbation analysis, following the approach outlined by Clavreul et al. (2012). Perturbation analysis involved making small parameter changes of 10% to evaluate their impact on overall results. Sensitivity ratios (SR) were calculated by using Equation 5.2, and normalized sensitivity ratios (NSR) were used (Equation 5.3), to compare the SR results.

$$\text{Sensitivity Ratios (SR)} = \frac{\frac{\Delta \text{ Results}}{\text{Initial Result}}}{\frac{\Delta \text{ Parameter}}{\text{Initial Parameter}}} \quad (5.2)$$

Where normalized sensitivity ratios were calculated as:

$$\text{Normalized Sensitivity Ratio (NSR)} = \frac{\text{abs}(\text{SR})}{\max(\text{abs}(\text{SR}))} \quad (5.3)$$

Scenario sensitivity analysis was conducted to assess the influence of different choices on the final outcomes. Short-term effects of methane emissions from landfills were evaluated using global warming potentials for 20-year time horizons provided in the 6th assessment report of the Intergovernmental Panel on Climate Change (IPCC, 2021; Zhao et al., 2022). Additionally, the impurity ratio in the

containers, representing the uncertainty of source separation, was determined based on the optimal separate collection rate (Haupt et al., 2018b).

5.2.4.5 Uncertainty analysis

To identify critical parameters, DQA and NSR values were combined following the method employed by Andreasi Bassi (2017). DQA and NSR data were categorized as low, medium, or high, represented by green, yellow, and red colors, respectively. These values were plotted on a decision graph, and parameters falling within the yellow and red regions were considered critical and included in the uncertainty analysis (Andreasi Bassi, 2017). Uncertainty analysis was conducted using Monte Carlo simulation with 10,000 iterations. Detailed information regarding the critical data and uncertainty analysis can be found in the Appendix.

5.3 Result and Discussion

5.3.1 Material flow analysis

MFA provides valuable insights into future landfill diversion rates and the required capacity of waste management facilities. This helps prevent under or overestimation of investments in waste management. MFA results reveal the disparity between the actual landfill diversion and recovery of waste management systems, as certain factors like source separation and substitution ratios hinder achieving 100% landfill diversion. Recovery calculations were based on the inputs of waste management facilities, whereas the real landfill diversion took into account the overall performance and final outputs of these facilities.

The total annual waste amounts for the years 2020, 2035, and 2050 were calculated as 7.4, 9.2, and 11.2 million tonnes, respectively. In terms of real landfill diversion, Asia exhibits higher rates compared to Europe, mainly due to the presence of MBT facilities. However, emerging technologies and future waste management projects are predominantly implemented first on the European side. Consequently, in 2035, landfill diversion is higher in Europe, while in 2050, it becomes higher in Asia due to the establishment of new incineration plants.

In 2020, the amount of waste landfilled is 4.2 million tonnes, with a significant portion originating from TS (Transfer Station). In Europe, the landfilled waste amount in 2035 ranges between 5.2 million tonnes [BAU] and 2.4 million tonnes [Prosp+Inc], while in Asia, it ranges between 2.8 million tonnes [BAU] and 1.3 million [Prosp +Inc]. By 2035 and 2050, incineration and AD handle the majority of the waste, whereas landfilling remains prominent in BAU scenarios across all years. Overall, the maximum landfill diversion in Istanbul reaches 56.0% in 2035 and 95.2% in 2050. However, it is important to note that MFA results and landfill diversion rates do not encompass the environmental impacts associated with waste management or its actual impact on decoupling efforts.

Table 5.2 : Istanbul future waste management scenarios

Capacities	Total waste amount (tonne/year)	Food waste amount (tonne/year)	Source separated food amount (tonne/year)	AD capacity (tonne/day)	Packaging waste amount (tonne/year)	Source separated Packaging amount (tonne/year)	MRF capacity (tonne/day)	Incineration capacity (tonne/day)
<u>[BAU] 2020</u>	7,404,375	3,122,754	-	-	2,283,951	1,206,492	3,646	-
<u>[BAU]2035</u>	9,232,680	3,759,957	-	-	2,772,950	1,221,812	3,646	-
<u>[BAU]2050</u>	11,204,919	3,959,859	-	-	3,920,505	1,238,666	3,646	-
<u>[Prosp] 2035</u>	9,232,680	3,759,957	1,879,979	5,151	2,772,950	1,663,770	4,558	4,250
<u>[Pros+Inc] 2035</u>	9,232,680	3,759,957	1,879,979	5,151	2,772,950	1,663,770	4,558	7,250
<u>[Pros+Inc-AD]2035</u>	9,232,680	3,759,957	939,989	2,575	2,772,950	1,663,770	4,558	7,250
<u>[Pros] 2050</u>	11,204,919	3,959,859	3,959,859	10,849	3,920,505	3,920,505	10,741	7,250
<u>[Pros+Inc] 2050</u>	11,204,919	3,959,859	3,959,859	10,849	3,920,505	3,920,505	10,741	12,550
<u>[Pros+Inc-AD] 2050</u>	11,204,919	3,959,859	1,979,930	5,424	3,920,505	3,920,505	10,741	12,550

Capacities	Source separated food (%)	Actual Food waste recycling (%)	Source separated packaging (%)	Actual Packaging recycling (%)	Actual landfill diversion (%)
<u>[BAU] 2020</u>	-	2.7	55.4	35.3	15.5
<u>[BAU]2035</u>	-	2.2	44.1	26.7	11.4
<u>[BAU]2050</u>	-	2.1	31.6	20.0	11.1
<u>[Prosp] 2035</u>	50	42.0	60.0	34.0	44.0
<u>[Pros+Inc] 2035</u>	50	42.0	60.0	34.0	56.0
<u>[Pros+Inc-AD]2035</u>	25	22.0	60.0	34.0	48.0
<u>[Pros] 2050</u>	100	80.6	100.0	74.0	78.0
<u>[Pros+Inc] 2050</u>	100	80.6	100.0	74.0	95.2
<u>[Pros+Inc-AD] 2050</u>	50	41.7	100.0	74.0	81.5

5.3.2 Life cycle assessment

5.3.2.1 Contribution analysis

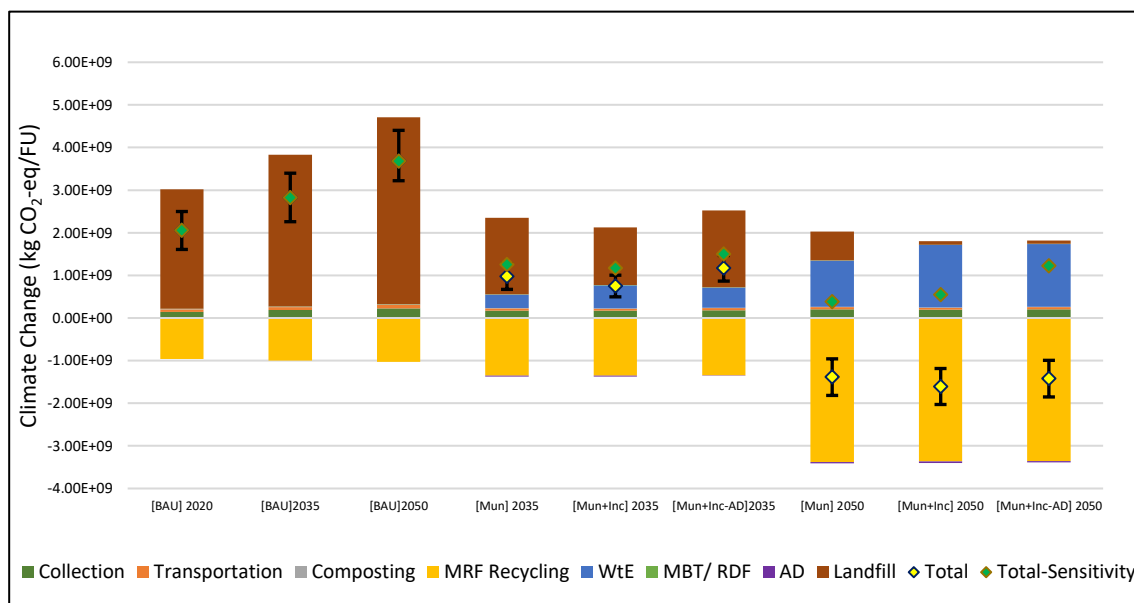
The contribution analysis results (Figure 5.4) indicate that the majority of burdens originate from landfills, while savings arise from the MRF, representing material substitution. Future Zero Waste plans have the potential for net savings in 2050. However, the composting facility's global warming savings from fertilizer substitution are outweighed by its burdens. Considering the lack of a compost market in Türkiye and the absence of energy recovery options, composting was not

selected as a viable future waste management strategy from an environmental perspective.

On the other hand, diverting organic waste from landfills through AD offers additional benefits beyond fertilizer substitution and energy recovery. The AD facility achieves net savings as it accepts source-separated food waste and utilizes the biogas, which outweighs the emissions released from the facility. These benefits comes on top of the avoided impact from the landfill emissions. These findings align with the study by Turner et al. (2016).

While municipality-driven scenarios are feasible for constructing new facilities, achieving 50% source separation of food waste, especially in households, poses significant challenges. Therefore, alternative ZW scenarios were designed based on high-capacity incineration plants capable of processing mixed collected waste. These findings support previous studies that recommend limiting the organic fraction in landfills, increasing MRF capacity (Ayvaz-Cavdaroglu et al., 2019), establishing new incineration plants, and efficiently utilizing AD by reducing operating costs (Coban et al., 2018) in Istanbul. Furthermore, Guven et al. (2019) found that incineration has the lowest environmental burden for climate change in food waste management. Additionally, Taşeli (2007) suggested the establishment of new incineration plants for Türkiye.

The transition of the energy mix change from fossil based to more non-fossil sources by 2050 results in an increase in the climate change impact category. Notably, the incineration plant show the highest burden in 2050, despite having the highest proportion of energy produced but substituting a renewable (non-fossil) electricity mix. In light of this predicament, emerging technologies such as carbon capture and storage (CCS) is strongly recommended (Bisinella et al., 2021) and results show that CCS can fundamentally alter the net impact of the WtE facility, converting it from a burden to a source of savings. This highlights the capacity of CCS to play a pivotal role in achieving net zero carbon cities while concurrently preserving WtE as a viable component of the waste management infrastructure.



* The sensitivity analysis is calculated based on the results of the study (Haupt et al., 2018b), which represent "impurities increase with higher source separation".

Figure 5.4 : Climate Change impact category results of default and sensitivity scenarios Istanbul (kg CO₂-total waste)

5.3.2.2 Decoupling analysis

The decoupling analysis presented in Figure 5.4 provides insights into the decoupling of climate change impacts and its relationship with landfill diversion. A figure for all impact categories are available in the Appendix.

Climate change

The analysis of the climate change category reveals that coupling of emissions occurs in the BAU scenario, whereas decoupling is observed in the alternative improved scenarios. Specifically, it was found that landfill diversion contributes to emissions reduction, and diverting especially food waste to WTE and AD facilities generally leads to increased saving. The impacts of landfill are more dramatic from a shorter time point of view (GWP-20 years), due to the large releases of methane (see Appendix). However, as landfill diversion increases, methane impacts will decrease in the improved scenarios. In addition, savings from MRF facilities lead to higher savings relative to 100-year GWP results. Consequently, total net emissions of GWP-20 years in 2035 are greater than GWP-100 years, but were more reversed in 2050 due to increased landfill diversion and recycling of packaging waste (Figure 5.5).

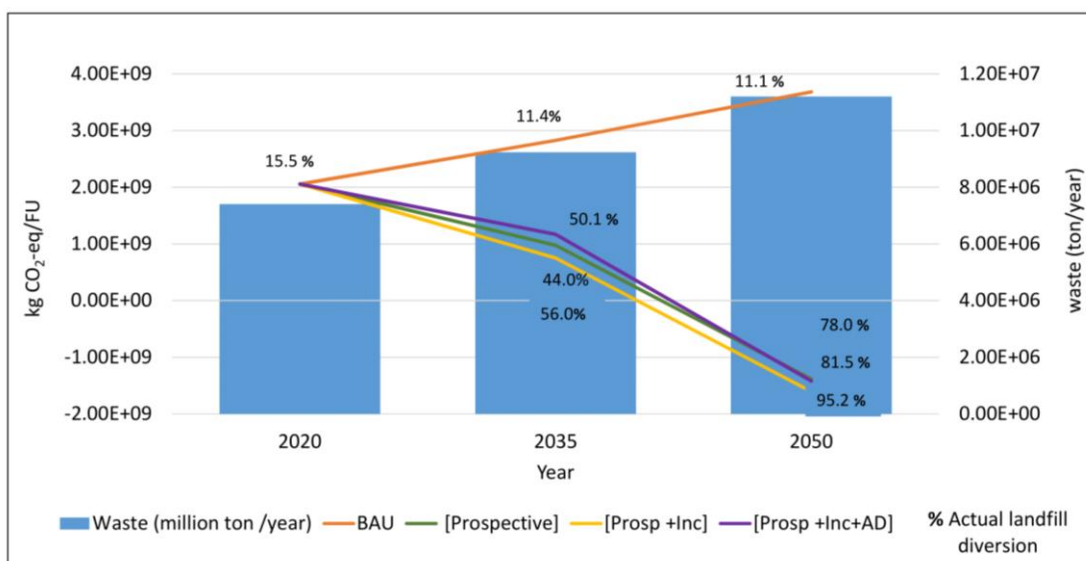


Figure 5.5 : CO₂ emission (kg CO₂-total waste), waste amount vs. landfill diversion- Istanbul

Impact categories with similar decoupling patterns to climate change (*human toxicity cancer, eutrophication terrestrial, terrestrial acidification and eutrophication marine*)

Saving coming from MRF, especially plastics (PE), is larger than the burden coming from WtE. Therefore, all improved scenarios other than BAU are decoupled for the human toxicity category. Collection, transportation and electricity consumption in transfer station dominate the burdens in all scenarios for eutrophication terrestrial category. In addition, all improved scenarios have additional burdens coming from engines in AD and electricity consumption MRFs. However, savings coming from energy recovery exceeding burdens in overall results, and they end up with the decoupling. Consequently, human toxicity cancer and eutrophication terrestrial categories show similar decoupling WM emission patterns for different reasons.

Collection and landfill gas emissions have largest burdens in all scenarios for terrestrial acidification category. Municipality and other improved scenarios have additional burden coming from biogas in AD facility. Hence, there is a slight coupling occurs in 2035. However, due to savings in paper, cardboard, aluminum, plastic recycling and PET recycling, WM emissions eventually decoupled in 2050 under improved scenarios. A similar decoupling pattern, a slight coupling in 2020-2035 and a decoupling in 2035-2050, has been observed in the eutrophication

marine category. Similarly, digestate from the AD facilities dominates burdens in all improved scenarios and the largest savings coming from recycling in MRFs.

Impact categories with opposite decoupling patterns (*human toxicity non-cancer, particulate matter, photochemical ozone formation, and eco-toxicity freshwater*)

All scenarios have coupling trend for human toxicity-non cancer category. Soil use of compost has major non-cancer impacts due to heavy metals on soils in the BAU scenario. In addition to composting, the improved scenarios increase human toxicity non-cancer burden, with increased incineration capacity due to mercury emissions. There is some savings coming from paper and aluminum recycling but also burdens such as steel recycling for human toxicity-non cancer category (Damgaard et. al., 2010). All scenarios show a coupling for the particulate matter category because the incineration plant causes particulate matter, mainly from ash, while the largest savings are from MRFs. The coupling occurs with increasing WtE and AD capacities in improved scenarios in photochemical ozone formation category. Because photochemical ozone formation is important for combustion activities (Hodge et al., 2016). NO_x and non-methane volatile compound (NMVOC) are the major contributors of this category because WtE flares lead to NMVOCs and biogas engines in AD facilities cause NO_x emissions. There is a coupling from 2020 to 2050 in the ecotoxicity freshwater category. Transportation, collection, electricity consumption in transportation station and MRFs are major sources of burdens. Additional burdens coming from digestate and biogas production in AD, landfilled bottom ash, and reprocessing of steel scrap from incineration plant. Secondary steel production is responsible for major heavy metal emissions in surface waters. Consequently, there is no WM technology in the enhanced scenarios that will result in decoupling in the ecotoxicity freshwater category.

5.3.2.3 Perturbation analyses and scenario analysis

The substitution ratio of PET, aluminum, cardboard, paper and plastic in MRFs, electricity recovery of incineration, substitution of aluminum from incineration plant, gas collection efficiency (leaking of gas) and CHP efficiency in landfill are most sensitive parameters in the model in the climate change category. The

impurities in the source separation. Impurity ratios are assumed same as in all future scenarios however, previous studies showed that impurities are increasing with higher source separation (Haupt et. al., 2018b). Also, with increasing source separation of packaging wastes, characterization and energy recovery potential of incineration facility changes. This is affecting the energy recovery from the incineration due to change of fossil fuel based wastes such as plastic (Figure 5.4). All scenarios (including 20-year and 100-year GWP) were preferred to WtE over landfill (Hodge et al., 2016).

5.3.2.4 Data quality assessment

DQA results show waste characterization and amount have higher quality data while WtE (AD and incineration) have basic quality data since they are based on literature sources. In addition, there are very few data available for landfill gas collection hence it was estimated. DQA results showed that there is need for data about performance analysis of WM facilities. Details DQA results were shown in Appendix in detail.

5.3.2.5 Uncertainty analysis

Uncertainty results show high variance, mostly due to limited data about gas collection efficiency. Also, substitution ratio change in MRFs has a significant effect on environmental savings. Although, energy recovery of incineration plant has high sensitivity value, the data have a low uncertainty hence effect is less on overall results (Figure 5.4).

Policy implications

The study evaluated the potential environmental impacts of municipal SWM development in Istanbul, considering reduced landfilling, increased incineration with energy recovery, and source separated food waste treated with AD, along with recycling of packaging wastes using LCA. The results indicate that even with optimal implementation of non-realistic recycling measures, the prospective municipality scenarios are unlikely to achieve the 60% and 100% landfill diversion targets by 2035 and 2050, respectively. There is a decrease in 2035 and 2050 landfilled waste, it is insufficient to achieve net zero to landfill or net savings targets

with regards to climate change. Although waste prevention is a priority in the waste management hierarchy, landfill diversion policies could serve as a transitional period toward the ultimate goal of ZW.

Source separated food waste policy: Istanbul's waste composition of is dominated by organic waste, posing challenges for technology adoption. To facilitate treatment and exploitation, new source separation and collection schemes are required in future SWM scenarios (Jaunich et al., 2019). Proper waste segregation can transform waste into value-added resources. Source separation of packaging waste has a positive effect on decoupling and climate impact, although uncertainties arise from ideal public behavior assumptions (Rousta K., 2018).

Source separated packaging waste policy: Global waste management data indicates that cities implementing waste sorting at the source have significantly reduced landfilling and increased recycling rates. While waste sorting is a cost-effective task, it requires considerable effort to address the social aspects (Rousta K., 2018). Comparing scenarios reveals that source separation of packaging waste has a notably positive impact on decoupling. Except for the BAU scenario, all future prospective scenarios assume ideal public behavior for source separation, introducing significant uncertainty in the calculations, particularly concerning the composition of the waste input to the treatment facilities. Previous SWM strategies in Türkiye have primarily focused on quantity-oriented target setting, overlooking the value of high-quality by-products derived from waste. Separating food waste or other moisture-rich waste before incineration, can enhance the energy recovery efficiency (Wang D., 2020). For example, diverting all plastic waste to MRF reduce the fossil carbon input of incineration plant in the prospective scenarios. This highlights the need to shift waste management technology thinking from simply considering which technology is used to emphasizing the amount of material or energy substitution (Andreasi Bassi et al., 2017). Overall, our results align with previous studies, emphasizing the significance of waste recycling capacity and energy recovery, particularly through incineration, as key points for waste management decoupling (Magazzino et al, 2020).

WtE facility policy: Despite data uncertainty and scenario limitations, the study demonstrates that the shift away from landfilling towards incineration with energy recovery is beneficial with respect to climate change. Additionally, AD may further reduce SWM's impact. Phasing out landfill is recommended to bridge the gap between waste management sector emissions and sustainability policy goals, as also observed by Magazzino (2021). SWM strategies need to adapt to societal changes, and incineration is one of the few mature existing methods, besides landfilling, for processing mixed municipal waste (Astrup et al., 2015). Moreover, WtE is generally preferable to landfilling in most cases (Eriksson, 2000). Hence, the prospective future scenarios in this study are based on landfill diversion using WtE technologies.

The scenarios in this study were designed by considering technological feasibility and environmental impacts. However, other indicators such as cost play a key role for decision makers. While increasing WtE and MRF capacity is preferable from an environmental perspective, their cost implications should be explored. During the decoupling analysis, we did not differentiate between relative or absolute decoupling, as both emissions and waste amounts were modelled rather than measured. Future extensions of this study should incorporate reliable economic data, and GDP data may be used instead of the amount of assumed waste in the waste management emissions decoupling analysis. Future studies should focus on evaluating both environmental and economic consequences to identify the most eco-efficient scenario. These evaluations should consider future SWM policies, taxes and campaigns as well. Additionally, waste prevention scenarios' effects on overall emissions and the potential of emerging technologies like CCS should be addressed before constructing new incineration plants. However, discussions on the actual feasibility of new incineration plants continue, as they may not fully serve raw material protection or substitution, and their construction costs are high compared to landfills.

Identifying optimal SWM strategies requires coordination among stakeholders at every level. (Jaunich et al., 2019). The main aim of SWM policies is to limit environmental burdens by leveraging critical resources embedded in outdated products (D'Adamo et al., 2022). Achieving coordination among different government levels and political groups in the environment sector can be particularly

complex (Levis et al., 2014). Scenarios that integrate technologically feasible solutions, even if they may be deemed socially or politically unfavourable, are crucial for supporting and SWM's decision-making within applicable time limits (Jaunich et al., 2019). SWM systems must proactively adapt to changing policy requirements, waste composition, and evolving energy systems to sustainably manage future solid waste. Since WM systems involve long-lived infrastructure, decision-makers should proactively plan for changing the waste compositions, policy requirements, and evolving energy systems to achieve cost-effective and sustainable solid waste management (Levis et al., 2014).

5.4 Conclusion

The objective of this study is to assess the decoupling of WM emissions and landfill diversion potential with current policies. The other purposes of this study are to provide methodological insights into the modelling of a complex WM system using MFA-LCA and discuss policy implications to support WM decision-making. The MFA results showed that even with high capacity WtE technology scenarios, 100% landfill diversion is not possible due to impurities in the source separated waste and efficiency of the WM facilities. Although waste prevention has priority in the WM hierarchy, landfill diversion policies could be a transition period until ultimate ZW goal. The results of the LCA show that the current development of landfill towards new WtE (incineration and AD with energy recovery) is beneficial in terms of climate change. This study contributes the literature by giving quantitative relations between WM emissions and goal originated WM strategies. The approach introduced in this study may also help to understand WM effects on global policy targets like mitigating climate change. Assessing not only one impact factor but evaluation of other impact categories helps to comprehensive assessment of WM results on environment. This study initial attempt for embracing LCA as decoupling analysis of WM system but assessing environmental impacts after collecting high quality data from WtE facilities is suggested. Economic, social and technical aspects should be evaluated as well in the future.

6. ECO-EFFICIENCY ANALYSIS OF WASTE MANAGEMENT SYSTEMS BY USING A DECISION-MAKING TOOL

The methodological framework is described in the next section (Figure 6.1). The proposed methodology involves seven steps (I-VII) which form a open loop: (I) Problem Settling: identification research questions (RQs), (II) Food Waste Management Scenarios: that consists of identifying and mapping the WM systems, (III) Eco-efficiency Tool: determining the general environment, economy, social and technical indicators related to WM and their relationship to the SDGs. (IV) Case Spesific Indicator Selection: environment, economy, social, technical indicator selection for fw management in Istanbul (V) Data Inventory for MCDM: providing data by using LCA, cost calculations, questionnaire, literature (primary and secondary sources), (VII) MCDM application: by using AHP-I and TOPSIS methods, (VII) Results- Comments- Conclusion: Identify future needs scenarios, policy strategies, presenting potential polices and recommendations while reflecting on the process.

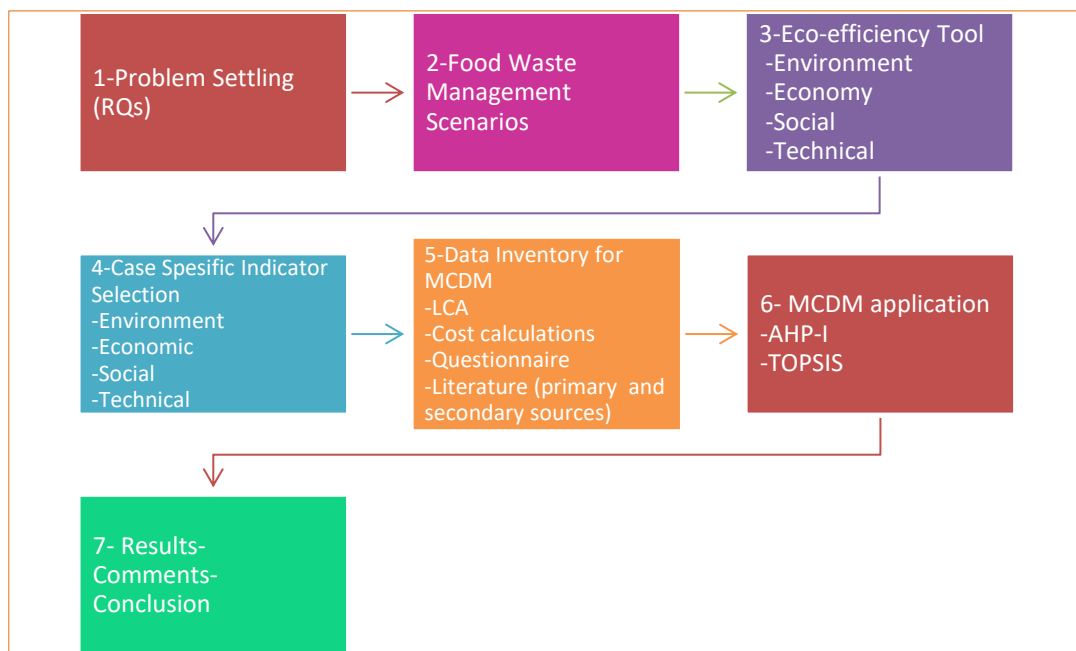


Figure 6.1 : Methodology

A variety of waste management options were assessed using AHP and TOPSIS methods to select the most suitable food waste management technology in Istanbul. Istanbul is the most populated city in the country and has a population of around 16 million. As a most developed city, Istanbul has a momentum on new strategies and technology investment for WM.

6.1 The Analytical Network Process (AHP)

The AHP method can be used to assist decision-makers to calculate the weight for each criterion by using pair-wise comparison judgments. The AHP can build a pairwise comparison matrix to establish a decision.

Every element of matrix A is normalised by dividing each element in a column by the sum of the elements in the same column to create a normalised pairwise comparison matrix A_{norm} . A_{norm} is the normalised matrix of A(1), where $A(x_{ij})$ is given by Eq. (6.1). $A_{norm}(a_{ij})$ is expressed as follows:

$$a_{ij} = \frac{x_{ij}}{\sum_{i=1}^n x_{ij}} \quad (6.1)$$

Table 6.1 : Nine scales of pairwise comparisons

Intensity of Importance	Definition	Explanation
1	Equal importance	Two activities contribute equally to the objective
3	Weak importance of one over another	Experience and judgment slightly favour one activity over another
5	Essential or strong importance	Experience and judgment strongly favour one activity over another
7	Demonstrated importance	Activity is strongly favoured and its dominance is demonstrated in practice
9	Absolute importance	The evidence favouring one activity over another is of the highest possible order of affirmation

AHP pairwise comparison uses mathematical calculations to convert judgments to provide weights for all criteria. After obtaining the responses on the pairwise comparisons, a reciprocal matrix is created from the pairwise comparisons. The weights of decision factor I can be calculated as Eq. (6.2): where n is the number of the compared elements.

$$w_i = \frac{\sum_{j=1}^n a_{ij}}{n} \text{ and } \sum_{j=1}^n w_i = 1 \quad (6.2)$$

Calculation of Consistency Ratio (CR) which expresses the internal consistency of judgments, is calculated. The following terms are defined to develop a quantitative measure of the degree of inconsistency within a pairwise comparison matrix. The consistency index (CI) is calculated with Eq. (6.3):

n	2	3	4	5	6	7	8	9	10
RI	0	0.58	0.9	1.12	1.24	1.32	1.41	1.45	1.51

$$CI = \frac{\lambda_{\max} - n}{n - 1} \quad (6.3)$$

The random index (RI) CI measures the degree of inconsistency. RI is the corresponding measure of the degree of inconsistency of a pairwise comparison matrix. CR is defined in Eq. (6.4):

$$CR = \frac{CI}{RI} \quad (6.4)$$

A value of CR less than 0.1 is generally acceptable; otherwise the pair-wise comparisons should be revised to reduce incoherence.

6.2 Technique for Order Preference by Similarity to Ideal Solution (TOPSIS)

TOPSIS based upon the concept that the chosen alternative should have the shortest distance to the positive ideal solution (S^+) and the longest distance from the negative ideal solution (S^-).

1. Identify the criteria and sub-criteria that have the largest impact on the problem.
2. Construct a hierarchy decision model for the problem.
3. Determine the comparison matrix for each level by using AHP technique to obtain the local weight of each criterion and subcriterion.
4. Determine the global weight by normalizing the local weight.
5. Use the TOPSIS technique to assess the alternatives where the most appropriate one can be easily selected.
6. Select the best alternative.

Calculate Normalised Matrix;

Create an evaluation matrix consisting of m alternatives and n criteria, with the intersection of each alternative and criteria given. Eq. (6.5):

$$\overline{X}_{ij} = \frac{X_{ij}}{\sqrt{\sum_{i=1}^n X_{ij}^2}} \quad (6.5)$$

Calculate weighted Normalised Matrix;

Normalized weighted. With the weight $w_j = (w_1, w_2, w_3, \dots, W_n)$, where w_j is the weight of the criteria for all j and $\sum_j = 1$ $w_j = 1$, The normalization of weight matrix V, i

$$V_{ij} = \overline{X}_{ij} \times W_j \quad (6.6)$$

Calculate the ideal best and ideal worst value and the Euclidean distance from the ideal best; Eq. (6.6):

$$S_i^+ = \left[\sum_{j=1}^m (V_{ij} - V_j^+)^2 \right]^{0.5} \quad S_i^- = \left[\sum_{j=1}^m (V_{ij} - V_j^-)^2 \right]^{0.5} \quad (6.7)$$

Calculate Performance Score; Eq. (6.8):

$$P_i = \frac{S_i^-}{S_i^+ + S_i^-} \quad (6.8)$$

6.3 Results

In a final step feedback from stakeholders on the proposed set was received through a survey. A total of 50 interveners were taken into account and 34% responded. The pairwise comparison was obtained by seeking the views of 17 experts.

Table 6.2 : Questionnaire results

Sector	
Public (Ministry, Municipality)	5
Private Sector	6
University	3
Semi Private Company	3
TOTAL	17
Experience	
1-5 year	6
6-10 year	3
11-15 year	3
>15 year	5
TOTAL	17
Population of the expert city	
100 000-750 000	1
750 000- 1 500 000	3
1 500 000- 5 000 000	1
> 5 000 000	12
TOTAL	17

Additionally, the selection of criteria and sub-criteria is derived from the results of AHP analysis. which demonstrates that the primary factors that determine decisions in a waste management system.

	Sub-criteria	AD	Incineration	Landfill	Composting
ENVIRONMENT	Circularity	42.6%	0	0	36.1%
	Climate Change	(-126 kg CO2-eq/ ton FW)	(-192 kg CO2-eq/ton FW)	(924 kg CO2-eq/ton FW).	(119 kg CO2-eq/ton FW)
	Particulate Matter	(-108.8 kg PM2.5-eq/ ton FW)	(-106.5 kg PM2.5-eq/ ton)	(-61.1 kg PM2.5-eq/ ton)	(31.3 kg PM2.5-eq/ ton)
	Eutrophication Terrestrial	(944.8 kg N eq.)	(-676.9 kg N eq.)	(689.6 kg N eq.)	(2254.8 kg N eq.)
	Eutrophication Marine	(919.2 kg N eq.)	(-65.2 kg N eq.)	(113.1 kg N eq.)	(662.8 kg N eq.)
ECONOMIC	Investment Cost	71 €	105 €	63 €	58 €
	Operation Cost	55 €	72 €	40 €	40 €
	Public Fee	3 €	20 €	13 €	7 €
	Revenue	- 32 €	- 36 €	- 5 €	- 6 €
SOCIAL	Partnership Potential	1	3	4	2
	Job Creation	0.05 person/ton waste*day	0.25 person/ton waste*day	0.1012651 person/ton waste*day	0.16 person/ton waste*day
	Urban Space Requirement	0.00608828 m2/ton	0.0054063 m2/ton	0.0049122 m2/ton	0.0060761 m2/ton
	Public Acceptability	1	3	4	2
	Public Health	1	3	4	2
TECHNICAL	Flexibility	1	4	3	2
	Reliability	2	4	3	1
	Qualified Employee	0.055 person/ton waste*day	0.055 person/ton waste*day	0.050 person/ton waste*day	0.080 person/ton waste*day

Figure 6.2 : MCDM inventory

Table 6.3 : Criteria weight and global weight calculation with AHP method.

	Criteria	criteria weight	
All	Environment	0.58	
	Economic	0.29	
	Social	0.04	
	Technical	0.09	
	Total	1.00	
Environment	Sub-Criteria	criteria weight	global weight
	Circularity	0.4923	0.283
	Climate Change	0.2527	0.145
	Particulate Matter	0.1427	0.082
	Eutrophication Terrestrial	0.0342	0.020
	Eutrophication Marine	0.0781	0.045
Economic	Sub-Criteria	criteria weight	global weight
	Investment Cost	0.110	0.032
	Operation Cost	0.539	0.157
	Public Fee	0.056	0.016
	Revenue	0.295	0.086
Social	Sub-Criteria	criteria weight	global weight
	Partnership Potential	0.12	0.0050
	Job Creation	0.02	0.0007
	Urban Space	0.06	0.0028
	Requirement	0.28	0.0118
	Public Acceptability	0.53	0.0224
Technical	Sub-Criteria	criteria weight	global weight
	Flexibility	0.07	0.007
	Reliability	0.64	0.058
	Qualified Employee	0.28	0.026

In conclusion, the organic fraction of MSW can be processed by setting up an anaerobic digestion facility in Istanbul (food, park and bio waste) as it is the best option in terms of environmental, economic, technical and social aspect.

Table 6.4 : TOPSIS method results

Scenario	Si+	Si-	Pi	Rank
AD	0.0046	0.271719	0.98350	1
Incineration	0.2218	0.188558	0.459457	3
Landfill	0.2686	0.086203	0.242931	4
Composting	0.1099	0.221085	0.690712	2

The acceptance of specific technologies hinges on the good engagement between local authorities, industry, and regulatory bodies with stakeholders. MCDM principles should be used by decision-makers to address a WM management problem. A robust strategy is essential for a sustainable WM system.

7. CONCLUSIONS AND RECOMMENDATIONS

The purpose of this dissertation was to evaluate the environmental and economic aspects of WM in the context of an increasingly applied ZW approach. To achieve this goal, this thesis develops new adapted tools capable of tackling the complexity of WM and anticipating consequences at the system level. As far as the author is aware, this is the first study to provide an extensive environmental and economic sustainability assessment of municipal solid waste management through ZW goal in multiple scales (Istanbul and Ayazağa Campus). The conclusions of this work will be of interest to WM sector experts, local authorities, policymakers, researchers and the general public. The principal results and conclusions for the research questions addressed in this thesis are presented in the following.

Q1. What are the data and information gaps and how these gaps could be filled to strengthen the WM system?

WM systems must proactively adapt to changing policy requirements, waste composition, and an evolving energy system to sustainably manage future solid waste. WM systems consist of long-lived infrastructure, decision-makers need to proactively plan for changing the composition of waste and policy requirements and a changing the energy system for cost-effective and sustainable management of future solid waste. DQA results show waste characterization and amount have higher quality data while WtE (AD and incineration) have basic quality data since they are based on literature sources. In addition, there are very few data available for landfill gas collection hence they had to be estimated. DQA results showed that there is need for data about performance analysis of WM facilities especially in Türkiye. Uncertainty results show high variance, mostly due to limited data about gas collection efficiency. Also, substitution ratio change in MRFs has a significant effect on environmental savings. Although, energy recovery of incineration plant has high sensitivity value, the data have a low uncertainty hence effect is less on overall results.

Data/information gaps could be filled to strengthen the WM system by:

- ❖ Develop frameworks, benchmarks, baselines and objectives to continue moving forward.
- ❖ Identify common areas where current research will not meet future demands or where research needs to be pursued to meet future needs.
- ❖ Determine variances between scenarios and progress.
- ❖ Enhance the formulation of policies according to their relevance and impact on the governance model.
- ❖ Enhance reliability of the data by combination of qualitative and quantitative methods.

Q2. Whether the ZW to landfill goal is possible and landfill diversion always the best option in terms of environmental impacts and circularity?

The net environmental savings and circularity increase with diversion rates but there is no direct relation. Even though landfill diversion increases, with WtE, thanks to incineration plant, circularity ratio remains same. As Slorach, (2019) mentioned circular does not always mean environmentally sustainable. Their study is showed that composting is the worst option in terms of LCA. Nevertheless, it is preferable to incineration and landfill within the waste hierarchy (Slorach et al, 2020). Details of environmental impacts, landfill diversion and the circularity relationship is significant topic to be addressed in future studies. Consequently, landfill diversion does not always serve to improve circularity or directly address sustainable consumption and public attitudes change towards ZW goal.

In Addition, which technology is used to substitute landfill is affecting net impacts and circularity (Andreasi Bassi, 2017). The decarbonizing ranking results revealed that all scenarios other than the landfill scenario achieve the zero-waste-to-landfill target, albeit the results are very different in terms of nutrient circularity, carbon emissions, and costs. This indicates that a variety of aspects should be considered before selecting technologies for goal-originated WM systems.

Q3. Which environmental changes are expected with the ZW management?

The potential environmental impacts of the development in municipal WM in Istanbul, with less landfilling, more incineration with energy recovery, and source separated food waste treated in anaerobic digestion, and recycling of packaging wastes were assessed through an LCA. Results show that in the municipality scenario case is unlikely to reach 60% and 100% landfill diversion targets in 2035 and 2050 respectively even if non-realistic recycling measures are implemented optimally. Current policy could not reach the net saving in 2035 due to high landfill burden. There is a decrease in 2035 landfilled waste amount, which leads to a decoupling of WM emissions, but that is not enough to meet net zero or net savings targets. The impacts of landfill are more dramatic from a shorter time point of view (GWP-20 years), due to the large releases of methane. In spite of the uncertainty and limitations of data and scenarios used for this evaluation, the results show that the current development away from landfilling and toward incineration with energy recovery is beneficial with respect to climate change (Zhao et al., 2022). AD may further reduce WM's impact. Phasing out landfill recommended before to close gap between WM sector emissions and sustainability policy goals (Magazzino, et. al., 2021). WM strategies are always under change due to changes in the society. Even if direct landfilling of waste can be avoided in the future, ash and slag from thermal treatment will end up in the landfill.

Human Toxicity cancer, Eutrophication Terrestrial, Terrestrial Acidification and Eutrophication Marine have similar decoupling patterns to Climate Change. Human Toxicity non-cancer, Particulate Matter, Photochemical Ozone Formation, and Ecotoxicity Freshwater have opposite decoupling patterns. The major cause of particulate matter is incineration. Incineration also has the greatest formation of photochemical oxidants because of direct NMVOC emissions from treated flue gases. The primary terrestrial acidification factor is the volatilization of ammonia during composting and digestate application. Similarly, digestate from the AD facilities dominates burdens in eutrophication marine category and the largest savings coming from recycling in MRFs.

Q4.What criteria can assess eco-efficiency of the ZW management system?

The scenarios in the LCA of WM systems were designed by considering technological feasibility and environmental impacts. However, there are also other indicators which play a key role for decision makers. One of the biggest obstacles of sustainability targets like ZW is the need of engagement of all stakeholders and environmental results are not enough for decision making. Future studies should focus on both environmental and economic consequences to find out most eco-efficient scenario.

All future scenarios for Istanbul;

- ❖ [Prosp]₂₀₃₅-124€/ton, [Prosp +inc]₂₀₃₅-127€/ton, [Prosp +inc-AD]₂₀₃₅-129€/ton, have lower net cost (€/ton) than the business-as-usual 130€/ton for 2035. This is primarily due to the reduction in landfill cost and revenues coming from AD and incineration plant.
- ❖ [Prosp]₂₀₅₀- 99€/ton, [Prosp+inc]₂₀₅₀- 122€/ton, [Prosp +inc-AD]₂₀₅₀-107€/ton, have lower net cost (€/ton) than the BAU₂₀₅₀ 160€/ton for 2050. These mainly consist of waste from municipal waste recycling, energy recovery from the AD and incineration facilities.

Source separated campus scale FW management cost;

- ❖ Landfilling and composting has the lowest operation costs (40€/ton FW) however landfill has lower net cost (80€/ton FW) compared to composting (92€/ton FW) due to additional revenue coming from landfill gas energy recovery in 2020.
- ❖ AD had the second highest net cost (94 €/ton FW) in 2020, due to high OPEX (operating expenses) compared to landfill and composting. AD has lower net cost than (138 €/ton FW) composting (139 €/ton FW) in 2050 due to high revenue coming from energy recovery.
- ❖ The combined CAPEX and OPEX per tonne of FW are highest for incineration in both 2020 (112 €/ton FW) and 2050 (183 €/ton FW).

Identifying optimal WM strategies requires coordination of every level of stakeholders. WM systems must proactively adapt to changing policy requirements, waste composition, and an evolving energy system to sustainably manage future solid waste as WM systems consist of long-lived infrastructure, decision-makers need to

proactively plan for changing the composition of waste and policy requirements and an changing the energy system for cost-effective and sustainable management of future solid waste. The indicators have been identified based on the following requirements: meaningful, clear in content, appropriate in scale, robust and reproducible, comparable describe and understandable.

The environmental indicators included in the study were those typically considered in LCA studies (Hauschild et. al., 2013). For the MCDM study; circularity, climate change, particulate matter, eutrophication terrestrial and eutrophication marine were selected as sub-indicators. Capital/investment costs, operation cost, public fee and revenue were considered for the economic sustainability assessment. Social indicators; considered within the decision-support framework wre classified into five sub-categories namely; partnership potential, job creation, urban space requirement, public acceptability and public health. Finally, technical indicators; were classified into three categories as, flexibility, reliability and qualified employee.

On the basis of the knowledge and experience acquired during the PhD thesis, the following recommendations were made:

- ❖ New WM systems or decisions are expected to prioritise waste prevention.
- ❖ The environmental and economic sustainability has been evaluated for both the current (2020) and potential future (2035 and 2050) WM of Istanbul. The calculations results of the case study of Istanbul revealed that an optimal MSW management system that achieves ZW to landfill target requires a more intensive use of incineration. Hence, incineration should have a higher priority than landfill.
- ❖ However, switching to a non-fossil electricity mix in the country may reduce the climate benefits of incineration. As a result, AD has been an attractive option to minimize the impacts of climate change, especially on FW.
- ❖ The composition of Istanbul's waste is largely dominated by organic waste, which presents a challenge to technology adoption. In addition, it is important to know that, future WM scenarios in this study requires new systems for source separation and collection schemes.
- ❖ The government should consider imposing a penalty for landfilling unprocessed organic waste.

- ❖ Identifying compost and digestate market substitution ratio national data in the future is essential.
- ❖ IMM's objectives to recycle 100% of plastic waste by 2050 appear unlikely to be achieved without the commitment of all stakeholders, the public sector and the WM sector. Related master plans and policies should be developed.
- ❖ While the work focuses on Istanbul (and Ayazağa campus for smaller scale) conditions, the broader findings could be apply to other cities as well.
- ❖ For the MSW management high cost could be a significant barrier. There is also ongoing debate on diversion policies and CE.
- ❖ The results of this thesis show that landfill diversion is not always serving circularity. There is need for a deeper research for CE and incineration relationship.
- ❖ The necessary changes in the legislative and economic contexts to shift from linear to circular systems should be discussed from a multi-stakeholder perspective.
- ❖ Assessment of WM across the whole supply chain to understand its stress on raw material decoupling and circularity is significant topic to be addressed.
- ❖ This study has focused on gathering new data at the MFA level, but have used literature data for substance level flows such as carbon, nitrogen effects. Future studies could focus on (if they have available data) substance flow analysis (SFA) to understand WM effects on system level such as carbon cycle or the nitrogen cycle.
- ❖ Social indicators need to be considered such as; how public behavior could change and how these changes could be monitored in line with CE (Dilkes-Hoffman et al., 2019).
- ❖ The focus of this study is exclusively on midpoint categories, as suggested by Hauschild (2013). Future studies can incorporate endpoint categories if they have high-quality data.
- ❖ This study involves forecasting data with uncertainty, including waste amount, characterization, and cost. The use of a data system and monitoring can help decrease these uncertainties in future.

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APPENDICES

APPENDIX A: Waste Characterization

APPENDIX B: Environmental Impact Categories

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APPENDIX U: Food Waste Management in University Campus

APPENDIX V: Waste Composition (Urban Scale)

APPENDIX W: Waste Amount Forecasting Asian and European Side of Istanbul (MFA results)

APPENDIX X: Waste Management Facilities in Istanbul

APPENDIX Y: Contribution Analysis of Climate Change Impact Category

APPENDIX Z: Economic Inventory and Calculations



APPENDIX A: WASTE CHARACTERIZATION



Figure A.1 : Containers before the ZW (No labelling for waste separation at source)



Figure A.2 : Containers after implementing the ZW system



1-Faculty of Civil Engineering



2-Student Affairs Building



3-Girls Dormitory



4-Student's Dining-hall



4-Staff Dining-hall

Figure A.3 : Pilot areas in the study



Figure A.4 : Hand sorting during the waste characterization



Figure A.5 : Sorted waste following the hand sorting



APPENDIX B: ENVIRONMENTAL IMPACT CATEGORIES

Table B.1 : Environmental impact categories used in the impact assessment (JRC, 2011; Hauschild et al., 2013).

Impact category	Recommended default LCIA method
Climate change	Baseline model of 100 years of the IPCC (IPCC 2021)*
Stratospheric ozone depletion	Steady-state ODPs 1999 as in WMO assessment (WMO 1999)
Human toxicity, carcinogenic	USEtox model (Rosenbaum et al. 2008)
Human toxicity, non- carcinogenic	USEtox model (Rosenbaum et al. 2008)
Particulate matter/Respiratory inorganics	Humbert (Humbert 2009)
Photochemical ozone formation	ReCiPe (Goedkoop et al. 2008)
Terrestrial acidification	Accumulated Exceedance (Seppälä et al. 2006)
Terrestrial eutrophication	Accumulated Exceedance (Seppälä et al. 2006)
Freshwater eutrophication	EUTREND model as implemented in ReCiPe (Goedkoop et al. 2008)
Marine eutrophication	EUTREND model as implemented in ReCiPe (Goedkoop et al. 2008)
Freshwater ecotoxicity	USEtox model (Rosenbaum et al. 2008)

*most recent method is used.



APPENDIX C: WASTE AMOUNT AND COMPOSITION

Waste amount and characterization studies were carried out for four different units in the campus.

Table C.1 : Waste amount in ITU Ayazağa Campus-annual average for 2019.

(tonne/year) Waste sub categories	Academic Blue	Grey	TOTAL	Residental Blue	Grey	TOTAL	Administrative Blue	Grey	TOTAL	Cafeteria Blue	Grey	TOTAL	Total Blue	Grey	TOTAL
Aluminium	2.4	1.9	4.3	0.7	0.9	1.7	0.2	0.0	0.2	0.0	0.0	0.0	3.3	2.8	6.1
Cardboard	75.3	37.9	113.1	22.5	18.8	41.4	45.6	0.0	45.6	29.3	0.0	29.3	172.7	56.7	229.4
Ceramic	0.8	0.5	1.2	0.0	0.4	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.9	1.6
Colorful glass	4.1	6.4	10.5	4.3	9.1	13.5	2.3	0.0	2.3	5.5	0.0	5.5	16.2	15.5	31.7
e-waste	0.6	0.8	1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.8	1.4
Glass	2.4	14.1	16.5	5.2	6.3	11.5	0.9	0.0	0.9	17.2	0.0	17.2	25.7	20.3	46.1
Inert waste (C&D etc)	10.5	17.4	27.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	10.5	17.4	27.9
Organic waste (cooked)	1.4	106.8	108.3	22.6	20.4	43.0	0.0	75.2	75.2	0.0	78.1	78.1	24.0	280.5	304.5
Organic waste(raw)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	425.4	425.4	0.0	425.4	425.4
Other	14.0	85.5	99.5	59.9	66.0	125.8	0.0	7.9	7.9	0.0	0.0	0.0	73.9	159.4	233.2
Other Metal	0.0	2.6	2.6	0.8	0.3	1.1	0.0	0.0	0.0	8.3	0.2	8.5	9.0	3.1	12.1
Other Plastic	15.7	25.6	41.3	12.8	11.2	24.0	3.7	0.0	3.7	7.5	7.0	14.5	39.8	43.7	83.5
Paper	88.6	21.0	109.6	1.3	1.7	3.0	41.9	0.0	41.9	0.0	0.0	0.0	131.7	22.7	154.4
Pet	5.8	13.2	19.0	1.3	2.0	3.2	0.0	0.0	0.0	0.0	0.0	0.0	7.0	15.1	22.2
Plastic Bag	5.7	9.9	15.7	12.3	13.2	25.5	2.0	0.0	2.0	1.6	0.4	2.0	21.6	23.6	45.1
Tetrapak	0.4	1.6	2.0	3.2	2.0	5.1	0.2	0.0	0.2	2.7	0.0	2.7	6.4	3.6	10.0
Textile	0.1	7.0	7.1	4.7	3.7	8.3	0.0	0.0	0.0	0.0	0.0	0.0	4.8	10.7	15.4
Toilet paper/paper															
towel	22.9	73.6	96.4	5.1	0.0	5.1	0.0	24.7	24.7	0.3	7.2	7.5	28.3	105.4	133.7
Total	250.6	425.6	676.2	156.5	155.9	312.4	96.7	107.7	204.4	72.4	518.2	590.6	576.2	1207.4	1783.6

Future waste management scenarios were planned by using questionnaire study results in the campus. Some waste sub-categories in the cafeteria and administrative units have lower impurities in the blue bins hence questionnaire results only applied the high impurity waste sub-categories for this two units.

Table C.2 : Waste amount in ITU Ayazağa Campus- change in sorption after ZW according to questionnaire

(tonne/year) Waste sub categories	Blue	Academic Grey	TOTAL	Residential Blue	Grey	TOTAL	Administrative Blue	Grey	TOTAL	Cafeteria Blue	Grey	TOTAL	Total Blue	Grey	TOTAL
Aluminium	3.9	0.4	4.3	1.5	0.2	1.7	0.2	0.0	0.2	0.0	0.0	0.0	5.5	0.6	6.1
Cardboard	101.8	11.3	113.1	37.2	4.1	41.4	41.1	4.6	45.6	29.3	0.0	29.3	209.4	20.0	229.4
Ceramic	0.1	1.1	1.2	0.0	0.4	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.2	1.5	1.6
Colorful glass	9.4	1.1	10.5	12.1	1.4	13.5	2.1	0.2	2.3	5.0	0.6	5.5	28.5	3.2	31.7
e-waste	0.1	1.3	1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	1.3	1.4
Glass	14.8	1.7	16.5	10.4	1.2	11.5	0.8	0.1	0.9	15.5	1.7	17.2	41.5	4.6	46.1
Inert waste (C&D etc)	2.8	25.1	27.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.8	25.1	27.9
Organic waste (cooked)	10.8	97.4	108.3	4.3	38.7	43.0	7.5	67.7	75.2	0.0	78.1	78.1	22.6	281.8	304.5
Organic waste(raw)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	425.4	425.4	0.0	425.4	425.4
Other	10.0	89.6	99.5	12.6	113.2	125.8	0.8	7.1	7.9	0.0	0.0	0.0	23.3	209.9	233.2
Other Metal	2.3	0.3	2.6	1.0	0.1	1.1	0.0	0.0	0.0	8.3	0.2	8.5	11.5	0.6	12.1
Other Plastic	37.2	4.1	41.3	21.6	2.4	24.0	3.4	0.4	3.7	7.5	7.0	14.5	69.6	13.9	83.5
Paper	98.6	11.0	109.6	2.7	0.3	3.0	37.7	4.2	41.9	0.0	0.0	0.0	139.0	15.4	154.4
Pet	17.1	1.9	19.0	2.9	0.3	3.2	0.0	0.0	0.0	0.0	0.0	0.0	20.0	2.2	22.2
Plastic Bag	14.1	1.6	15.7	22.9	2.6	25.5	1.8	0.2	2.0	1.6	0.4	2.0	40.4	4.7	45.1
Tetrapak	1.8	0.2	2.0	4.6	0.5	5.1	0.2	0.0	0.2	2.7	0.0	2.7	9.3	0.7	10.0
Textile	0.7	6.4	7.1	0.8	7.5	8.3	0.0	0.0	0.0	0.0	0.0	0.0	1.5	13.9	15.4
Toilet paper/paper towel	9.6	86.8	96.4	0.5	4.6	5.1	2.5	22.2	24.7	0.3	7.2	7.5	12.9	120.7	133.7
Total	335.2	341.1	676.2	135.0	177.4	312.4	97.8	106.6	204.4	64.4	526.2	590.6	632.4	1151.2	1783.6

Table C.3 : Waste Biological and physico-chemical properties of waste materials. LHV: lower heating value, VS: volatile solids, BMP: biochemical methane potential, d.w.: dry weight (EASETECH database; Ribier et al., 2009)

Fraction name	Water (%)	TS (%)	VS (%TS)	Ash (%TS)	Energy (MJ/kgTS)	C bio (%TS)	C bio and (%TS)	C fossil (%TS)	Ca (%TS)	Cl (%TS)	F (%TS)	H (%TS)	K (%TS)	N (%TS)
Aluminium	8.3	91.7	0	100	0	0	0	0	0.00356	0	0	0	0.0162	0
Cardboard	13.1	86.9	85.1	14.9	16.97	42.7	14.2	0.431	3.46	0.13	0.01	5.8	0.079	0.3
Ceramics	2.3	97.7	1.00E-30	100	0	0	0	0	0.345	0	0	0	1.86	0
Colorful glass	3.43	96.57	1.00E-30	100	0	0	0	0	6.9	0	0	0	0.775	0
e-waste	8.9	91.1	14.2	85.8	0.57	4.35	0	4.35	0.0876	1.58	0.01	1.1	3.97	0.1
Glass	11.98	88.02	1.00E-30	100	0	0	0	0	6.77	0	0	0	0.365	0
Inert waste (C&D etc)	0	100	1.00E-30	100	0	0	0	0	5.01	1.07	0	0	1.7	0.103
Organic waste (cooked)	57.1	42.9	91.3	8.7	24.55	55.4	36	1.13	4.09	1.63	0.01	7.9	0.533	7
Organic waste(raw)	77	23	94.8	5.2	18.3	47.5	42.3	0.239	0.555	0.56	0.01	6.6	1.27	1.9
Other	45.5	54.5	91.7	8.3	22.22	49.8	3.9	5.53	0.962	0.14	0.01	8	0.141	0.9
Other Metal	13.18	86.82	1.00E-30	100	0	0	0	0	0.0244	0	0	0	0.0532	0
Other Plastic	6.8	93.2	74.8	25.2	27.59	14.85	0	44.55	0.761	2.76	0.01	6.9	0.0278	2.2
Paper	7.4	92.6	82.61	17.39	13.15	38.1	21.9	0.192	4.39	0.06	0.01	5	0.073	0.2
Pet	10.5	89.5	93.9	6.1	36.54	0.386	0	76.8	0.314	0.17	0.01	11.3	0.0372	0.1
Plastic Bag	7.1	92.9	94.5	5.5	31.96	0.355	0	70.6	1.09	4.68	0.01	9.7	0.121	0.5
Tetrapak	16.1	83.9	90.4	9.6	23.76	50.6	8.7	1.03	0.782	0.11	0.01	7.7	0.0571	0.2
Textiles	6	94	96.4	3.6	19.8	39.1	0	13	0.44	0.35	0.01	6	0.0706	3.2
Toilet paper/paper towel	46.9	53.1	97.3	2.7	16.97	44.7	25.2	0.452	0.393	0.26	0.01	6.3	0.151	0.8
Yard Waste	56	44	45	55	8.98	17.11	8.6	0.488	0.86	0.14	0.005	3.1	1.25	1.2

Table C.4 : Biological and physico-chemical properties of waste materials. LHV: lower heating value, VS: volatile solids, BMP: biochemical methane potential, d.w.: dry weight-**cont.**(EASETECH database; Riber et. al, 2009)

Fraction name	Na (%TS)	O (%TS)	P (%TS)	S (%TS)	Al (%TS)	As (%TS)	Cr (%TS)	Cu (%TS)	Fe (%TS)	Mg (%TS)	Mn (%TS)	Ni (%TS)	Pb (%TS)	Zn (%TS)
Aluminium	0.0165	0	0.011	0.003	62.8	0.000913	0.0109	0.0892	34.5	1.18	0.514	0.00612	0.00184	0.0101
Cardboard	0.273	35.8	0.0347	0.126	2.15	2.28E-05	0.000528	0.00338	0.0467	0.109	0.00223	0.000323	0.000456	0
Ceramics	0.576	0	0.0152	0.0104	2.21	0.000871	0.0104	0.00224	0.726	0.052	0.0145	0.0322	0.103	0.261
Colorful glass	2.49	0	0.00955	0.0111	0.762	0.000653	0.119	0.00143	0.176	0.48	0.0199	0.0162	0.00889	0.0045
e-waste	0.148	2.7	0.001	0.165	0.13	0.0083	0.00334	0.496	6.04	0.0196	3.74	0.809	0.023	26.2
Glass	2.24	0	0.0064	0.0832	0.686	0.000312	0.0037	0.000351	0.0477	0.968	0.00606	0.0143	0.00757	0.00325
Inert waste (C&D etc)	1.18	0	0.0439	0	1.18	0.000285	0.00131	0.00132	1.32	0.188	0.0456	0.000812	0.00126	0.00364
Organic waste (cooked)	1.08	18.2	0.996	0.378	0.0289	6.67E-05	0.000118	0.000609	0.00516	0.103	0.000678	6.91E-05	7.00E-06	0.00492
Organic waste(raw)	0.312	39.5	0.231	0.184	0.103	2.62E-05	0.000524	0.00125	0.031	0.121	0.00861	0.000257	0.000104	0.0025
Other	2.19	27.3	0.0608	0.0718	0.0454	2.00E-05	7.28E-05	0.000509	0.0152	0.034	0.000484	3.67E-05	4.33E-05	0.00403
Other Metal	0.0539	0	0.0212	0.0099	21.5	0.0025	0.0172	0.042	72.7	0.0849	0.354	0.0152	0.00241	0.0275
Other Plastic	0.0575	3.56	0.0148	0.0551	3.16	0.000322	0.0091	0.924	2.21	0.0657	0.0189	0.0476	0.0576	0.46
Paper	0.0977	38.5	0.00957	0.178	1.17	6.70E-05	0.000668	0.00428	0.0789	0.124	0.00438	0.000307	0.000269	0.00238
Pet	0.133	5.2	0.027	0.109	6.68	2.00E-05	0.00243	0.00343	0.183	0.0449	0.0195	0.000453	0.000435	0.00873
Plastic Bag	0.117	11.1	0.561	0.052	0.565	2.00E-05	0.000287	0.00948	0.0849	0.0456	0.00101	0.000444	0.000128	0.0074
Tetrapak	0.174	30.8	0.0189	0.0534	5.58	2.00E-05	0.000289	0.00176	0.0539	0.0179	0.00157	0.000276	0.000176	0.00113
Textiles	0.359	34.8	0.23	0.397	0.0879	2.00E-05	0.0475	0.00214	0.034	0.0409	0.00151	0.000147	0.0149	0.0211
Toilet paper/paper towel	0.206	44.8	0.11	0.0883	0.0681	2.00E-05	0.000582	0.00529	0.072	0.0501	0.00172	0.000264	0.000432	0.00428
Yard Waste	0.17	16.5	0.18	0.11	0.8	0.0002	0.00165	0.00105	0	0.156	0.026	0.000213	0.000626	0.00661

Table C.5 : Waste from the outside of the buildings

Source		unit
Glass waste- drop off	12	tonne/year
Yard Waste	195	tonne/year
Total	207	tonne/year

Functional unit of the LCA study= $1783+207=1990.5$ tonne/year

Waste composition from the campus were classified under four groups namely; packaging, cardboard and paper as first group, glass is second group, food and yard waste are third group and others are fourth group. Aluminium, cardboard, other metal, other plastic, paper, pet, plastic bag and tetrapak are classified under the first group, glass and colorful glass are second group, organic waste (cooked), organic waste(raw) and yard waste are third group and ceramics, e-waste, inert waste (C&D etc), other, textiles, toilet paper/paper towel were classified as fourth group. Mass flow analysis (MFA) and life cycle assessment (LCA) studies were carried according to this classification.



APPENDIX D: COLLECTION AND TRANSPORT

Exhaust emissions from the trucks in this study are compatible with EURO-5 standard. Diesel consumption of collection trucks 5.69 L/tonne waste in Sarıyer (Yaman, 2011) and transportation trucks 0.018 L/tonne-km (Yaman, 2011; Sezer, 2019) in Istanbul. European Side of Istanbul has 4 transport station (TS) and Baruthane TS is closest to the Ayazağa Campus. The distance between Baruthane TS and the facilities were taken from the Istanbul Metropolitan Municipality (IMM) and they are checked by using Google Earth and same software used for determination of campus distance to other WM facilities. Diesel (0.42 L/tonne) and electricity (4 kWh/tonne) consumption in the Transport Station (TS) were taken from literature (Turner et al., 2016).

Table D.1 : Campus- Transport Distance-Europe

Facilities	Distance(km)
Campus- Kısırmandıra Facility	30
Campus- MRF packaging	7.5
Campus- MRF glass	35.5
Campus- Baruthane Transfer Station (TS)	12.7
Campus- Seymen Landfill	120
Kısırmandıra Facility - Seymen Landfill	86
MRF-Seymen Landfill	110
MRF packaging-Recycling Center	52.5
MRF glass-Recycling Center	27.5
MRF Kısırmandıra-Recycling Center	22.5
Baruthane TS - Seymen Landfill	112
Baruthane TS – Kısırmandıra Facility	26
RDF to cement factory	50
Fine compost to use on land	20
Compost leachate to Odayeri landfill	8

*Odayeri landfill is no longer accept municipal solid waste, however leachate treatment is still in operation.

Table D.2 : Transport Trucks Fuel Consumption-(Yaman, 2011; Sezer, 2019)

Transfer Station	Seymen Landfill (L/tonne-km)
Baruthane TS	0.018

Table D.3 : Collection Trucks Fuel Consumption-Sarıyer (Yaman, 2011)

Diesel Consumption Unit	Value
L diesel /tonne waste	5.69



APPENDIX E: KISIRMANDIRA FACILITY

Kısırmandıra Facility (KF) is an integrated WM facility in European Side of Istanbul which consists of composting and MRF units. The facility is accepting mixed waste from Baruthane TS. Mass balance of the KF was done by using 12 months observation data of the facility (Sezer and Arikan, 2011). The waste components in the study by (Sezer and Arikan, 2011) are not identical to this study characterization, so these differences were also considered during the MFA study. Waste receiving in the facility is first going through the $\varnothing 80$ mm trommel screen and under size material is going to composting unit while oversize part is going to MRF and RDF (near the KF). Diesel consumption in $\varnothing 80$ mm trommel screen is 2.31 (L/t) (Grzesik and Malinowski, 2016). “Diesel, burned in building machine, GLO” fromecoinvent was used for calculations. The MRF outputs are sent to recycling center for substituting raw material while RDF product is substituted petroleum-coke in cement factories (Kanat et al, 2006).

Table E.1 : Mass balance of the Kısırmandıra Facility (%) based on (Sezer and Arikan, 2011)

	Composting	MRF	RDF
Organic waste (cooked)	94	1	5
e-waste	0	88.9	11.1
Aluminum	50	25	25
Ceramics	20	20	60
Glass	65	9.5	25.5
Other	5.1	43.2	51.7
Cardboard	35.6	50.9	13.5
Other Metal	24	54	22
Yard Waste	90	0	10
Colorful glass	65	9.5	25.5
Tetra Pak	4.9	25	70.1
Toilet paper/paper towel	40	0	60
Plastic Bag	0	25	75
Paper	35	50	15
Pet	6.7	71.1	22.2
Other Plastic	8.1	32.2	59.7
Inert waste	70	28.5	1.5
Textiles	50	0	50
Organic waste(raw)	94	1	5

Material Recycling Facility (MRF)

Different MRFs were modelled in the case study. MRF in KF is accepting mixed waste while MRF-1 (for packaging, paper, cardboard wastes) and MRF-2 (glass) accepting source separated waste. Literature values (Pressley et al. 2015) were used for the performance of MRFs. Electricity consumption in the MRF-1 (Kurdoğlu, 2013) and MRF-2 (Yıldız-Geyhan, 2016) facilities were taken from previous studies conducted in Istanbul. The recycling efficiency (Pressley et al. 2015) and market substitution ratios (EASETECH database; Fernández-Braña et al., 2020) were taken from the literature.

Table E.2 : Material recovery efficiencies at single-stream MRF (%) of input material amount (Pressley et al. 2015)

Fraction name	Glass	Steel	Al	Paper	Cardboard	HDPE	LDPE	PET	Residues
Aluminium	0	0	97	0	0	0	0	0	3
Office Paper	0	0	0	77	0	0	0	0	23
Cardboard	0	0	0	0	54	0	0	0	46
Folding Cartons	0	0	0	0	54	0	0	0	46
Mixed Paper	0	0	0	77	0	0	0	0	23
HDPE	0	0	0	0	0	98	0	0	2
PET	0	0	0	0	0	0	0	98	2
Mixed Plastic	0	0	0	0	0	98/3	98/3	98/3	2
Other Plastic	0	0	0	0	0	0	98	0	2
Ferrous Metal	0	98	0	0	0	0	0	0	2
Glass - Clear	95	0	0	0	0	0	0	0	5
Glass - Colorful	95	0	0	0	0	0	0	0	5

Table E.3 : Substitution factors and electricity consumption of MRFs

	Range	Value	Source
Market subst. factor (plastic)	0.67-0.90	0.85	(ELPCA, 2012; Fernández-Braña et al., 2020)
Market substitution factor (paper)	0.47-0.90	0.85	(ELPCA, 2012; Fernández-Braña et al., 2020)
Market substitution factor (cardboard)	1	1	(ELPCA, 2012)
Market substitution factor (glass)	0.83-0.84	0.83	(ELPCA, 2012; Fernández-Braña et al., 2020)
Market substitution factor (Non-Fe metals)	0.65-100	0.85	(ELPCA, 2012; Fernández-Braña et al., 2020)
Market substitution factor (Fe metals)	0.51-1	0.7	Tagliaferri et al., 2016
Market substitution factor (Aluminium)	1	1	(ELPCA, 2012)
Market substitution factor (PET)	0.81	0.81	(ELPCA, 2012)
Kısırmandıra MRF and MRF-1 electricity consumption (kWh/t)	10.0-14.3	14.3	(Kurdoğlu, 2013; Yıldız-Geyhan, 2016)
MRF elect consumption waste (kWh/t)	8.97	8.97	(Yıldız-Geyhan, 2016)

Composting

Table E.4 : Kısırmandıra Composting Facility (Gencsoy, 2010)

Parameter	Unit	Input (waste)	Compost Product
pH	-	6.1	7.3
Electrical conductivity	µS/cm	9458	8013
Water content	%	59	28
Organic matter	%	57	38
Ammonia (NH ₃)	mg/kg	317	95
TKN	g/100 g dry matter	1.34	1.55
C/N	-	23	17
DOC	g/kg	101	11.1
Calcium	g/100 g dry matter	5.25	8.67
Potassium	g/100 g dry matter	0.89	0.91
Phosphorus	g/100 g dry matter	0.16	0.25
Magnesium	g/100 g dry matter	0.35	0.57
Sulphur	g/100 g dry matter	0.49	0.75
Sodium	g/100 g dry matter	0.54	0.51
Iron	g/100 g dry matter	1.25	2.12
Aluminum	g/100 g dry matter	1.11	1.72
Manganese	mg/kg	385	666
Zinc	mg/kg	251	460
Copper	mg/kg	130	311
Boron	mg/kg	78	104
Cadmium	mg/kg	0.71	1.24
Cobalt	mg/kg	9.9	13.5
Chromium	mg/kg	423	446
Nickel	mg/kg	121	191
Lead	mg/kg	39	96

The KF composting unit is designed as tunnel system with eight-week active composting (Sezer and Arıkan, 2011). The facility fed with mixed MSW coming from TS and the moisture content of input material changes between 55-63% while output is changing between 20-28% (Gencsoy, 2010). Aerated static piles temperatures kept as 45-60 °C for the first pile, 55-70 °C (65 °C), for 2-4 piles and 45-50 °C for 5-6 piles. After composting the materials are screened with Ø15 mm trommel screen and undersized material used on IMM parks and oversize material sent to the landfill. Leachate is transported to Landfill Leachate Treatment Facility. C/N ratio of the input and output materials are 23 and 17 respectively. VS / C_{bio} ratio is assumed as 1:1 (Boldrin et al., 2009) for modelling. The CO₂, CH₄, CO, NH₃ and N₂O emissions and their value after biofilter treatment taken from literature (Boldrin et al. 2009; Boldrin et al. 2011) The heavy metals were assumed to kept into solid phase. Electricity consumption in tunnel composting was taken as 53 kWh/tonne (Boldrin et al., 2011).

Table E.5 : Gas emissions from the composting unit

Transformation rate[%]	Transformedfrom	Transformed into	Comment
99.8	kg C bio	CO ₂ , non-fossil	95% methane oxidation
0.2*0.05	kg C bio	Methane, non-fossil	
0.2*0.95	kg C bio	CO ₂ , non-fossil	
0.1	kg N	Nitrogen	99% NH ₃ oxidation
98.5*0.01	kg N	Ammonia	
1.4	kg N	Dinitrogen monoxide	

Table E.6 : Emissions after biofilter

Name	Compartment	Amount	Unit	Per
Terpenes	air	0.001222/1000	kg	kg Total Wet Weight
Hydrogen sulfide	air	0.0001929/1000	kg	kg Total Wet Weight

Table E.7 : Leachate from the composting facility

Name	Compartment	Sub compartment	Amount	Unit	Per
Arsenic, ion	water	unspecified	0.000219/1000	kg	kg Total Wet Weight
Bromide	water	unspecified	0	kg	kg Total Wet Weight
Calcium, ion	water	unspecified	0.6658/1000	kg	kg Total Wet Weight
Cadmium, ion	water	unspecified	9.08E-10	kg	kg Total Wet Weight
Chloride	water	unspecified	0.4985/1000	kg	kg Total Wet Weight
Chromium, ion	water	unspecified	2.08E-08	kg	kg Total Wet Weight
Copper, ion	water	unspecified	0.000238/1000	kg	kg Total Wet Weight
Iron, ion	water	unspecified	0.02593/1000	kg	kg Total Wet Weight
Mercury	water	unspecified	1.01E-09	kg	kg Total Wet Weight
Potassium, ion	water	unspecified	1.069/1000	kg	kg Total Wet Weight
Magnesium	water	unspecified	0.09886/1000	kg	kg Total Wet Weight
Manganese	water	unspecified	0.001316/1000	kg	kg Total Wet Weight
Molybdenum	water	unspecified	0.000261/1000	kg	kg Total Wet Weight
Sodium, ion	water	unspecified	0.2124/1000	kg	kg Total Wet Weight
Nitrate	water	unspecified	0.09456/1000	kg	kg Total Wet Weight
Nickel, ion	water	unspecified	0.000119/1000	kg	kg Total Wet Weight
Lead	water	unspecified	8.22E-08	kg	kg Total Wet Weight
Sulfate	water	unspecified	0.5/1000	kg	kg Total Wet Weight
Antimony	water	unspecified	4.17E-08	kg	kg Total Wet Weight
Zinc, ion	water	unspecified	0.00227/1000	kg	kg Total Wet Weight
TOC	water	unspecified	0.1/1000	kg	kg Total Wet Weight

Refuse Derived Fuel (RDF)

The RDF unit is located near to the KF. The wastes are first brought to the size of 250-300 mm in a coarse crusher. Afterwards, metal and organic materials are separated. The final product, 0-30 mm size, in the fine crusher unit, is transferred to cement factory (Kara et al, 2010). “*petroleum coke, petroleum coke production, petroleum refinery operation_Europe without Switzerland_2020_Allocation, APOS*” data from ecoinvent was used for RDF substitution calculation. Gas emissions from stack per kg waste were calculated by using emission data from (Kara et al, 2010). According to study 6030.77 kg/h petroleum coke is equivalent to 13,440 kg/h RDF (Kara et al, 2012). Electricity consumption in RDF was taken as 40 kWh/tonne (Burnley et al.,2011).

Table E.8 : Combustion gas emission results of stack gas measurement results (Dust, TOC, and halogens) (Kara et al, 2010).

Parameter	mg/kg waste
CO	14.575
SO ₂	0.002
NO	2.771
NO _x	4.270
Dust	0.131
HCl	0.010
HF	0.001
Total organic carbon (TOC)	0.085
PAH	-
	0.016

Table E.9 : Combustion gas emission results of stack gas measurement results (heavy metals) (Kara et al, 2010)

Parameter	mg/kg waste
Thallium and compounds (as Tl),	4.66E-05
Cadmium and compounds (as Cd),	5.82E-06
Mercury and compounds (as Hg),	5.82E-06
Chromium and compounds (as Cr),	1.75E-05
Copper and compounds (as Cu),	1.75E-05
Cobalt and compounds (as Co),	1.75E-05
Manganese and compounds (as Mn),	1.63E-04
Nickel and compounds (as Ni),	1.75E-05
Lead and compounds (as Pb),	1.75E-05
Antimony and compounds (as Sb),	5.82E-06
Tin and compounds (as Sn),	7.57E-05
Arsenic and compounds (as As),	4.08E-05
Vanadium and compounds (as V),	5.82E-06
Dioxin and Furan,	4.31E-06

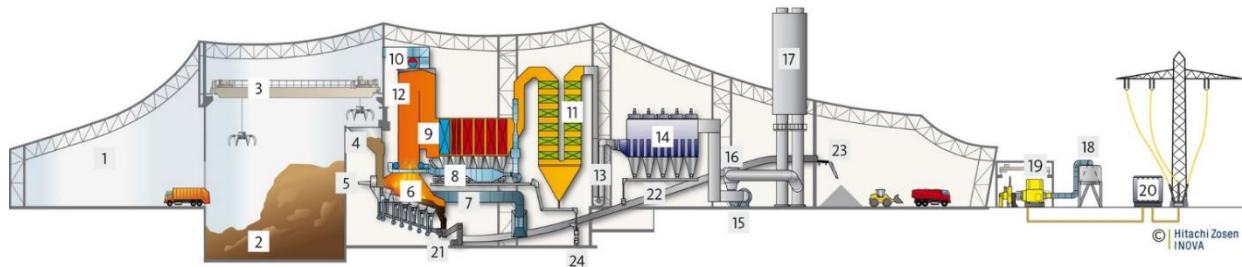
APPENDIX F: INCINERATION

The incineration plant in Istanbul is still in preliminary phase so design and capacity data were taken from environmental impact assessment report of the facility (IMM, 2012). The plant will operate with 78 kW/day electricity and 175 kW/day heat generation. The ferrous metal separation from the plant, missing data for waste specific and process specific emissions were taken from (Bisinella et al., 2021).

Table F.1 : New Incineration Plant in Istanbul – Design Parameters (IMM,2012)

Process	Value
Operation-annual	8000 hr/year
Number of transportation trucks	65
Incinerator air flow	120.000 Nm ³ /hr
Temperature	850-1100 C
Turbine conditions	40 bar - 400 C temp
Electricity production capacity	78 MWH/day
Heat production capacity	175 MWH/day

It is planned to use 50,000 Sm³ natural gas at the start-up phase of the system. In cases where natural gas is insufficient, 100 m3 fuel oil will be used. Fly ash residues will be collected and stored in the dedicated silos with a storage period of at least 4 days.



Waste Receiving and Storage	Combustion and Boiler	Flue Gas Treatment	Energy Recovery	Residue Handling and Treatment
1 Tipping hall	4 Feed hopper	12 SNCR	18 Air cooled condenser	21 Bottom ash extractor
2 Waste bunker	5 Ram feeder	13 Xerosorp® Reactor	19 Turbine	22 Bottom ash conveying
3 Waste crane	6 HZI grate	14 Fabric filter	20 Transformer	23 Bottom ash discharge
	7 Primary air system	15 Induced draught fan		24 Fly ash discharge
	8 Secondary air system	16 Silencer		
	9 Five-pass boiler	17 Stack		
	10 Boiler drum			
	11 Economiser			

Figure F.1: Incineration plant scheme from the environmental impact assessment report of the facility (IMM, 2012). https://www.hz-inova.com/projects/istanbul-tur/?utm_source

Table F.2 : External Process- (Bisinella et al., 2021).

Name	Amount	Unit	Per
MISSING - TMT 15	0.000395	kg	kg Total Wet Weight
Sodium hydroxide, 1996, RER, ELCD	0.00123	kg	kg Total Wet Weight
Hydrogen chloride gas (HCl), 1996, RER ELCD	0.0005	kg	kg Total Wet Weight
Calcium carbonate, limestone, CaCO ₃ > 63 microns, EU-27, 2006, ELCD	0.01186	kg	kg Total Wet Weight
Process water from surface water, RER, 2005, ELCD	0.397	kg	kg Total Wet Weight
Polyethylene high density granulate (PE-HD), 1999, RER, ELCD	6E-07	kg	kg Total Wet Weight
Hydrated Lime EU 2007; Production at plant, RER, 2007, ELCD	0.00034	kg	kg Total Wet Weight
LIMITED - Activated Carbon	0.0005	kg	kg Total Wet Weight
LIMITED - Ammonia (NH ₃)	0.00286	kg	kg Total Wet Weight

Table F.3 : Elementary exchanges

Name	Compartment	Sub compartment	Amount	Unit
Ammonia	air	non-urban air or from high stacks	4.31E-06	kg
Antimony	water	surface water	8.8E-09	kg
Arsenic, ion	water	surface water	5.6E-10	kg
Barium	water	surface water	7.2E-09	kg
Cadmium, ion	water	surface water	4E-11	kg
Calcium, ion	water	surface water	4.16E-05	kg
Carbon monoxide, fossil	air	non-urban air or from high stacks	2.56E-05	kg
Chloride	water	surface water	0.000146	kg
Chromium, ion	water	surface water	4.48E-09	kg
Cobalt	water	surface water	4E-11	kg
Copper, ion	water	surface water	2E-07	kg
Dioxins, measured as 2,3,7,8-tetrachlorodibenzo-p-dioxin	air	non-urban air or from high stacks	4.31E-14	kg
Fluoride	water	surface water	2.08E-06	kg
Hydrogen chloride	air	non-urban air or from high stacks	2.16E-06	kg
Hydrogen fluoride	air	non-urban air or from high stacks	2.16E-07	kg
Iron, ion	water	surface water	4E-08	kg
Lead	water	surface water	5.6E-10	kg
Magnesium	water	surface water	2.56E-08	kg
Manganese	air	non-urban air or from high stacks	1.12E-05	kg
Manganese	water	surface water	6.4E-10	kg
Mercury	water	surface water	1.6E-11	kg
Molybdenum	water	surface water	5.76E-08	kg
Nickel, ion	water	surface water	1.68E-09	kg
Nitrogen oxides	air	non-urban air or from high stacks	7.52E-05	kg
PAH, polycyclic aromatic hydrocarbons	air	non-urban air or from high stacks	4.31E-09	kg
Particulates, > 10 um	air	non-urban air or from high stacks	2.16E-06	kg
Selenium	water	surface water	1.12E-09	kg
Silicon	water	surface water	2.4E-07	kg
Sulfur dioxide	air	non-urban air or from high stacks	8.63E-06	kg
Zinc, ion	water	surface water	2.56E-09	kg



APPENDIX G: LANDFILL

The Seymen Landfill has been operated since 2017, so landfill gas collection rate (%) was assumed by using default average gas collection performance of EASETECH. The waste degradation rates for each waste composition was taken from (De la Cruz and Barlaz, 2010; U.S. EPA 2011) which they already defined in the EASTECH as default values. Uncollected gas was partly oxidized in intermediate and top layers.

Table G.1 : Methane decay rates in different climates (EASTECH; U.S. EPA 2011)

Waste fraction	Moderate condition [year- 1]
Advertisements	0.116
Animal excrements and bedding	0.137
Animal food waste	0.137
Books, phone books	0.029
Cotton, bandages	0.028
Diapers, sanitary towels, tampons	0.029
Dirty cardboard	0.019
Dirty paper	0.029
Disposable sanitary products (cloths, gloves)	0.029
Juice cartons (carton/plastic/aluminum)	0.019
Kitchen towels	0.029
Magazines	0.116
Milk cartons (carton/plastic)	0.019
Newsprint	0.031
Office paper	0.028
Other clean cardboard	0.019
Other clean paper	0.028
Paper and carton containers	0.019
Vegetable food waste	0.137
Yard waste, flowers	0.162

Table G.2 : Landfill gas collection module. Collection efficiencies for state-of-the-art, average and limited landfill gas collection. (U.S. EPA 2011; Spokas et al. 2006; Barlaz 2012; Sullivan 2010;; Henriksen et al, 2017)

Time period duration (years)	Collected %	Not collected % (year 0-5)	Not collected % (year 5-15)	Not collected % (year 15-100)
5	35	65	0	0
10	65	0	35	0
40	75	0	0	25
45	0	0	0	100

Table G.3 : Landfill gas oxidation module. Average performing landfill. Top cover. Daily cover Oxidation efficiencies.(Chanton, Powelson, and Green 2009; Sullivan 2010; Oonk 2012; U.S. EPA 2011)

Material property	Transformed at (%) into	Elementary exchange	Compartment	With the conversion factor
kg Benzene	96	Benzene	air	1
kg Ethylbenzene	95	Benzene, ethyl-	air	1
m ³ CH ₄	10	Carbon dioxide, non-fossil	air	44/Volume_gas
kg Chloroform	91	Chloroform	air	1
kg CFC ₁₁₃	100	Ethane, 1,1,2-trichloro-1,2,2-trifluoro-, CFC-113	air	1
kg PCE	99	Ethene, tetrachloro-	air	1
kg H ₂ S	97	Hydrogen sulfide	air	1
kg Hg	100	Mercury	air	1
kg HCFC ₂₂	94	Methane, chlorodifluoro-, HCFC-22	air	1
kg Dichloromethane	89	Methane, dichloro-, HCC-30	air	1
kg CFC ₁₂	96	Methane, dichlorodifluoro-, CFC-12	air	1
kg HCFC ₂₁	91	Methane, dichlorofluoro-, HCFC-21	air	1
m ³ CH ₄	90	Methane, non-fossil	air	16/Volume_gas
kg Carbon tetrachloride	87	Methane, tetrachloro-, R-10	air	1
kg CFC ₁₁	87	Methane, trichlorofluoro-, CFC-11	air	1
kg NMVOC	91	NMVOC, non-methane VOC, unspecified origin	air	1
kg Toluene	94	Toluene	air	1
kg Xylenes	95	Xylene	air	1
kg Chloromethane	89	Methane, monochloro-, R-40	air	1
kg Ethylchloride	91	Monochloroethane	air	1
kg Propylbenzene	97	Cumene	air	1
kg Dichlorobenzene	95	Benzene, dichloro	air	1
kg Chlorobenzene	94	Benzene, chloro-	air	1
kg TCE	92	Ethene, trichloro-	air	1
kg VC	89	Ethene, chloro-	air	1
kg Phenol	94	Phenol	air	1
kg Naphtalene	99	Naphthalene	air	1

Table G.4 : Landfill gas oxidation module. Average performing landfill. Intermediate cover

Material property	Transformed at (%) into	Elementary exchange	Compartment	With the conversion factor
kg Benzene	93	Benzene	air	1
kg Ethylbenzene	90	Benzene, ethyl-	air	1
m ³ CH ₄	20	Carbon dioxide, non-fossil	air	M_CO ₂ /Volume_gas
kg Chloroform	83	Chloroform	air	1
kg CFC ₁₁₃	100	Ethane, 1,1,2-trichloro-1,2,2-trifluoro-, CFC-113	air	1
kg PCE	97	Ethene, tetrachloro-	air	1
kg H ₂ S	95	Hydrogen sulfide	air	1
kg Hg	100	Mercury	air	1
kg HCFC ₂₂	89	Methane, chlorodifluoro-, HCFC-22	air	1
kg Dichloromethane	79	Methane, dichloro-, HCC-30	air	1
kg CFC ₁₂	91	Methane, dichlorodifluoro-, CFC-12	air	1
kg HCFC ₂₁	83	Methane, dichlorofluoro-, HCFC-21	air	1
m ³ CH ₄	80	Methane, non-fossil	air	M_CH ₄ /Volume_gas
kg Carbon tetrachloride	75	Methane, tetrachloro-, R-10	air	1
kg CFC ₁₁	74	Methane, trichlorofluoro-, CFC-11	air	1
kg NMVOC	85	NMVOC, non-methane volatile organic compounds, unspecified origin	air	1
kg Toluene	88	Toluene	air	1
kg Xylenes	90	Xylene	air	1
kg Chloromethane	79	Methane, monochloro-, R-40	air	1
kg Ethylchloride	85	Monochloroethane	air	1
kg Propylbenzene	95	Cumene	air	1
kg Dichlorobenzene	90	Benzene, dichloro	air	1
kg Chlorobenzene	88	Benzene, chloro-	air	1
kg TCE	84	Ethene, trichloro-	air	1
kg VC	79	Ethene, chloro-	air	1
kg Phenol	88	Phenol	air	1
kg Naphtalene	97	Naphtalene	air	1

Table G.5 : Landfill gas oxidation module. Average performing landfill. Final Cover

Material property	Transformed at (%) into	Elementary exchange	Compartment	With the conversion factor
kg Benzene	87	Benzene	air	1
kg Ethylbenzene	82	Benzene, ethyl-	air	1
m ³ CH ₄	36	Carbon dioxide, non-fossil	air	M_CO ₂ /Volume_gas
kg Chloroform	69	Chloroform	air	1
kg CFC ₁₁₃	100	Ethane, 1,1,2-trichloro-1,2,2-trifluoro-, CFC-113	air	1
kg PCE	95	Ethene, tetrachloro-	air	1
kg H ₂ S	90	Hydrogen sulfide	air	1
kg Hg	100	Mercury	air	1
kg HCFC ₂₂	80	Methane, chlorodifluoro-, HCFC-22	air	1
m ³ CH ₄	64	Methane, non-fossil	air	M_CH4/Volume_gas
kg Carbon tetrachloride	56	Methane, tetrachloro-, R-10	air	1
kg CFC11	55	Methane, trichlorofluoro-, CFC-11	air	1
kg NMVOC	70	NMVOC, non-methane volatile organic compounds, unspecified origin	air	1
kg Toluene	78	Toluene	air	1
kg Xylenes	82	Xylene	air	1
kg Chloromethane	62	Methane, monochloro-, R-40	air	1
kg Ethylchloride	70	Monochloroethane	air	1
kg Propylbenzene	90	Cumene	air	1
kg Dichlorobenzene	82	Benzene, dichloro	air	1
kg Chlorobenzene	78	Benzene, chloro-	air	1
kg TCE	72	Ethene, trichloro-	air	1
kg VC	63	Ethene, chloro-	air	1
kg Phenol	78	Phenol	air	1
kg Naphtalene	95	Naphtalene	air	1

Table G.6 : Combustion and treatment

Material property	Transformed at (%) into	Elementary exchange	Component	With the conversion factor
m ³ CH ₄	99	Carbon dioxide, non-fossil	air	M_CO2/Volume_gas
m ³ CH ₄	1	Methane, non-fossil	air	M_CH4/Volume_gas
kg NMVOC	97.2	Carbon dioxide, non-fossil	air	1
kg NMVOC	2.8	NMVOC, non-methane volatile organic compounds, unspecified origin	air	1
kg Hg	100	Mercury	air	1
kg H ₂ S	2.8	Hydrogen sulfide	air	1
kg H ₂ S	97.2	Sulfur dioxide	air	64.06/34.08
kg VC	7	Ethene, chloro-	air	1
kg VC	93	Carbon dioxide, non-fossil	air	M_CO2/62.5
kg TCE	7	Ethene, trichloro-	air	1
kg TCE	93	Carbon dioxide, non-fossil	air	M_CO2/131.4
kg PCE	7	Ethene, tetrachloro-	air	1
kg PCE	93	Carbon dioxide, non-fossil	air	M_CO2/165.8
kg Benzene	13.9	Benzene	air	1
kg Benzene	86.1	Carbon dioxide, non-fossil	air	M_CO2/78.11
kg Chlorobenzene	13.9	Benzene, chloro-	air	1
kg Chlorobenzene	86.1	Carbon dioxide, non-fossil	air	M_CO2/112.56
kg Dichlorobenzene	13.9	Benzene, dichloro	air	1
kg Dichlorobenzene	86.1	Carbon dioxide, non-fossil	air	M_CO2/147
kg Ethylbenzene	13.9	Benzene, ethyl-	air	1
kg Ethylbenzene	86.1	Carbon dioxide, non-fossil	air	M_CO2/106.16
kg Propylbenzene	13.9	Cumene	air	1
kg Propylbenzene	86.1	Carbon dioxide, non-fossil	air	M_CO2/120.19
kg Ethylchloride	7	Monochloroethane	air	1
kg Ethylchloride	93	Carbon dioxide, non-fossil	air	M_CO2/64.51
kg Chloromethane	7	Methane, monochloro-, R-40	air	1
kg Chloromethane	93	Carbon dioxide, non-fossil	air	M_CO2/50.49
kg Dichloromethane	7	Methane, dichloro-, HCC-30	air	1
kg Dichloromethane	93	Carbon dioxide, non-fossil	air	M_CO2/84.93
kg Chloroform	7	Chloroform	air	1
kg Chloroform	93	Carbon dioxide, non-fossil	air	M_CO2/119.39
kg Carbon tetrachloride	7	Methane, tetrachloro-, R-10	air	1
kg Carbon tetrachloride	93	Carbon dioxide, non-fossil	air	M_CO2/153.84



APPENDIX H: LEACHATE

Leachate amount was calculated by using method suggested by (Yang et al., 2015), which include water coming from both precipitation and waste itself. HELP model was used to quantify leachate coming from precipitation while theoretical calculations were used to determine leachate from gravity, compaction and degradation processes.

Theoretical calculation (leachate from waste moisture)

Moisture content was taken as 61% (Ak and Sezer, 2020) while field capacities for fresh and old wastes were assumed as $0.45(w \cdot w^{-1})$ and $0.30(w \cdot w^{-1})$ respectively (Yang et al., 2015). Details of the method and calculations were explained in (Yang et al., 2015) however related equations and assumptions were given below.

$WS_C = IDM \times (IMC_{DM}/100 - FCC_{DM}/100) \times 1000$ (water coming from gravity and compaction),

- WS_C is the amounts of water squeezed by compaction and gravity from 1 tonne of raw waste
- IDM represents the initial dry matter weight in 1 tonne of raw waste
- IMC_{DM} is the initial moisture content based on the weight of dry matter
- FCC_{DM} is field capacity after compaction based on the weight of dry matter

$WS_D = (IDM \times FFC_{DM}/100 - DMA \times FCA_{DM}/100) \times 1000$ (water coming from degradation)

- WS_D is water squeezed by degradation,
- IDM_i is the initial dry matter weight of waste fraction i in 1 tonne of raw waste, (“ i ” separated to three parts as fast, slow and non-degradable)

Fast degradable (i.e. food waste) ratio was taken as 38% for the degradation calculations while slow degradable (i.e. paper, wood and textile) was taken 25% and non-degradable (i.e. plastics, metal, slag and glass) was 37% (Yang et al., 2015).

- DMA represents the dry matter weight of aged waste remaining from 1 tonne of raw waste,
- FCA_{DM} is the field capacity of aged waste on a dry basis

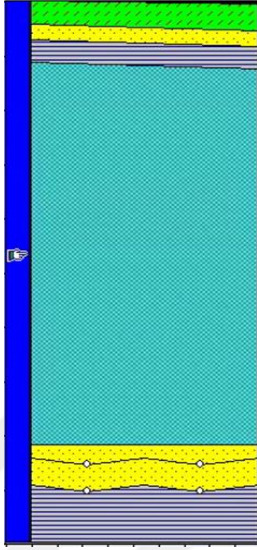
Total leachate due to squeezed water from waste = $WS = WS_C + WS_D = 333.7$ L/tonne waste

HELP model results (leachate from precipitation)

The HELP model was developed by EPA to find out real infiltration rate on landfills (Schroeder et al, 1994). Leachate amount is affected by precipitation, evaporation and landfill layers to find out real amount the HELP model was run for 100 years. The model using 100 years precipitation (mm/year), temperature ($^{\circ}C$) and solar radiation (mj/m^2) values of Istanbul. The landfill layer (profil structure) is complying with Turkish Landfill Regulation requirements.

Table H.1 : Profile Structure

Layer	Thick ness (cm)
 Fine Sandy Loam	50
 Coarse Sand	30
 Low Density Polyethylene	0.10
 Silty Clay	40
 Municipal Waste	3000
 Sand	30
 Drainage Net (0.5cm)	0.50
 High Density Polyethylene	0.10
 Fine Sand	50
 Butyl Rubber	0.10
 Sandy Clay Loam	100



The porosity and field capacity values of the materials used were taken from HELP model and (Yang et al., 2015).

Table H.2 : Parameters of the materials used in the HELP model

	Fine Sandy Loam	Coarse Sand	Clay	Sand	Geotextiles and Geonets (Drainage)	Fine Sand	Sandy Clay Loam
Total porosity ($v \cdot v^{-1}$)	0.4730	0.417	0.479	0.437	0.8500	0.4570	0.3980
Field capacity ($v \cdot v^{-1}$)	0.2220	0.045	0.371	0.062	0.01	0.0830	0.2440
Wilting point ($v \cdot v^{-1}$)	0.1040	0.018	0.251	0.024	0.005	0.0330	0.1360
Saturated hydraulic conductivity ($\text{cm} \cdot \text{s}^{-1}$)	44.928 (cm/day)	0.01	2.5E-5	0.0058	86400 (cm/day)	267.84 (cm/day)	0.01

The total leachate amount is affected by run-off ratios. Hence model was run by three different runoff assumptions namely; 0%, 50% and 100%. Results shows 0% run off has highest infiltration rate so to be on the safe side 0% run infiltration result (355.1 L/tonne waste) was used to calculate total infiltration.

Table H.3 : Runoff ratio and vegetation class assumptions

Parameter	Value	Unit
Runoff Area	0, 50, 100	(%)
Vegetation Class	Good stand of grass	(-)

Total leachate (Ws + Precipitation) = 688.8 L/tonne waste

Table H.4 : Leachate collection module (Olesen and Damgaard 2014; Rowe and Islam 2009)

Time period duration [years]	Collected [%]	Not collected [%]
80	99.9	0.1
20	87	13

The non-collected leachate amount was modelled by assuming the surface water as an ending point. The landfill module was modelled for 100 years and 99.9% leachate collection was assumed for first 80 years while 87% collection was assumed for remain 20 years. WWTP energy consumptions-avg data for Türkiye with secondary treatment is 0.30 kWh/m³ (Turkish water institute, 2019) and average COD is approximately 0.45 kg/m³ (450 mg/L). Based on that electricity consumption is found as 1.5 kWh/kg COD. Normally, EASETECH assumes the dewatered sludge is applied to industrial soil however by considering current conditions in Istanbul, fate of sludge is change to landfill instead of industrial soil area.



APPENDIX I:ANAEROBIC DIGESTER

The anaerobic digestion (AD) process is a single-stage wet mesophilic process and there is a hygienization tank was placed to remove pathogens at 70°C and 1 hour (Akça et al. 2020). The solid content in the digester is 8% and hydraulic retention time is 30 days. The AD fed by source separated food waste from main dining-hall of the campus. The AD will accept waste from other cafeteria units of the campus when it will operate in full-scale. The electricity generated from the biogas will used in the electrical car charging station, heat will be used by the AD itself to kept digester in the mesophilic conditions. The methane in the biogas taken as 65%. A heat demand by the facility for keeping it in mesophilic conditions (38°C) was also calculated in EASTECH by taking average ambient temperature in campus as 16 °C. The biogas leaking in the facility was taken as 2% (Scheutz and Fredenslund, 2019). Total energy consumption in the AD is 49 kWh/tonne waste.

Table I.1 : Biogas Facility in Ayazağa Campus- primary source and (Akça et al., 2020)

Biogas Facility in Ayazağa Campus		
Volume	50	m ³
HRT	30	day
Ambient avg temperature	16.2	C°
Digester temperature	38	C°
TS in digester (wet type)	8	%
max_capacity_FW (wet)	1.6	tonne/day
min_VS	54	%
max_VS	87	%
avg_VS	80	%
biogas yield (avg)	572.5	L/kg VS
CH ₄ in biogas	65	%

The digestate from the facility fed into French type constructed wetland. The effluent from the wetland will used for irrigation in the campus. Unfortunately, currently no data is available for the irrigated area due to Covid-19 restrictions in the campus. The missing data for the wetland modelling was taken from the literature (Wang et al. 2009; Paing et al. 2015). The heavy metals are assumed to kept into liquid phase of digestate (Boldrin et al. 2011).

Table I.2 : Wetland in Ayazağa Campus- digestate preliminary results

Biogas Facility in Ayazağa Campus		
Q	0.8	m ³ /day
Type	French vertical	Flow direction
Hydraulic slope	0.5-1	%
Bed size –first step	120	m ²
Bed size –second step	80	m ²
Bed-media	gravel	-
Macrophyte		-
PhragmitesAustralis,		
Typha latifolia		
Influent COD	14208	mg/L
Influent TKN	1302	mg/L
Influent TP	450	mg/L
Effluent COD	142	mg/L (Wang et al., 2009)
Effluent TKN	91	mg/L (Wang et al., 2009; Paing et al., 2015)
Effluent TP	315	mg/L (Paing et al., 2015)

Table I.3 : Emissions from wetland area was found as flux (mg m⁻² h⁻¹) from the literature (Mander et al., 2014; Maucieri et al., 2017) and annual emissions were calculated by using total wetland area (200 m²).

Name	Compartment	Amount	Unit	Per
Methane	air	0.001496 x 10 ⁻⁶	kg	kg Total Wet Weight
Dinitrogen monoxide	air	2.64E-05 x 10 ⁻⁶	kg	kg Total Wet Weight
Carbon dioxide, non-fossil	air	0.08731 x 10 ⁻⁶	kg	kg Total Wet Weight

APPENDIX J: USE ON LAND

The majority of carbon and nitrogen is emitted to air but some remains in the soil or is leached out of the soil. The fate of carbon and nitrogen is greatly affected by the climate and soil type. C,N,P related emissions, leaching and carbon sequestration were calculated by using literature sources (Gençsoy, 2010; Yoshida et al., 2014; Cobo et al. 2018). The chemical fertilizer content was provided from EASETECH database to calculate substitutions of compost and digestate.

Table J.1 : Distribution of applied C (%) (Gençsoy, 2010)

Product	Unit	CO ₂	Leach C	Stored C process in soil
Compost	Tunnel	90.11	0.11	9.88
Solid Digestate	Anaerobic Digester	87.15	0.05	12.80

Table J.2 : Distribution of the applied N (%) (Gençsoy, 2010; Cobo et al. 2018)

Product	Unit process	N uptake	Leach NO ₃	N ₂ Oair emissions	NO air emissions	Stored N in soil
Compost	Tunnel	25.01	69.12	0.84	0.07	4.96
Solid Digestate	Anaerobic Digester	52.93	41.36	0.96	0.01	4.74

Table J.3 : Distribution of applied P (%) (Cobo et al. 2018)

Process	Unit	P uptake	P leach	Stored P
Compost	Tunnel	10.73	71.29	17.98
Solid Digestate	Anaerobic Digester	19.28	56.23	24.48



APPENDIX K: ENERGY SOURCES

The missing data in the background systems were obtained from the Ecoinvent 3.7.1 (Ecoinvent, 2020). Electricity grid of Istanbul was assumed as same with Türkiye electricity mix. The marginal electric mix data was taken from literature (Ecoinvent, 2020; EPDK, 2020). However, electricity mix calculated by using fossil and renewable electricity source values, instead of directly taken Turkish mix value in Ecoinvent, since the data based on 2017 and electricity mixture in Türkiye change thanks to new hydro-electricity facilities (EPDK, 2020).

Table K.1 : Licensed Electricity Generation and Consumption in Türkiye (EPDK.2020)

Resource Type	2019 (GWh)
Hydro Gam	88.850,17
Import Hard Coal	60.381,27
Natural Gas	56.522,71
Lignite	46.893,73
Wind	21.636,28
Geothermal	8.929,73
Biomass	4.266,32
Hard Coal	3.518,87
Asphalite Coal	2.323,95
Fuel Oil	732,92
Solar	194,37
Diesel	1,00
Total	294.251,32

Table K.2: Licensed Electricity Generation and Consumption in Türkiye (EPDK.2020)

Resource Type	Share (%)
Hydro Gam	30.2
Import Hard Coal	20.5
Natural Gas	19.2
Lignite	15.9
Wind	7.4
Geothermal	3.0
Biomass	1.5
Hard Coal	1.2
Asphalite Coal	0.8
Fuel Oil	0.3
Solar	0.1
Diesel	0.0
Total	100.0

Table K.3 : Türkiye Electricity mix GWP 100 for 2020.

Climate change w/o LT; midpoint; GWP100	
Unit	kg CO₂-Eq
Sum	0.5532
electricity, high voltage,heat and power co-generation, lignite,TR	0.1971
electricity, high voltage,heat and power co-generation, hard coal,TR	0.2651
electricity, high voltage,electricity production, wind, >3MW turbine, onshore,TR	0.0014
electricity, high voltage,electricity production, natural gas, combined cycle power plant,TR	0.0811
electricity, high voltage,electricity production, hydro, run-of-river,TR	0.0012
electricity, high voltage,electricity production, oil,TR	0.0023
electricity, low voltage,electricity production, photovoltaic, 3kWp, multi-Si, panel, mounted,TR	0.0000
electricity, high voltage,electricity production, deep geothermal,TR	0.0020
electricity, high voltage,heat and power co-generation, biogas, gas engine,TR	0.0030

A lower carbon cost assumption was applied and this cost was supported by the purchase guarantee policies to be applied to renewable energy sources for balanced Policy Action Scenario for 2030 (SHURA,2020). The main objective in this scenario is to achieve a balanced implementation of different policy objectives such as carbon emission reduction, use of domestic resources and cheap electricity. Under the scenario, the carbon price reaches 25 USD/tonne CO₂ equivalent in 2030.

Table K.4 : Türkiye Electricity mix forecasting- 2030 (SHURA,2020)

Resource Type	Balanced Policy Action Scenario -2030
Coal	17.9
Gas	23.8
Nuclear	3
Hydro	17.9
Wind	19.0
Solar	14.3
Other	4.1
Total	100

Table K.5 : Turkiye Electricity mix GWP 100 for 2030.

Climate change w/o LT; midpoint; GWP100	
Unit	kg CO ₂ -Eq
Sum	0.33735
electricity, high voltage,heat and power co-generation, lignite,TR	0.221341
electricity, high voltage,electricity production, wind, >3MW turbine, onshore,TR	0.003711
electricity, high voltage,electricity production, natural gas, combined cycle power plant,TR -	0.100416
electricity, high voltage,electricity production, deep geothermal,TR	0.002724
electricity, high voltage,electricity production, hydro, run-of-river,TR	0.000726
electricity, low voltage,electricity production, photovoltaic, 3kWp slanted-roof installation, multi-Si, panel, mounted,TR	0.00825
electricity, high voltage,electricity production, nuclear, pressure water reactor,FR	0.000182



APPENDIX L: ECOINVENT DATA FOR WASTE PREVENTION

The waste reduction after ZW awareness campaign were calculated. 70 % waste prevention was assumed for paper, cardboard, other plastics, plastic bag and plastic bottle.

Table L.1 : Ecoinvent data for waste prevention

Waste component	Ecoinvent data
Paper	paper, newsprint,paper production, newsprint, recycled,Europe without Switzerland
Cardboard	waste paperboard, sorted,corrugated board box production,RER
Other Plastic	polyethylene, high density, granulate,polyethylene production, high density, granulate,RER
Plastic Bag	nylon 6,nylon 6 production,RER
Plastic bottles (pet)	polyethylene terephthalate, granulate, bottle grade,polyethylene terephthalate production, granulate, bottle grade,RoW



APPENDIX M: DATA QUALITY ASSESSMENT

The Data Quality Assessment indicators are based on (Weidema and Wesnæs, 1996).

Table M.1 : Quality scoring matrix (Weidema and Wesnæs, 1996)

Indicator scores / category	1	2	3	4	5
Technological correlation	Data from enterprises processes and materials under study	Data from processes and materials under study but from different enterprises	Data from processes and materials under study but from different technology	Data on related processes or materials but from same technology	Data on related processes or materials but from different technology
Geographic al correlation	Data from area under study	Average data from larger area in which the area under study is included	Data from area with similar production conditions	Data from area with slightly similar production conditions	Data from unknown area or area with very different production conditions
Temporal correlation	Less than 3 years of difference to year of study	Less than 6 years difference	Less than 10 years difference	Less than 15 years difference	Age of data unknown or more than 15 years of difference
Completeness	Representative data from an adequate sample of sites over an adequate period	Representative data from a smaller number of sites over an adequate period	Representative data from an adequate number of sites but over a shorter period	Representative data from a small number of sites over a shorter period or inadequate data from adequate number of sites	Unknown or incomplete data from a small number of sites
Reliability	Verified data based on measurements	Verified data based partly on assumptions or non-verified data based on measurements	Unverified data based partly on assumptions	Qualified estimate	Unqualified estimate

$$DQR = \frac{TeR + GR + TiR + C + P + M + Xw * 4}{i + 4}$$

* DQR: Data Quality Rating

- TeR, GR, TiR, C, P, M: data quality indicators in EC-JRC (2011), where TeR is the Technological representativeness, GR is the geographical representativeness, TiR is the time-related representativeness, C is the completeness and P is the precision uncertainty. Xw: weakest quality level obtained, that is equal to the highest numeric value among the data quality indicators. i: number of data quality indicators

Table M.2 : Data Quality Rating (DQR)

Overall data quality rating (DQR)	Overall data quality level	representative color
< 1.6	High quality	
>1.6 to 3	Basic quality	
>3 to 4	Data estimate	

Table M.3 : Data Quality Analysis of the study

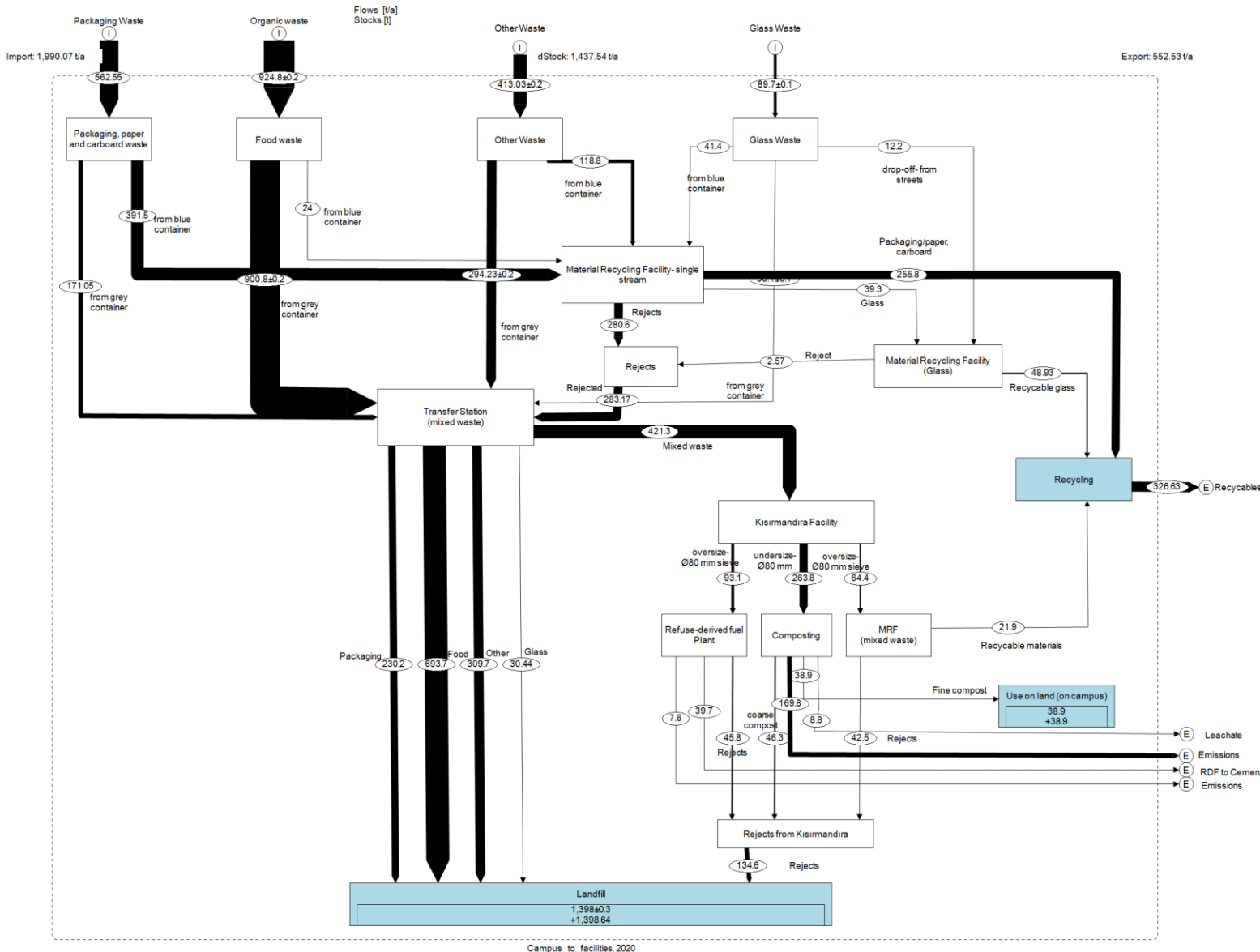
	Reliability	Completeness	Temporal Correlation	Geographical Correlation	Technological Correlation	DQR	Source
Waste characterization and composition							
Waste characterization	1	2	1	1	1	1.6	Field study
Mis-sorted waste fraction	1	2	1	1	1	1.6	Field study
Chemical Composition	1	1	4	4	1	3.0	Riber et al., 2009
Waste Amount							
Waste Amount	1	2	1	1	1	1.6	Field study
Yard waste-amount	2	1	1	1	1	1.6	Personal interview
Drop off glass waste-amount	2	2	1	1	1	1.7	Personal interview
Mis-sorted waste fraction	2	2	1	1	2	1.8	Maçın et al., 2020
Waste prevention-forecasting	2	2	1	1	2	1.8	Maçın et al., 2020
Collection and Transportation							
Fuel consumption	2	1	1	1	1	1.6	IMM, Sezer, 2019
Distances	1	1	1	1	1	1.0	IMM, Sezer, 2019
Waste Management and Facilities							
Composting-TC	1	1	4	1	1	2.7	Sezer and Arkan, 2010
Composting-Electricity consumption	1	1	3	3	1	2.3	Boldrin et al., 2011
Composting-Use on land	2	2	1	4	2	3.0	EASETECH; Cobo et al., 2018
RDF-Diesel consumption	1	1	2	3	2	2.3	Grzesik and Malinowski, 2016
RDF-Electricity consumption	2	2	3	3	2	2.7	Burnley et al., 2011
MRF-TC	1	2	1	4	2	2.9	Pressley et al., 2015
MRF-Electricity consumption	1	2	3	2	1	2.3	Kurdoğlu, 2013; Yıldız-Geyhan, 2016
MRF-market substitution ratios	2	2	1	4	2	3.0	EASETECH; Fernández-Braña et al., 2020
AD-TC	2	1	1	1	1	1.6	Primary source and Akça et al., 2020
AD-Electricity consumption	2	3	2	3	3	2.8	Boldrin et al., 2011
AD-Use on land	2	2	1	4	2	3.0	Cobo et al., 2018
Incineration-TC	2	1	3	3	4	3.2	Bisinella et al., 2021
Incineration-Emissions to air	2	1	3	3	4	3.2	Bisinella et al., 2021
Incineration-Ancillary	2	1	3	3	3	2.7	IMM 2012
Incineration-electricity efficiency	2	1	3	3	3	2.7	IMM 2012
Incineration-heat efficiency	2	1	3	3	3	2.7	IMM, 2012
Landfill -gas collection	2	1	3	3	4	3.2	EASETECH
Leachate Treatment	2	1	3	3	4	3.2	Turkish water institute, 2019
Leachate Amount	3	1	1	3	4	3.1	(Schroeder, et al., 1994; Yang et al., 2015)
Background data							
Electricity mix-TR	2	1	1	1	3	2.2	ecoinvent v3.7.1
Heat mix-TR	2	1	1	1	3	2.2	ecoinvent v3.7.1.
Electricity -forecasting-TR	2	1	1	1	3	2.2	EPDK, 2020 ;SHURA, 2020

*TC is “transfer coefficient”



APPENDIX N: MFA EXAMPLES OF THE WASTE MANAGEMENT SCENARIOS

MFA studies were conducted by using STAN software. MFA figure of were shown in this section.



*Some part of the KF output is fed into RDF facility (near to KF) to substituted petrocoke hence RDF unit was drawn under the KF in MFA.

Figure N.1 : MFA-scenario 1-a

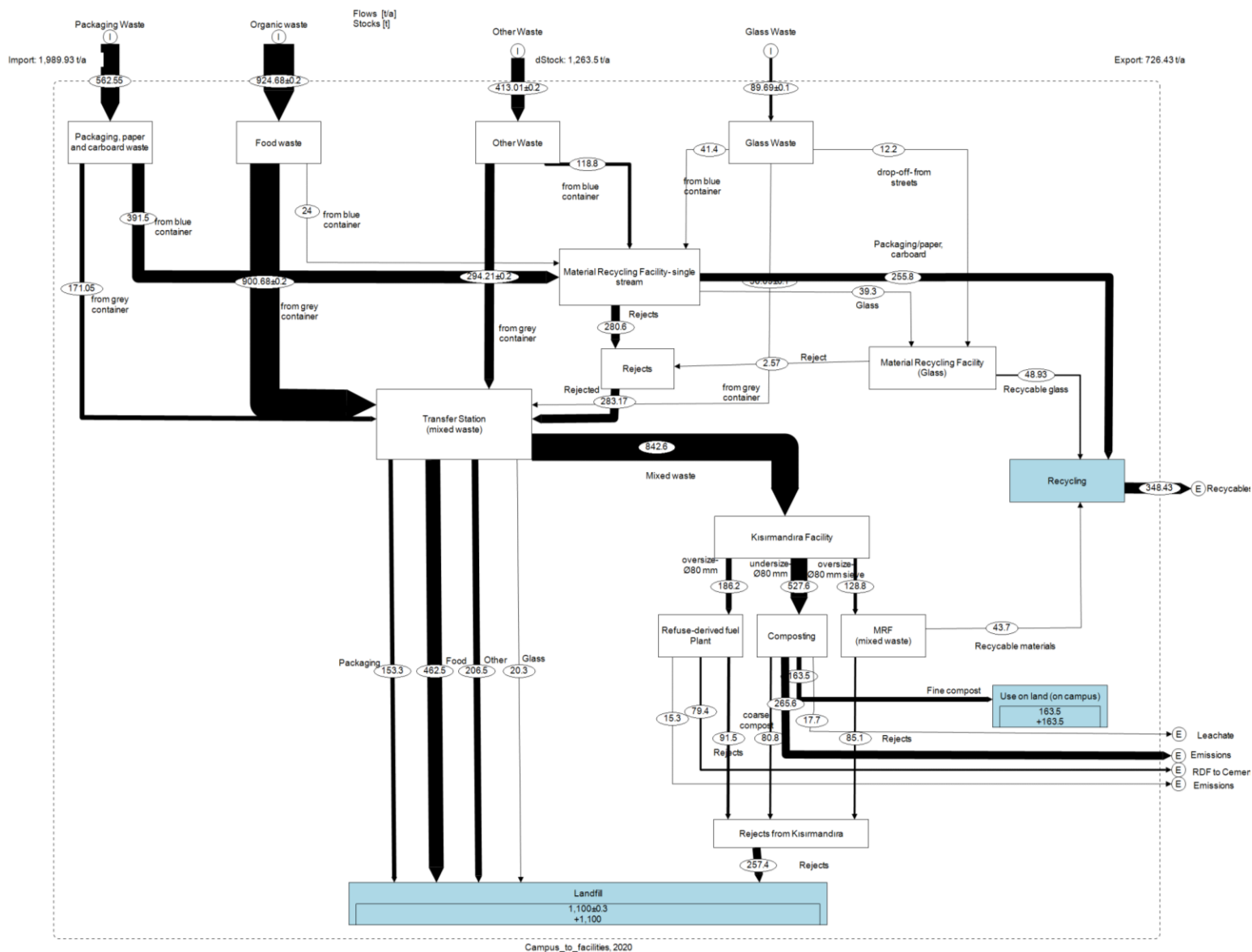


Figure N.2 : MFA-scenario 1-b

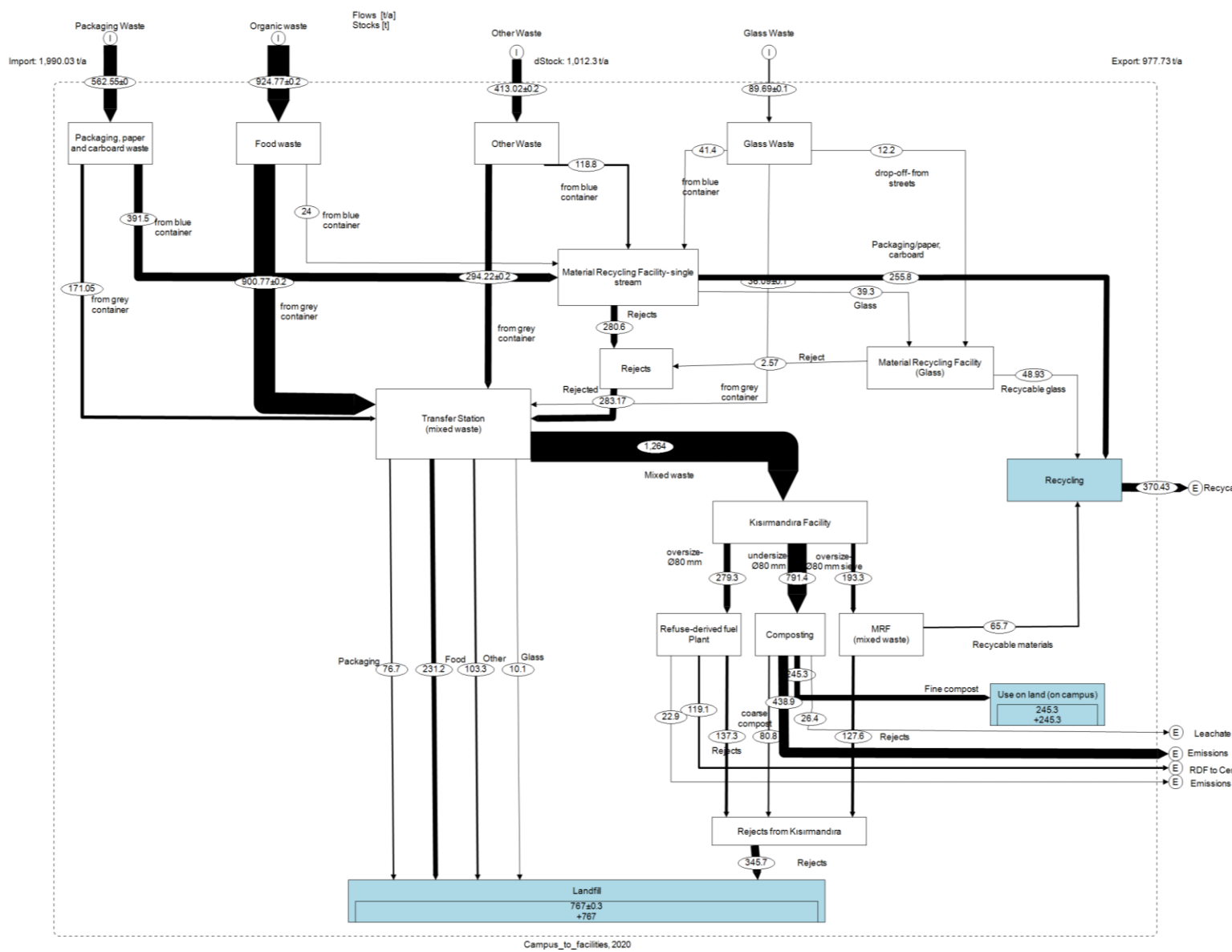


Figure N.3 : MFA-scenario 1-c

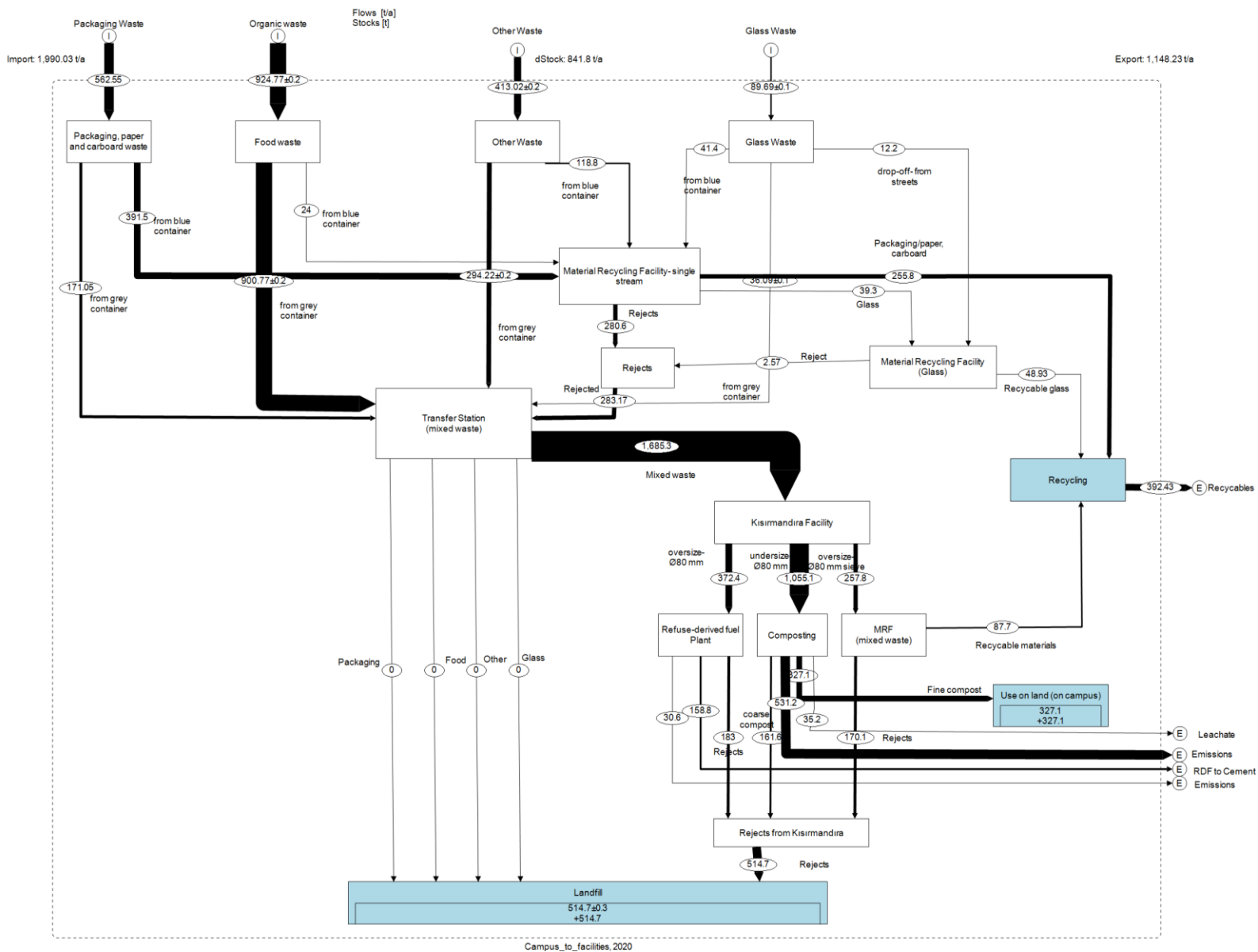


Figure N.4 : MFA-scenario 1-d

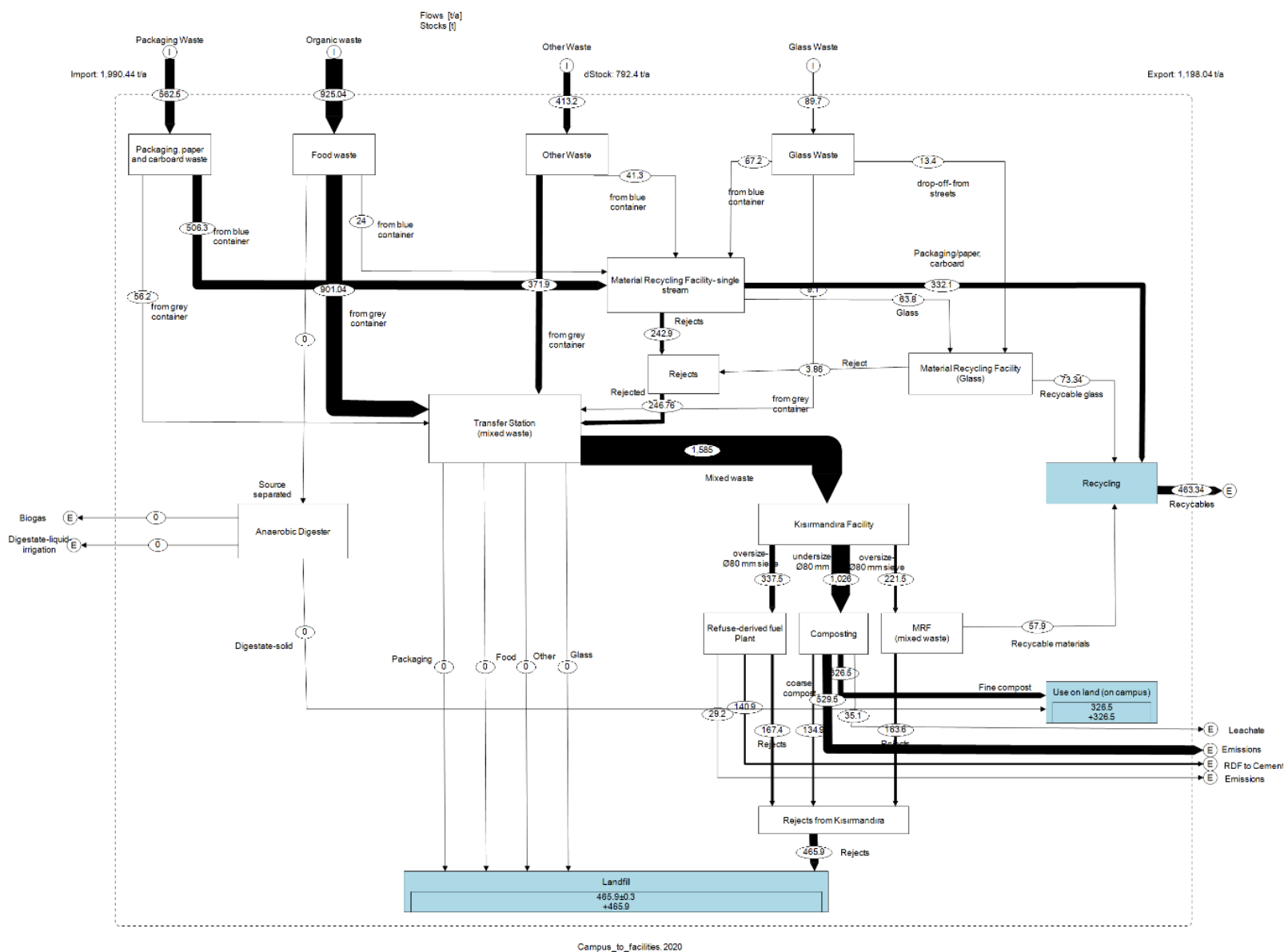


Figure N.5 : MFA-scenario 2

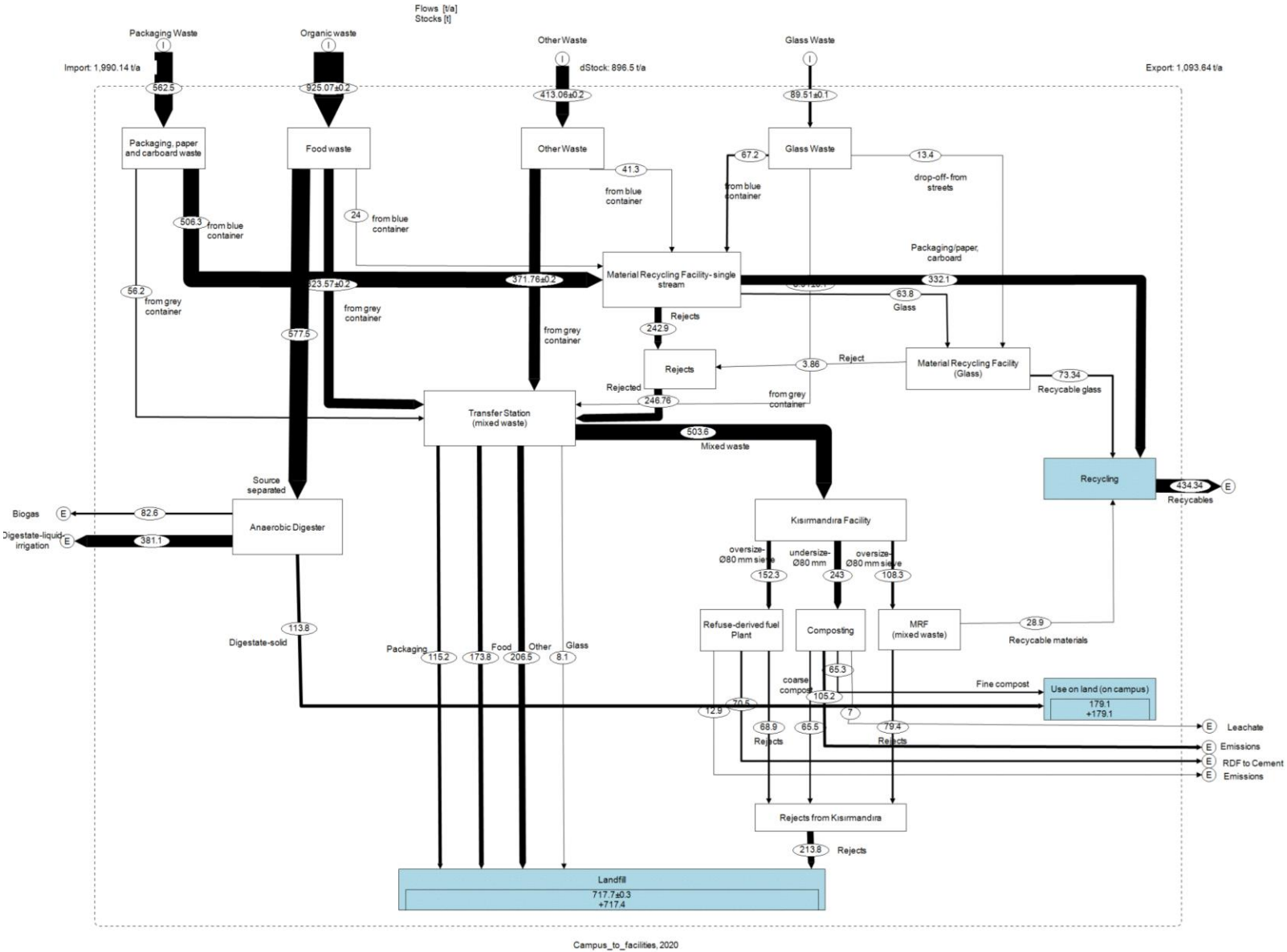


Figure N.7 : MFA-scenario 3-b

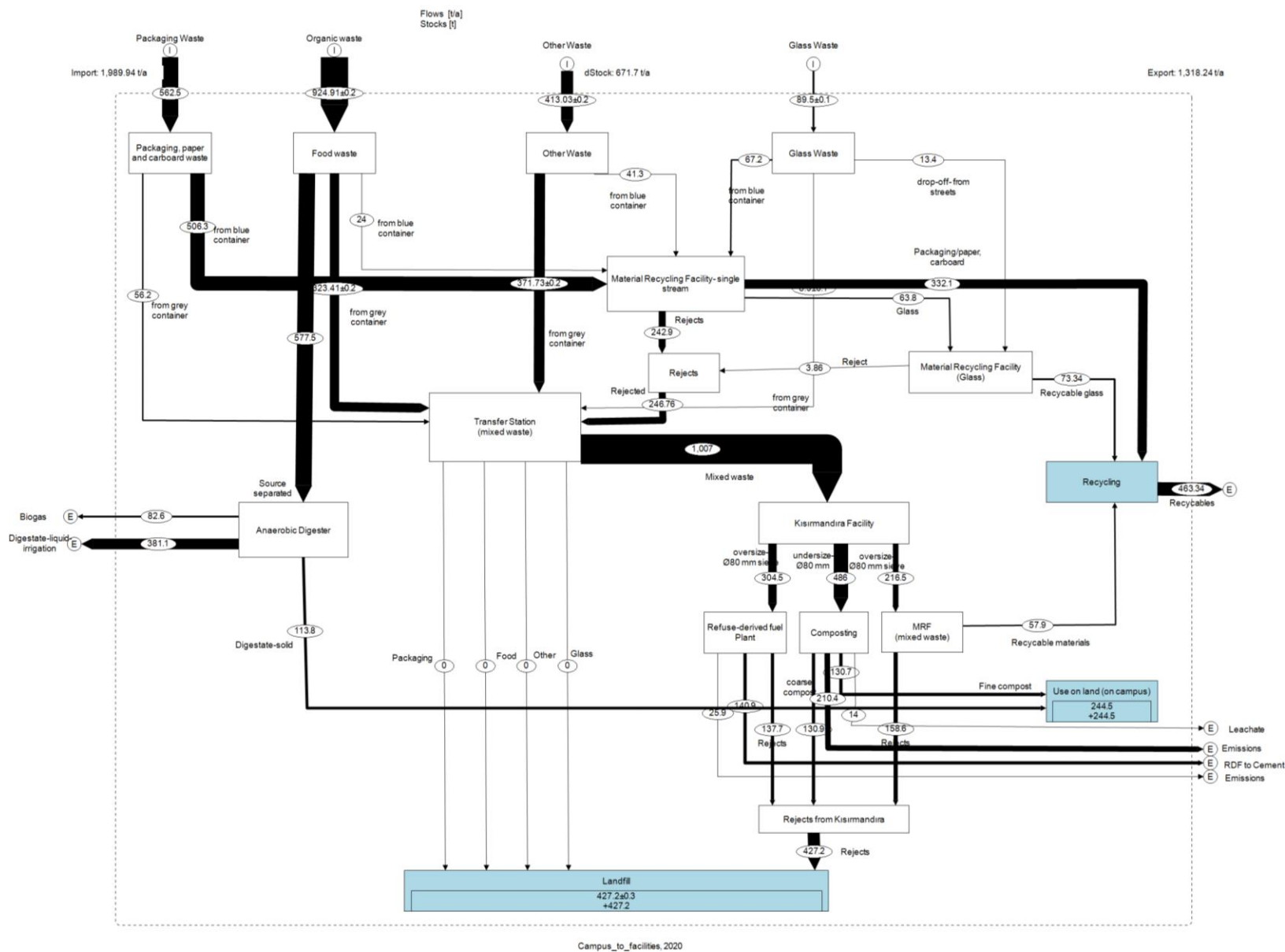


Figure N.9 : MFA-scenario 3-d

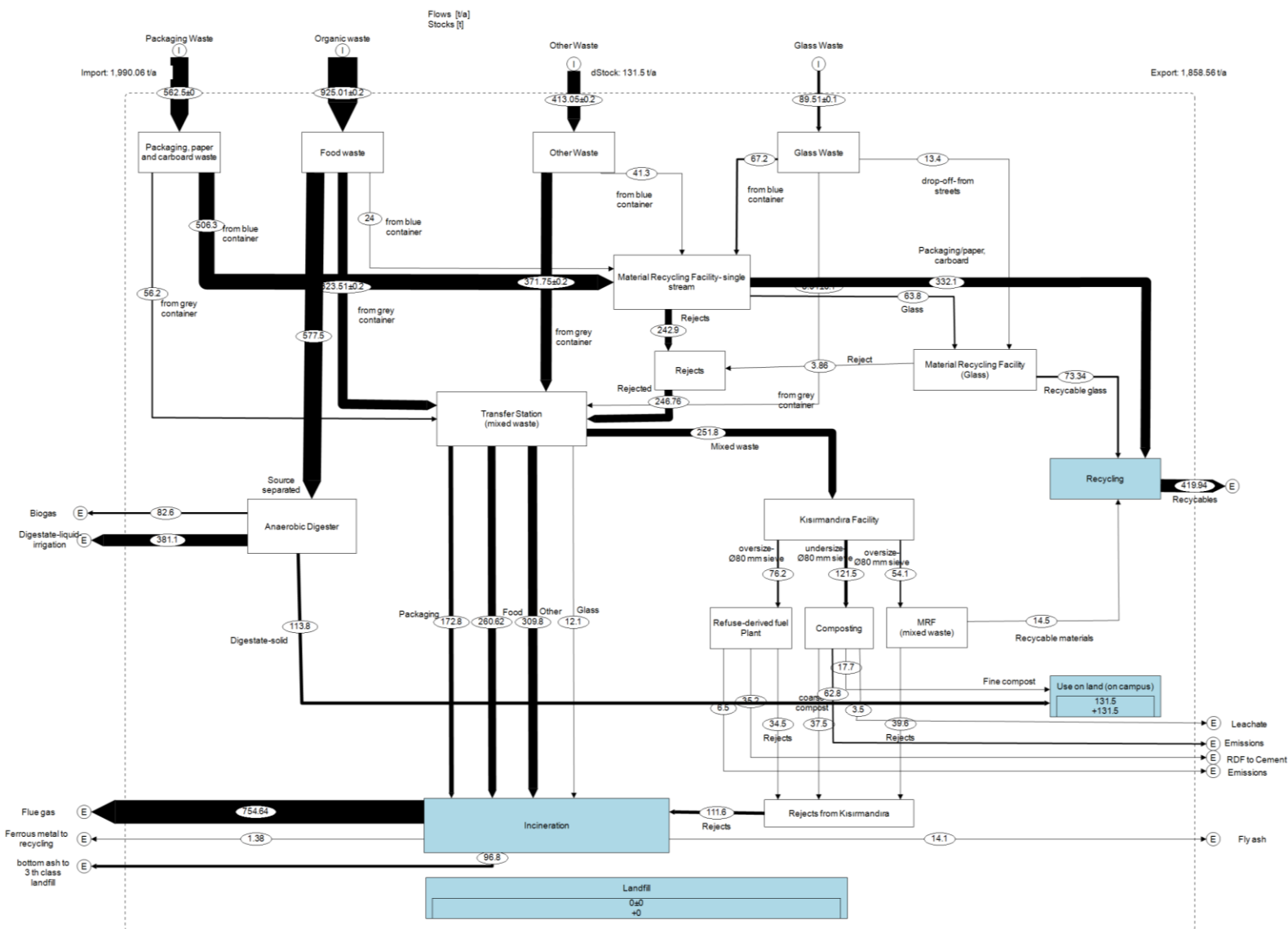


Figure N.10 : MFA-scenario 4-a

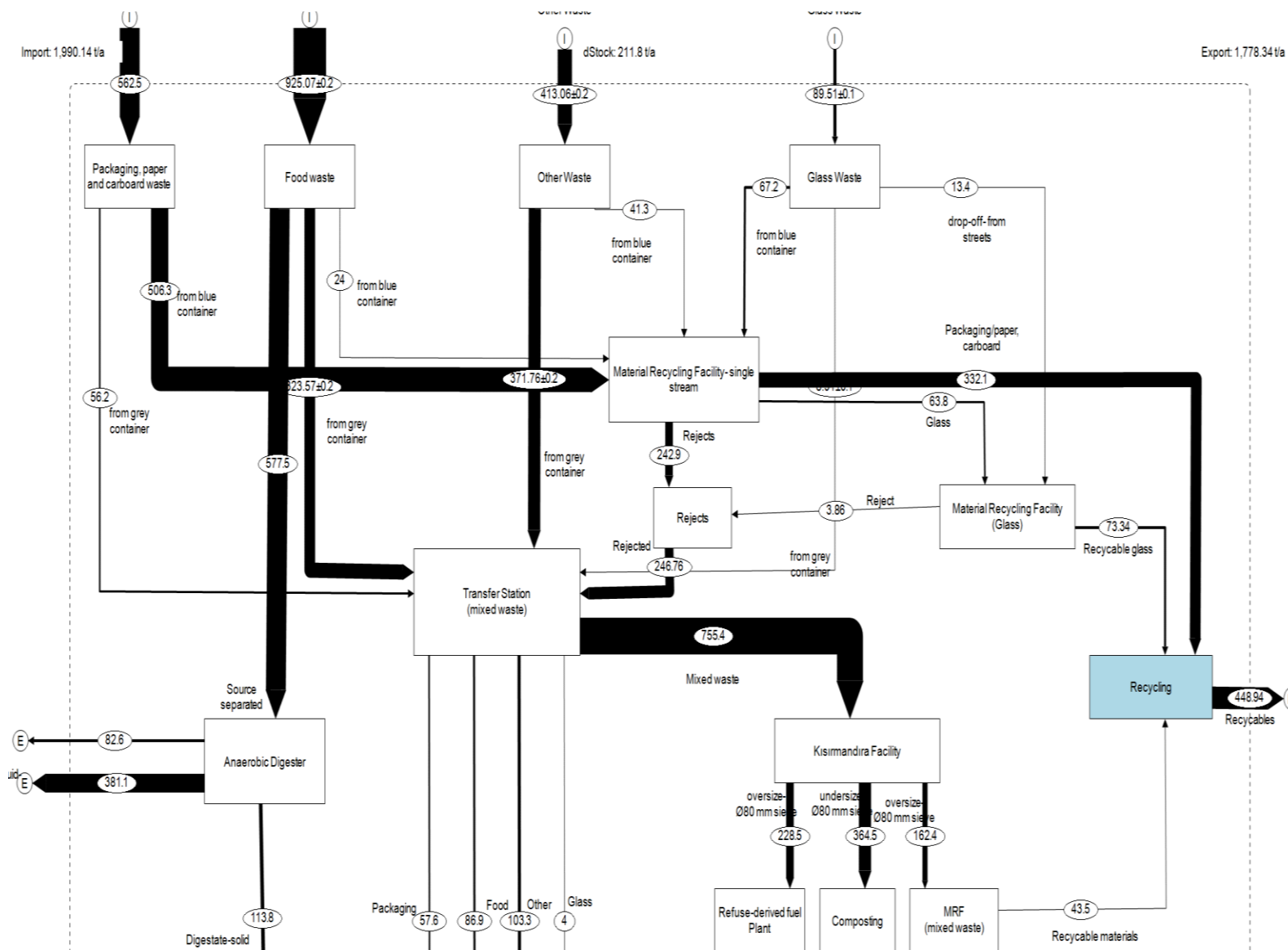


Figure N.12 : MFA-scenario 4-c

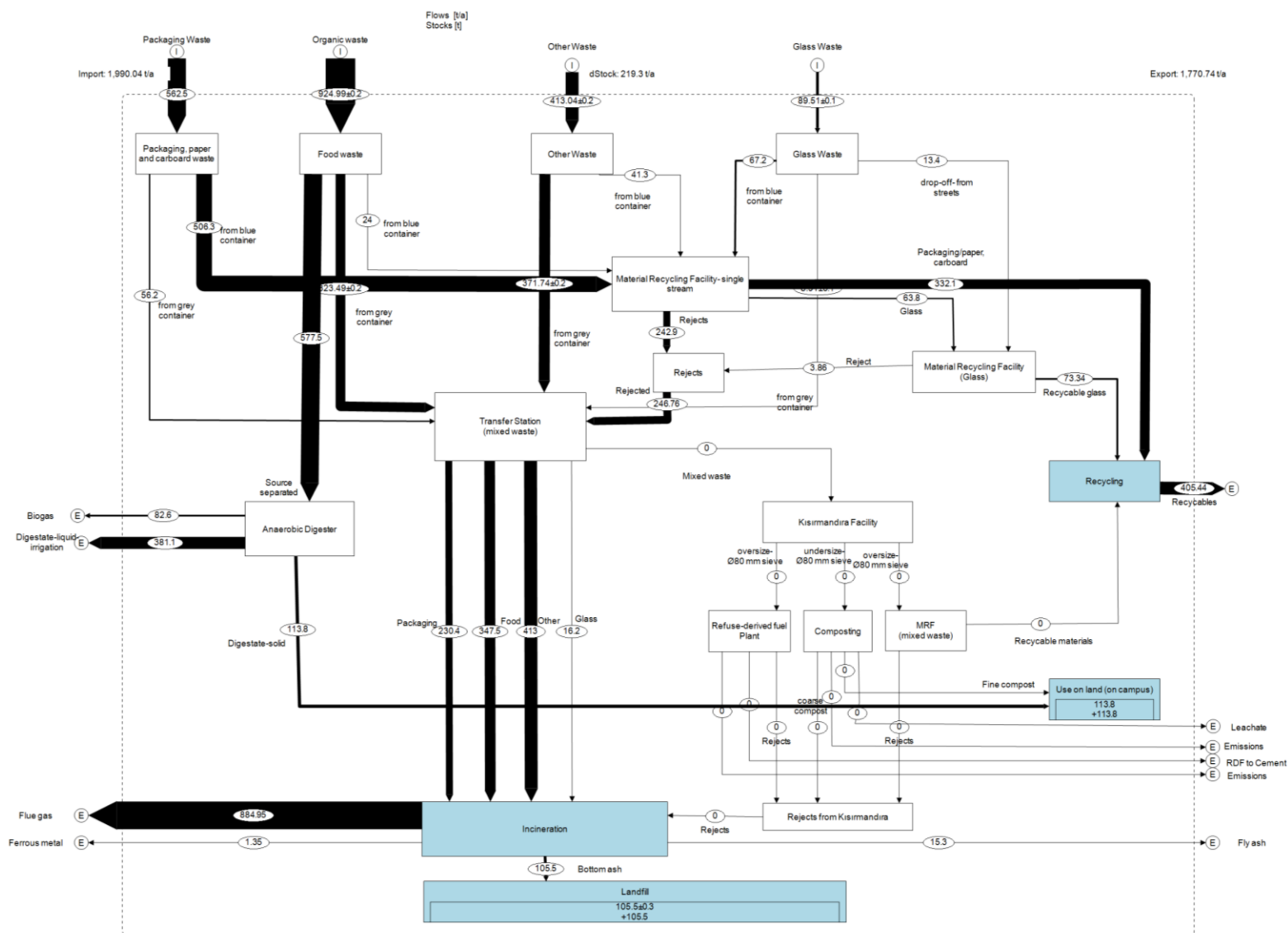


Figure N.14 : MFA-scenario 5

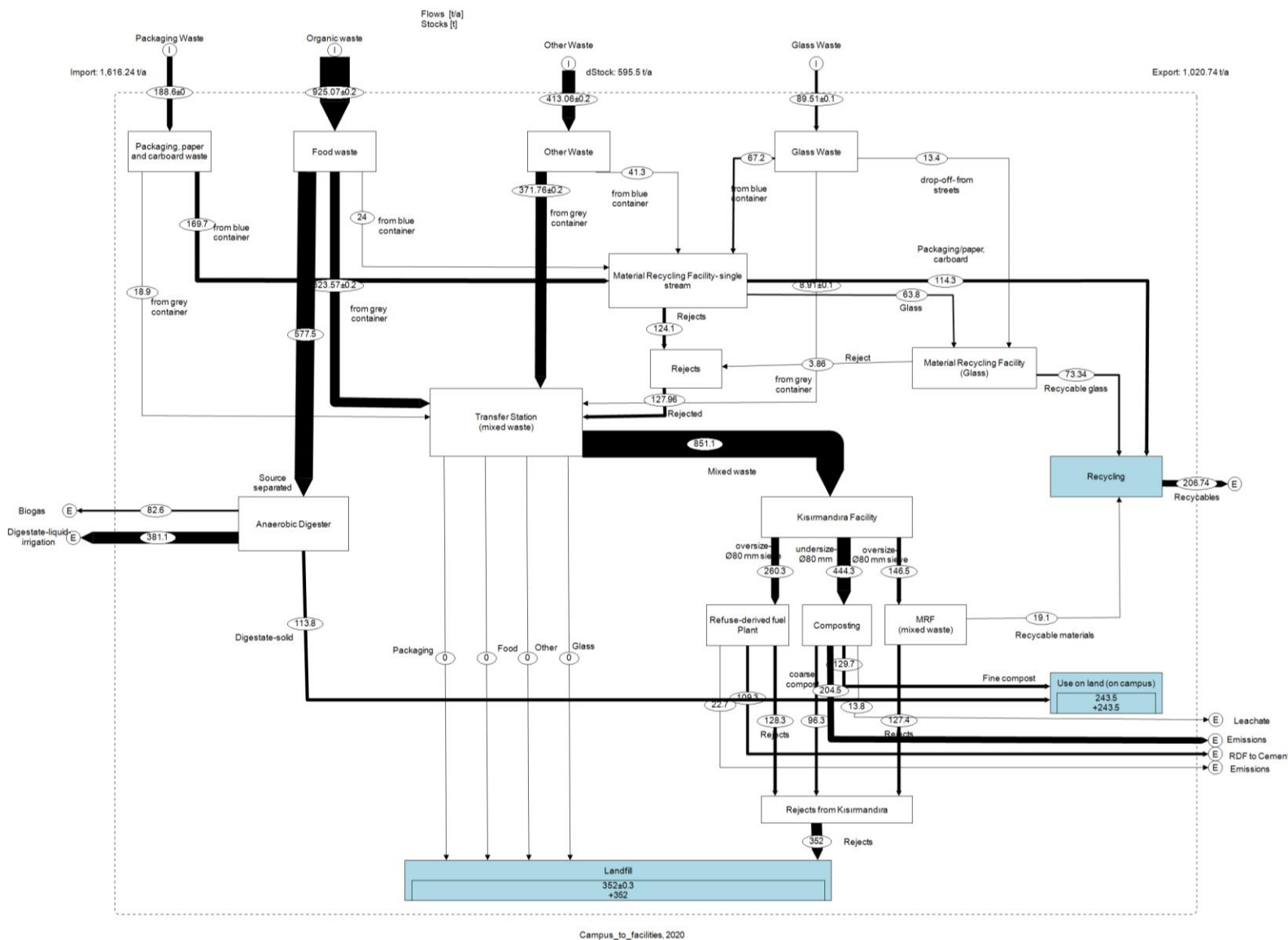


Figure N.15 : MFA-scenario 6

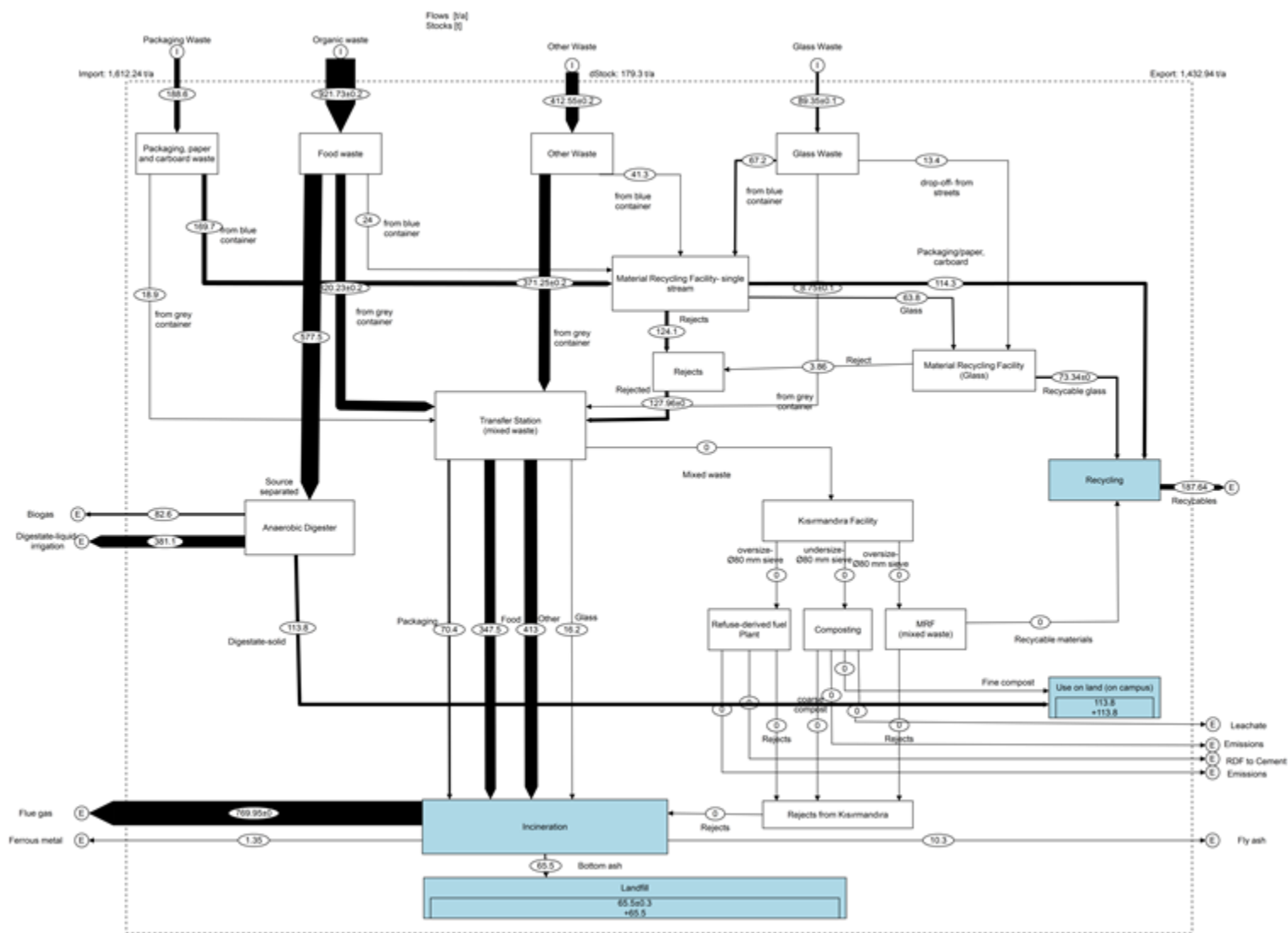


Figure N.16 : MFA-scenario 7

APPENDIX O: CIRCULARITY CALCULATIONS

Example of circularity calculation and related data shown in this section.

Table A.47 : Recycled and recovered materials in the WM system (tonne/year)

Scenario	Secondary Material (tonne/year)	Compost (tonne/year)	AD (biogas, digestate) (tonne/year)	RDF used instead of petrocoke (tonne/year)
1-a	326.0	38.9	-	39.7
1-b	348.0	163.5	-	79.4
1-c	370.0	245.3	-	119.1
1-d	392.0	327.1	-	158.8
2	420.0	326.5	-	140.9
3-a	420.0	17.7	577.5	35.2
3-b	434.0	65.3	577.5	70.5
3-c	449.0	98.0	577.5	105.7
3-d	463.0	130.7	577.5	140.9
4	206.4	129.7	577.5	109.4
5-a	420.0	17.7	577.5	35.2
5-b	434.0	65.3	577.5	70.5
5-c	449.0	98.0	577.5	105.7
5-d	463.0	130.7	577.5	140.9
6	405.0	-	577.5	-
7	206.4	-	577.5	-

$$\text{Circularity (\%)} = \frac{\text{Recycled and recovered materials in the WM system}}{\text{Total waste amount}}$$

$$\text{Circularity for scenario 1 - a (\%)} = \frac{326 + 38,9 + 39,7}{1990,5} = 20.3\%$$

Circularity was determined by the proportion of recycled and recovered material in the WM system relative to the total waste amount. For packaging waste, circularity was calculated based on the substitution of materials from MRFs. Energy recovery from the incineration of fossil or biomass waste is not considered as circularity. In the case of biodegradable waste, there were two destinations: AD and composting.



APPENDIX P: CONTRIBUTION ANALYSIS RESULTS

- Climate change impact changes between 609 kgCO₂-eq (scenario 1-a) to -1000 kgCO₂-eq (scenario 7) FU waste. Landfilling is the main contributor to the impact, followed by composting. The A.D. facility has net savings since it is accepting source separated food waste and also has additional savings from biogas utilization which are larger than the release from the A.D. facility.
- The net savings of ozone depletion category increases with landfill diversion except scenario 4. Savings coming from incineration are lower in scenario 4-d. Because KF have more recycling and this lead to decrease in plastic waste input to incineration which has high energy recovery potential.
- MRF and incineration plant have major savings in human toxicity cancer category. But similar to climate change and ozone depletion net impacts decrease from scenario 4-a to 4-d. Waste prevention has biggest saving however, since less waste sent to MRF in scenario 6, saving coming from MRF decreases.
- Human toxicity non-cancer category has significant burdens coming from collection and transportation but it has net negative results in all scenarios thanks to savings coming from MRF and composting.
- Particulate matter category results have net savings except incineration scenarios (4 and 6) and scenario 1-d. Major burdens coming from incineration plant, collection and composting plant while major savings coming from energy recovery and aluminum recycling from incineration plant.
- Photochemical ozone category has net burden in scenario 1,4 and 6 while others have net savings. Burdens coming from incineration, landfill, collection and transportation lead to increase while MRF (mostly; plastic (PE) PET, paper cardboard and aluminium recycling) have significant amount of saving.
- The net impact of terrestrial acidification is negative for all scenarios except 1-d. Savings mostly coming from recycling. Also, incineration has significant savings because it is substituting the Turkish electricity mix which is heavily based on fossil fuels. Similar to global warming category A.D. has savings (thanks to energy recovery) while composting has burden.
- The major contributors of the terrestrial eutrophication impact are collection, composting and A.D. and savings are mostly coming from recycling and waste prevention.
- The A.D. has significant impact on eutrophication freshwater due to digestate use on land as fertilizer, which have losses due to runoff and leaching to the groundwater. Scenario 2, 3,4,5 and 6 have net burdens while scenario 1 and 2 have net savings since they don't have A.D.
- The major contributors of the marine eutrophication impact are composting and A.D. and savings are mostly coming from recycling and waste prevention. All scenarios have net burdens.
- The A.D has net savings in ecotoxicity freshwater category due to substitution of electricity and incineration has the largest savings for same reason. Transportation has significant burdens while net result of the category has savings in all scenarios.

The net impacts of different scenarios were presented in **Table P.1**

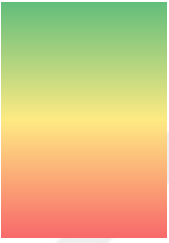
Table P.1 : The net results of different impact categories-FU

	1-a	1-b	1-c	1-d	2	3-a	3-b	3-c	3-d	4	5-a	5-b	5-c	5-d	6	7
Climate change	6.09E+05	4.39E+05	2.70E+05	1.01E+05	2.45E+04	6.13E+04	-8.77E+03	-7.88E+04	-1.49E+05	-6.14E+05	-6.05E+05	-5.96E+05	-5.87E+05	-6.23E+05	-6.31E+05	-1.00E+06
Ozone depletion	-1.16E-02	-1.18E-02	-1.19E-02	-1.20E-02	-1.29E-02	-1.48E-02	-1.52E-02	-1.56E-02	-1.60E-02	-6.46E-02	-6.31E-02	-6.15E-02	-6.00E-02	-6.61E-02	-1.75E-01	-2.21E-01
Human toxicity	-4.98E-03	-4.89E-03	-4.80E-03	-4.71E-03	-6.82E-03	-8.43E-03	-8.34E-03	-8.26E-03	-8.17E-03	-1.17E-02	-1.15E-02	-1.13E-02	-1.12E-02	-1.19E-02	-1.04E-02	-1.31E-02
Human toxicity, non-cancer	-7.68E-03	-1.20E-02	-1.63E-02	-2.06E-02	-1.97E-02	-6.80E-03	-9.84E-03	-1.29E-02	-1.59E-02	-2.34E-02	-2.60E-02	-2.86E-02	-3.12E-02	-2.08E-02	-3.38E-02	-3.97E-02
Particulate matter	-1.15E+02	-7.63E+01	-3.74E+01	1.54E+00	-5.08E+00	-2.12E+02	-1.94E+02	-1.77E+02	-1.59E+02	2.56E+02	2.39E+02	2.23E+02	2.06E+02	2.72E+02	-3.87E+02	-6.66E+02
Photochemical ozone formation	4.37E+02	2.66E+02	9.45E+01	-7.66E+01	-3.78E+02	-2.51E+02	-3.13E+02	-3.75E+02	-4.37E+02	3.42E+02	3.07E+02	2.73E+02	2.39E+02	3.76E+02	-1.57E+03	-2.39E+03
Terrestrial acidification	-6.91E+02	-4.55E+02	-2.20E+02	1.56E+01	-3.05E+02	-1.61E+03	-1.51E+03	-1.41E+03	-1.32E+03	-3.89E+03	-3.70E+03	-3.51E+03	-3.33E+03	-4.08E+03	-2.95E+03	-4.85E+03
Eutrophication Terrestrial	2.00E+03	2.49E+03	2.97E+03	3.45E+03	2.71E+03	3.79E+02	5.47E+02	7.14E+02	8.81E+02	-3.50E+03	-3.07E+03	-2.63E+03	-2.20E+03	-3.94E+03	-2.99E+03	-6.34E+03
Eutrophication Freshwater	-1.71E+01	-1.41E+01	-1.11E+01	-8.12E+00	-8.42E+00	3.53E+02	3.54E+02	3.55E+02	3.56E+02	3.24E+02	3.26E+02	3.28E+02	3.30E+02	3.23E+02	3.42E+02	3.19E+02
Eutrophication Marine	4.80E+02	6.61E+02	8.42E+02	1.02E+03	9.60E+02	1.02E+03	1.08E+03	1.14E+03	1.20E+03	5.89E+02	6.84E+02	7.79E+02	8.74E+02	4.93E+02	7.38E+02	2.31E+02
Ecotoxicity freshwater	-3.03E+04	-2.96E+04	-2.89E+04	-2.82E+04	-2.81E+04	-6.28E+04	-6.38E+04	-6.49E+04	-6.59E+04	-1.77E+05	-1.76E+05	-1.75E+05	-1.74E+05	-1.78E+05	-3.06E+05	-3.87E+05

APPENDIX Q: PERTURBATION AND SCENARIO ANALYSIS

Contribution analysis and data quality assessment results were used to select relevant parameters for perturbation analysis. (Clavreul et al., 2012) was followed for sensitivity analysis. A perturbation analysis was performed to understand effect of small change of parameters on overall results by increasing 10% each parameter.

Table Q.1 : Sensitivity level of parameters

Sensitivity Ratio	Sensitivity level	representative color
< 0 to 0.5	Not sensitive	
> 0.5 to 0.8	Sensitive	
> 0.8	Very sensitive	

Sensitivity ratios were calculated as:

$$\text{Sensitivity Ratios (SR)} = \frac{\frac{\Delta \text{ Results}}{\text{Initial Result}}}{\frac{\Delta \text{ Parameter}}{\text{Initial Parameter}}}$$

After calculating sensitivity ratio (SR) values normalized sensitivity ratios (NSR) were calculated to compare SR results of different impact categories.

Normalized sensitivity ratios were calculated as:

$$\text{Normalised Sensitivity Ratio (NSR)} = \frac{\text{abs}(SR)}{\max(\text{abs}(SR))}$$

Table Q.2: Sensitivity ratios of scenarios for climate change category

Process name	Scenario 1-a	Scenario 3-a	Scenario 4-a
Activated carbon	0.00	0.00	0.01
Substitution Aluminium, Market response	0.33	0.76	0.75
Substitution Aluminium Market ratio	0.12	0.30	0.18
100%Incineraiton			
Arsenic, ion	0.00	0.00	0.00
As	0.00	0.00	0.00
Barium	0.00	0.00	0.00
biochemical methane potential (yield) (%) animal	0.00	0.17	0.10
biochemical methane potential (yield) (%) garden	0.00	0.01	0.01
biochemical methane potential (yield) (%) vegetable	0.00	0.12	0.07
Calcium, ion	0.00	0.00	0.00
Limestone (CaCO ₃)	0.00	0.00	0.00
Cobalt	0.00	0.00	0.00
Cadmium, ion	0.00	0.00	0.00
Cd	0.00	0.00	0.00
Chloride	0.00	0.00	0.00
Cl	0.00	0.00	0.00
Carbon monoxide, fossil	0.00	0.00	0.00
N-plant uptake	0.00	0.00	0.00
P-plant uptake	0.00	0.00	0.00
Chromium, ion	0.00	0.00	0.00
Cr	0.00	0.00	0.00
Copper, ion	0.00	0.00	0.00
Cu	0.00	0.00	0.00
Oxidation in top cover, daily (-10%)	0.41	0.27	0.00
Oxidation in top cover, daily (-10%)	0.00	0.00	0.00
Oxidation in top cover, daily (-10%)	0.00	0.00	0.00
N-plant uptake	0.00	0.00	0.00
P-plant uptake	0.00	0.00	0.00
Marginal Electricity Process	0.00	0.00	0.00
Marginal Electricity Process	0.00	0.00	0.00
Marginal Electricity Process	0.08	0.13	0.08
Marginal Electricity Process	0.06	0.09	0.05
Marginal Electricity Process	0.01	0.02	0.01
Marginal Electricity Process	0.05	0.14	0.08
Marginal Electricity Process	0.01	0.01	0.01
Electricity consumption	0.00	0.00	0.11
Electricity recovery	0.00	0.00	1.00
Fluoride	0.00	0.00	0.00
Iron, ion	0.00	0.00	0.00
N-plant uptake	0.00	0.00	0.00
P-plant uptake	0.00	0.00	0.00
Oxidation in top cover, final (-10%)	0.00	0.00	0.00
Oxidation in top cover, final (-10%)	0.00	0.00	0.00
Oxidation in top cover, final (-10%)	0.00	0.00	0.00
Substitution of Glass	0.13	0.29	0.17

Table Q.2 (cont): Sensitivity ratios of scenarios for climate change category

Hydrogen chloride	0.00	0.00	0.00
Hydrogen chloride (HCl), gas	0.00	0.00	0.00
Heat recovery	0.00	0.00	0.38
Hydrogen fluoride	0.00	0.00	0.00
Mercury	0.00	0.00	0.00
Hg	0.00	0.00	0.00
Oxidation in top cover, intermediate (-10%)	0.00	0.00	0.00
Oxidation in top cover, intermediate (-10%)	0.12	0.09	0.00
Oxidation in top cover, intermediate (-10%)	0.00	0.00	0.00
Electrical efficiency of the gas collected (+10%)	0.11	0.09	0.00
Heat efficiency of the gas collected (+10%)	0.31	0.27	0.00
Leaking CH4 (+10%)	0.00	0.00	0.00
leaking of gas	0.63	0.41	0.00
leaking of gas	1.00	0.76	0.00
leaking of gas	0.89	1.00	0.00
Limestone substitution (CaCO3)	0.00	0.00	0.00
Magnesium	0.00	0.00	0.00
Manganese	0.00	0.00	0.00
Manganese	0.00	0.00	0.00
Molybdenum	0.00	0.00	0.00
Mo	0.00	0.00	0.00
Sodium hydroxide (NaOH)	0.00	0.00	0.00
Nickel	0.00	0.00	0.00
Ni	0.00	0.00	0.00
Nickel, ion	0.00	0.00	0.00
Nitrogen oxides	0.00	0.00	0.00
PAH, polycyclic aromatic hydrocarbons	0.00	0.00	0.00
Lead	0.00	0.00	0.00
Pb	0.00	0.00	0.00
Substitution PET, market ratio 81%	0.08	0.24	0.14
Particulates, > 10 um	0.00	0.00	0.00
Polyethylene high density granulate	0.21	0.56	0.33
S	0.00	0.00	0.00
Antimony	0.00	0.00	0.00
Sb	0.00	0.00	0.00
Selenium	0.00	0.00	0.00
Silicon	0.00	0.00	0.00
Sulfur dioxide	0.00	0.00	0.00
Substitution steel, market ratio 85%	0.08	0.14	0.08
Steel sheets (primary) substitution	0.00	0.00	0.01
Dioxins, measured as 2,3,7,8-	0.00	0.00	0.00
Substitution of virgin cardboard, market response 85%	0.32	0.50	0.29
Substitution virgin newspaper	0.27	0.48	0.28
Zinc, ion	0.00	0.00	0.00

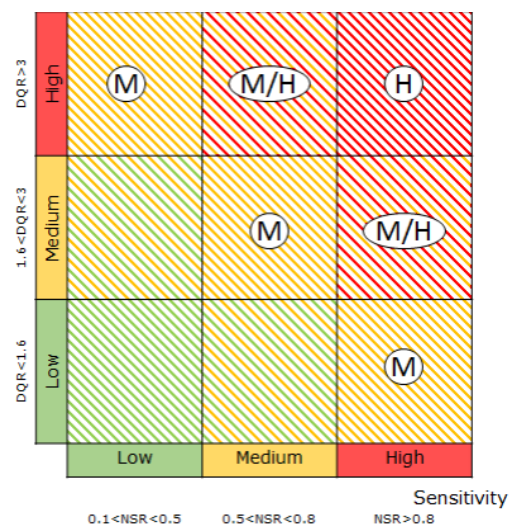


Figure Q.1 : Critical parameters classification by using NSR and DQA figure by (Andreasi Bassi et al., 2017)

Table Q.2 : Criticality of the parameters

Overall data quality rating (DQR)	Representative color in the
High (H)	
High/Medium (M/H)	
Medium (M)	
Low (L)	

Substitution ratio of glass, PET, aluminum, cardboard, paper and plastic in MRFs, biochemical methane potential of AD input, electricity and heat recovery of incineration plant, gas collection and CHP efficiencies in landfill are most **relevant parameters** in the model in the climate change category.

Table Q.3: Normalized sensitivity ratios (NSR) and Data Quality Assessment (DQA) classification

Parameter	NSR	DQA	Relevant/Critical Parameter
Marginal Electricity Process	low	medium	
Marginal Electricity Process	low	medium	
Marginal Electricity Process	low	medium	
Marginal Electricity Process	low	medium	
Marginal Electricity Process	low	medium	
Marginal Electricity Process	low	medium	
Marginal Electricity Process	low	medium	
Substitution of Glass Manufacturing	medium	medium	Relevant
Substitution steel, market ratio 85%	medium	medium	Relevant
Substitution PET, market ratio 81%	medium	medium	Relevant
Substitution Aluminium Market ratio 100%	high	medium	Relevant
Substitution virgin newspaper and CTMP, Market response 100%	medium	medium	Relevant
Substitution of virgin cardboard, market response 85%	medium	medium	Relevant
Polyethylene high density granulate (PE-HD), 1999, RER, ELCD	medium	medium	Relevant
As	low	medium	
Cd	low	medium	
Cl	low	medium	
Cr	low	medium	
Cu	low	medium	
Hg	low	medium	
Mo	low	medium	
Ni	low	medium	
Pb	low	medium	
S	low	medium	
Sb	low	medium	
Zn	low	medium	
Electricity recovery	high	medium	Relevant
Heat recovery	medium	medium	Relevant
Electricity consumption	low	medium	
Sodium hydroxide (NaOH)	low	medium	
Activated carbon	low	medium	
Limestone (CaCO3)	low	medium	
Ammonia (NH3)	low	medium	
Hydrogen chloride (HCl), gas	low	medium	
Ammonia	low	medium	
Antimony	low	medium	
Arsenic, ion	low	medium	
Barium	low	medium	
Cadmium, ion	low	medium	
Calcium, ion	low	medium	
Carbon monoxide, fossil	low	medium	
Chloride	low	medium	
Chromium, ion	low	medium	
Cobalt	low	medium	
Copper, ion	low	medium	
Dioxins, measured as 2,3,7,8-tetrachlorodibenzo-p-dioxin	low	medium	

Table Q.3 (cont):: Normalized sensitivity ratios (NSR) and Data Quality Assessment (DQA) classification

Fluoride	low	medium	
Hydrogen chloride	low	medium	
Hydrogen fluoride	low	medium	
Iron, ion	low	medium	
Lead	low	medium	
Magnesium	low	medium	
Manganese	low	medium	
Manganese	low	medium	
Mercury	low	medium	
Molybdenum	low	medium	
Nickel	low	medium	
Nickel, ion	low	medium	
Nitrogen oxides	low	medium	
PAH, polycyclic aromatic hydrocarbons	low	medium	
Particulates, > 10 um	low	medium	
Selenium	low	medium	
Silicon	low	medium	
Sulfur dioxide	low	medium	
Zinc, ion	low	medium	
Steel sheets (primary) substitution	low	medium	
Substitution Aluminium, Market response 100%	low	medium	
Limestone substitution (CaCO3)	low	medium	
leaking of gas	high	high	Relevant
Oxidation in top cover, daily (-10%)	medium	high	Relevant
Oxidation in top cover, intermediate (-10%)	low	high	Relevant
Oxidation in top cover, final (-10%)	low	high	Relevant
leaking of gas	high	high	Relevant
Oxidation in top cover, daily (-10%)	low	high	Relevant
Oxidation in top cover, intermediate (-10%)	low	high	Relevant
Oxidation in top cover, final (-10%)	low	high	Relevant
leaking of gas	high	high	Relevant
Oxidation in top cover, daily (-10%)	low	high	Relevant
Oxidation in top cover, intermediate (-10%)	low	high	Relevant
Oxidation in top cover, final (-10%)	low	high	Relevant
Electrical efficiency of the gas collected (+10%)	medium	high	Relevant
Heat efficiency of the gas collected (+10%)	medium	high	Relevant
biochemical methane potential (yield) (%) garden	low	medium	
biochemical methane potential (yield) (%) vegetable	low	medium	
biochemical methane potential (yield) (%) animal	low	medium	
Leaking CH4 (+10%)	medium	medium	Relevant
N-plant uptake	low	medium	
P-plant uptake	low	medium	
N-plant uptake	low	medium	
P-plant uptake	low	medium	
N-plant uptake	low	medium	
P-plant uptake	low	medium	

APPENDIX R: SCENARIO SENSITIVITY ANALYSIS

Sensitivity analysis were done for two different scenarios. First scenario sensitivity analysis was considered change in the electricity mix by year 2030 (SHURA, 2020). This scenario (yellow bars in the following figures- characterized-2030 balance) based on balanced implementation of different policy objectives such as carbon emission reduction, of domestic resources and cheap electricity. Second scenario sensitivity analysis (green bars in the following figures- characterized-urban character-2020) was based on change in the waste characterization. Istanbul waste characterization (Ak and Sezer, 2020) was used instead of campus waste. It is important to know that urban waste characterization scenario's MFA results are difference from the campus waste's MFAs. The comparison of original scenario (orange bars in the following figures- characterized all facilities) and two scenario sensitivity analysis were given in this section.



Figure R.1 : Climate change (kg CO₂/FU)

Under the characterized-2030 balance scenario, fossil source (coal and natural gas) use will decrease from 57.7% to 41.7% while the electricity coefficient changes from 0.55 kg CO₂ eq to 0.34 kg CO₂ by 2030. Sensitivity scenario analysis result shows net landfill emissions are higher in 2030 since emissions are same but the electricity generation from landfill gas is substituting less fossil fuels in 2030. The urban waste characterization scenario's MFA results are difference from the campus waste's MFAs. This effect clearly seen in waste prevention scenario (scenario 4) which result with more plastic bag reduction at source. In addition, urban waste characterization scenario has lower net burdens in scenario 1-a to 1-d compared to other two. Because of C_{bio} amount in landfilled waste is less even though total food waste amount is higher than the campus waste. Also, in scenario 6, urban waste characterization scenario has less savings due to high amount of organic input of the incineration plant.

Savings mostly coming from waste prevention and incineration while burdens coming from collection and transportation. Hence characterized all facilities and 2030 energy mix

scenarios have similar results while urban waste character scenarios has more burdens in scenario 1-a-b-c-d and scenario 2 due to composting plant.



APPENDIX S: UNCERTAINTY ANALYSIS

Uncertainty analysis was performed with the Monte Carlo simulation. Each scenario was analyzed by using the normal distribution with 10000 iterations. Uncertainty results shows high variance, mostly due to limited data about gas collection efficiency. Hence highest uncertainty was observed in scenario 1-a which has highest landfilled waste amount.

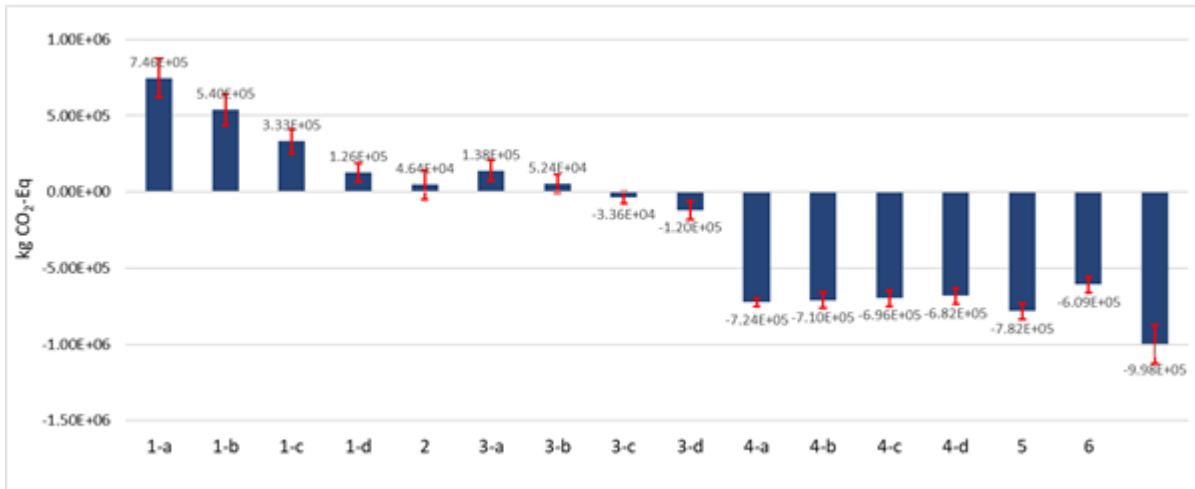


Figure S.1 : Uncertainty analysis of Ayazaga campus scenario



APPENDIX T: GLOBAL WARMING POTENTIAL 20 YEARS IPPC 2021

There is significant landfill emissions in all scenarios (except scenario 5 and 6 since, they do not have any direct landfill application). In scenario 4 burdens from landfill is less than the savings from other WM facilities.

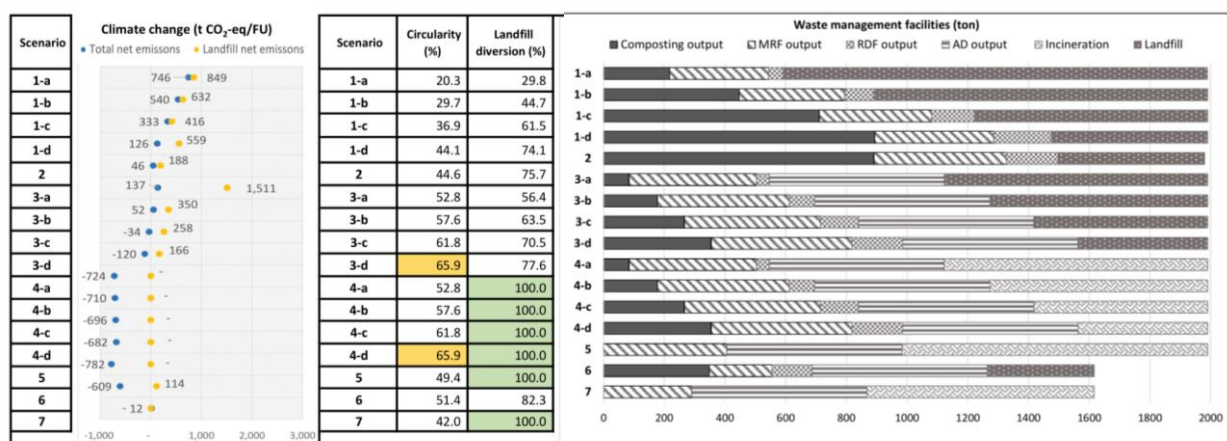


Figure T.1 : Circularity (%) vs landfill diversion (%) and Climate change emissions (kg CO₂-eq/FU)

Net savings were observed in Scenario 4-5 and 6. High landfill burdens in GWP 20 years calculations lead to net burdens in Scenario 1-2 and 3. In addition higher uncertainty (compared to 100 years effects) were calculated due to landfill gas.

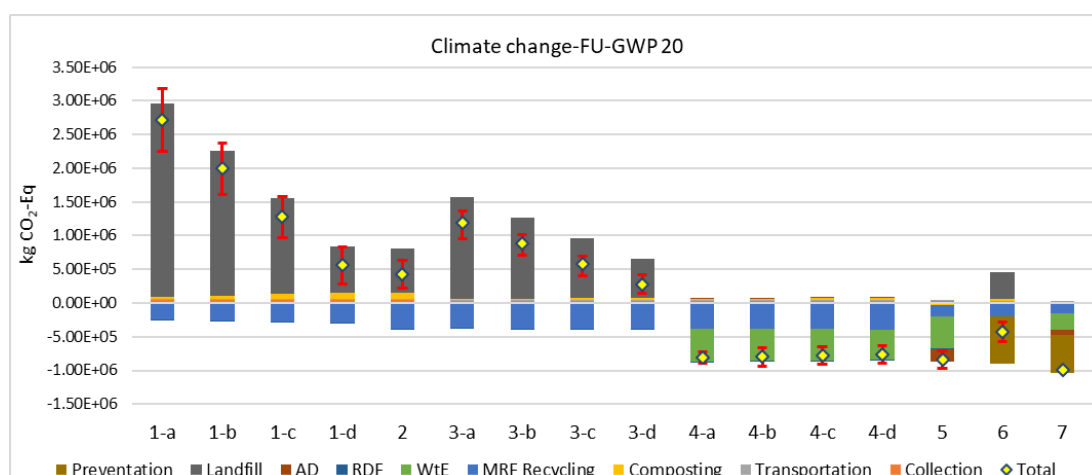


Figure T.2 : Climate change Contribution Analysis (kg CO₂/FU)

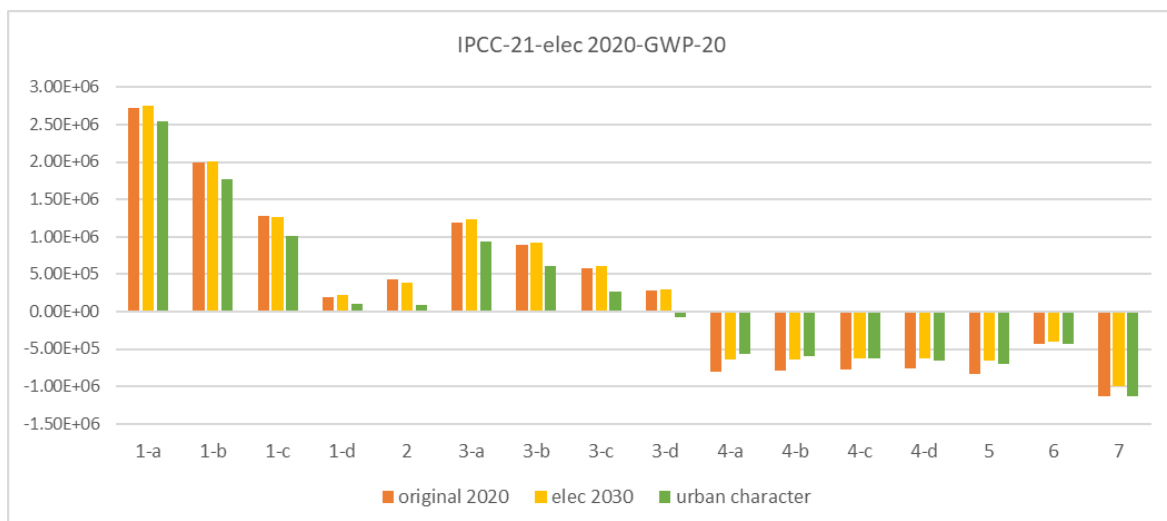


Figure T.3 : Climate change (kg CO₂/FU)

Sensitivity analysis of electricity mix and urban waste characterization scenarios have similar order with GWP 100-year results. However their effects in GWP 20 years calculations almost 3 times of the 100 years results.

APPENDIX U: FOOD WASTE MANAGEMENT IN UNIVERSITY CAMPUS



(a)



(b)

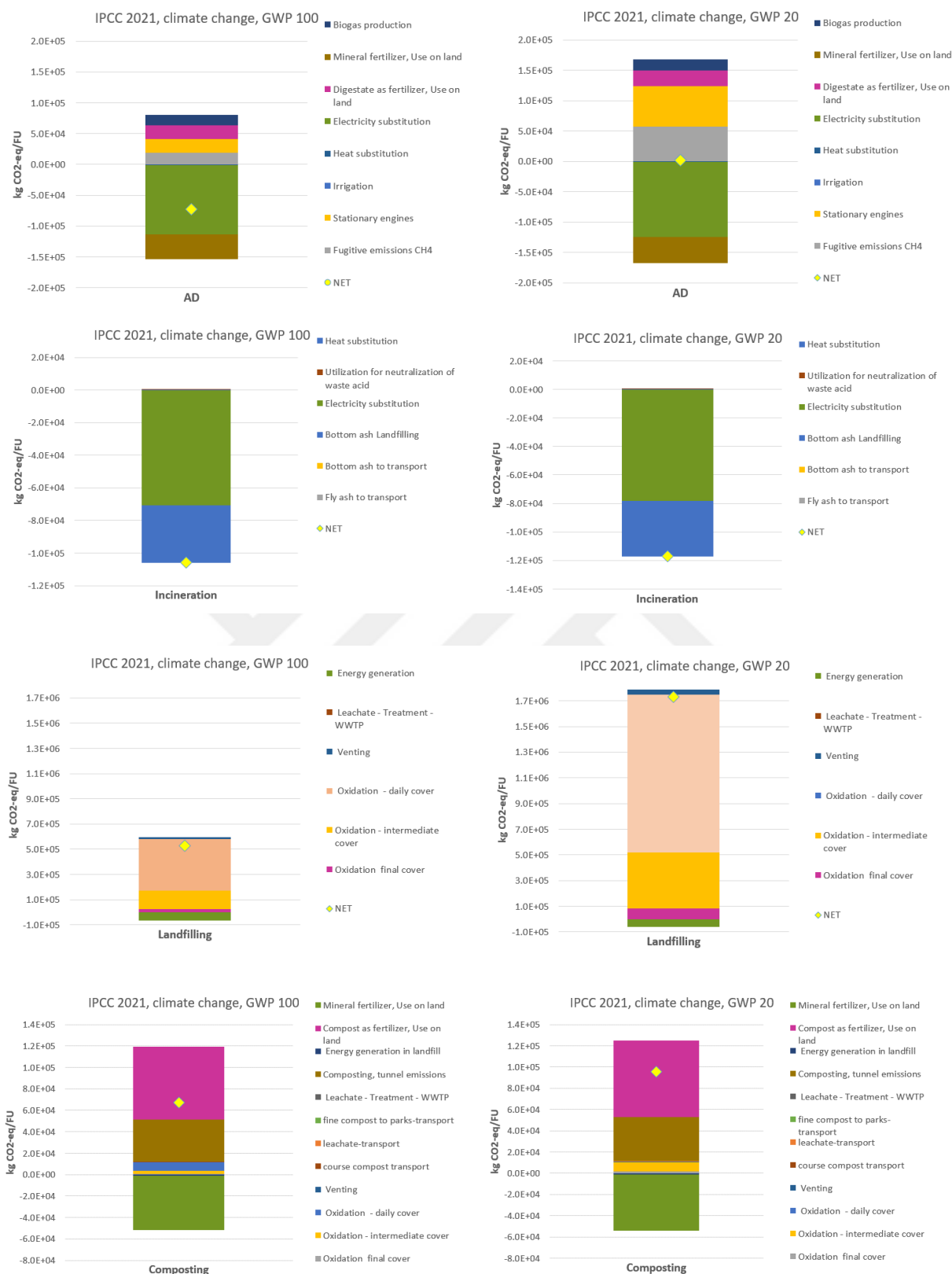


(c)



(d)

Figure U.41 : Photographs of the facilities (a) anaerobic digester. (b) incineration. (c) landfill. (d) composting.



ÜFigure U.2 : Contribution analysis of the carbon emission results. Elec mix 2020 (kg CO₂-eq/FU)

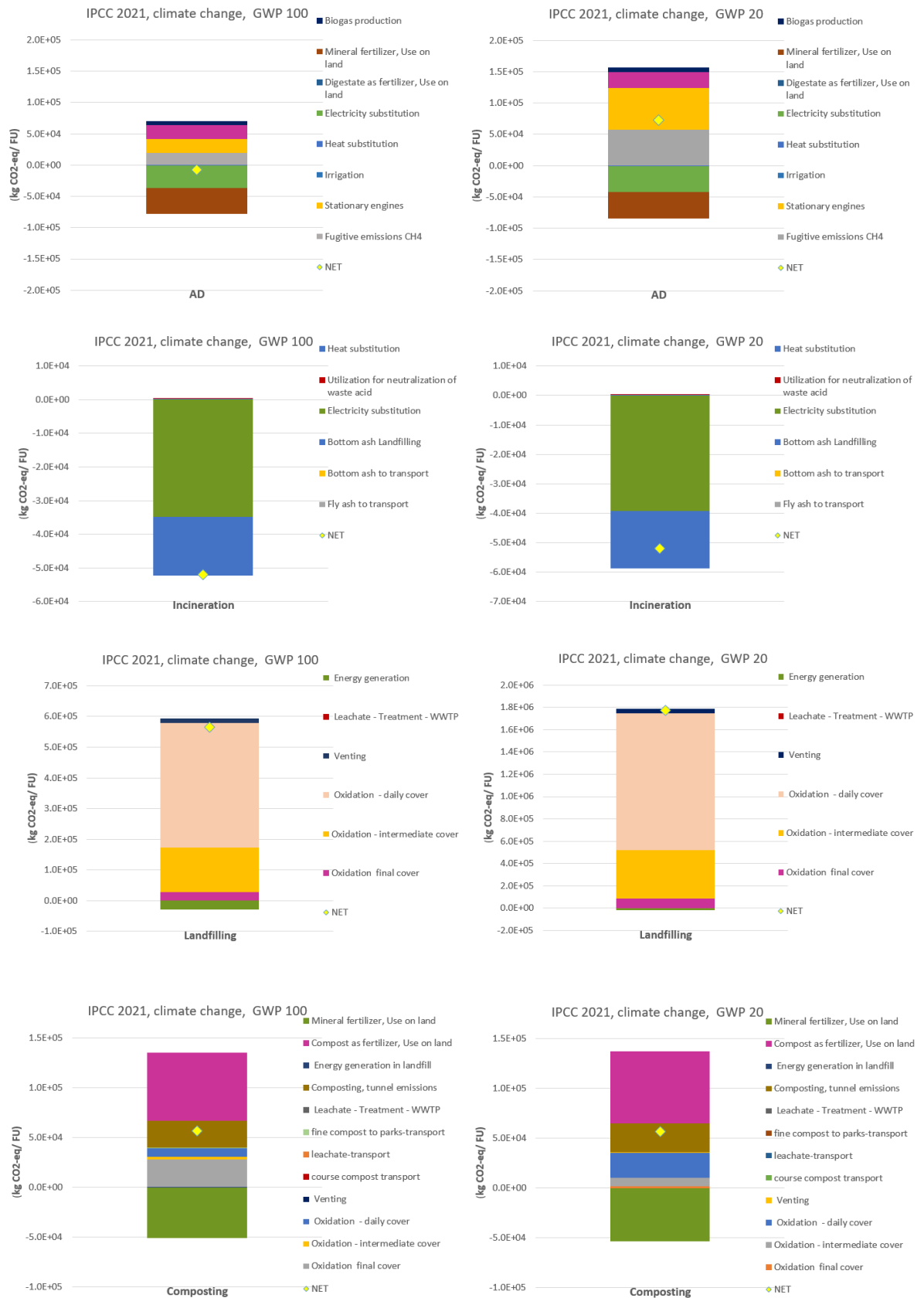


Figure U.3 : Contribution analysis of the carbon emission results. Elec mix 2050 (kg CO₂-eq/ FU)

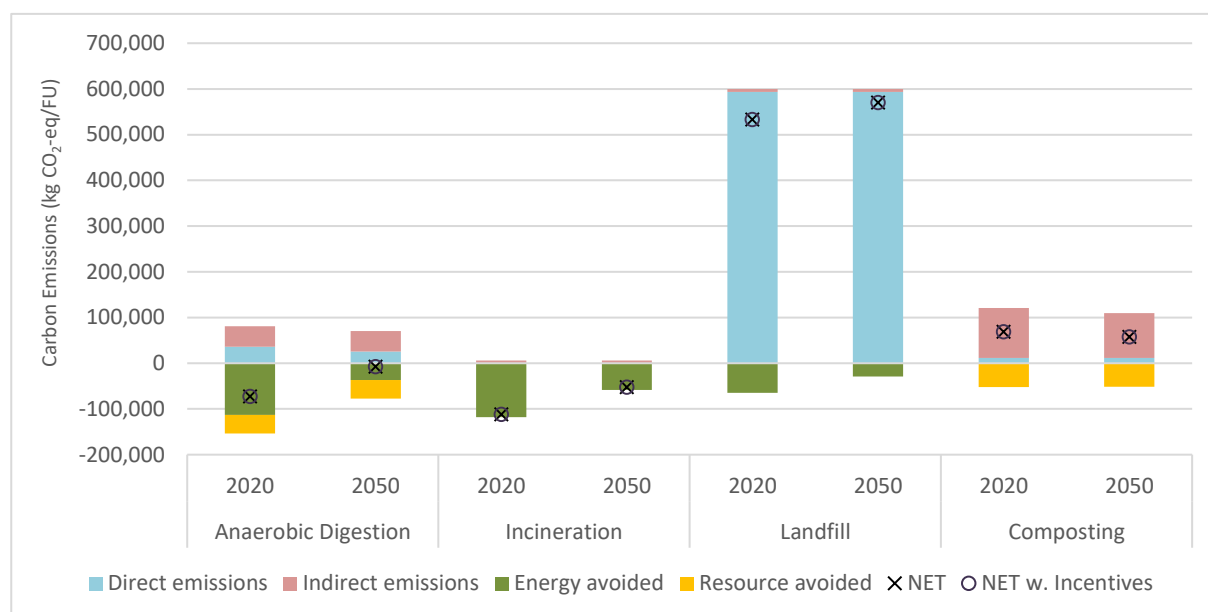


Figure U.4: Net carbon emissions-impact (kg CO₂-eq /FU)

Table U.1 : Sensitivity scenarios for change in the incineration electricity efficiency and biogas upgrading in anaerobic digestion (kg CO₂-eq/FU)

(kg CO ₂ -eq/FU)	Elec Mix 2020	Upgrading Biogas AD 2020	Elec mix 2020 – Inc-Efficiency	Elec Mix 2050	Upgrading Biogas AD Elec mix 2050	Elec Mix 2050 –Inc-Efficiency
Anaerobic Digestion	- 7.25E+01	-7.31E+01	-7.25E+01	-7.42E+00	-1.56E+01	-7.42E+00
Incineration	- 1.12E+02	-1.12E+02	-1.06E+02	-5.22E+01	-5.22E+01	-2.23E+01
Landfill	5.34E+02	5.34E+02	5.34E+02	5.70E+02	5.70E+02	5.70E+02
Composting	6.88E+01	6.88E+01	6.88E+01	5.81E+01	5.81E+01	5.81E+01

AD: anaerobic digester, Inc: incineration

Table U.2 : Net results, CAPEX, OPEX (€/ton FW).

	Anaerobic digestion		Incineration		Landfill		Composting	
	2020	2050	2020	2050	2020	2050	2020	2050
CO ₂ w/o I abatement cost	€ 90	€ 138	€ 100	€ 170	€ 672	1,243	€ 114	€ 157
CO ₂ w/ I abatement cost	€ 90	€ 111	€ 100	€ 87	€ 672	1,134	€ 114	€ 157
CO ₂ w/o I ETS Incentive or ETH Revenues	€ 90	€ 228	€ 100	€ 254	€ 672	2,139	€ 114	€ 258
CAPEX	-€ 24	€ 0	-€ 29	€ 0	-€ 18	€ 0	€ 0	€ 0
OPEX	-€ 8	-€ 15	-€ 36	-€ 56	-€ 5	-€ 9	-€ 6	-€ 13
NET	€ 71	€ 85	€ 105	€ 143	€ 63	€ 80	€ 58	€ 89
	€ 55	€ 68	€ 72	€ 96	€ 40	€ 54	€ 40	€ 63
	€ 94	€ 138	€ 112	€ 183	€ 80	€ 125	€ 92	€ 139

*2050 is showing without incentive scenario

Table U.3 : Net results, CAPEX, OPEX (€/FU)

	Anaerobic digestion		Incineration		Landfill		Composting	
	2020	2050	2020	2050	2020	2050	2020	2050
Incentive or ETH	-€ 13,687	€ 0	-€ 16,984	€ 0	-€ 10,372	€ 0	-€ 269	€ 0
Revenues	-€ 4,620	-€ 8,662	-€ 20,788	-€ 32,529	-€ 2,887	-€ 5,197	-€ 3,465	-€ 32,338
CAPEX	€ 40,999	€ 49,138	€ 60,633	€ 82,576	€ 36,380	€ 46,196	€ 33,492	€ 82,576
OPEX	€ 31,760	€ 39,267	€ 41,577	€ 55,436	€ 23,098	€ 31,183	€ 23,098	€ 55,436
NET	€ 54,453	€ 79,743	€ 64,437	€ 105,484	€ 46,219	€ 72,182	€ 52,857	€ 157,924

*2050 is showing without incentive scenario

Table U.4 : Other Environmental Impact Categories 2020

	Climate change	Human toxicity	Human toxicity non-cancer	Terrestrial acidification	Eutrophication terrestrial	Eutrophication freshwater	Eutrophication marine	Ecotoxicity freshwater
Unit	kg CO₂-eq	CTUh	CTUh	mol H⁺ eq.	mol N eq.	kg P eq.	kg N eq.	CTUe
Anaerobic digestion	-9.7E+04	-1.5E-02	-7.3E-02	4.5E+02	4.6E+03	-4.7E+01	7.6E+02	-2.0E+05
Incineration	-1.2E+05	-1.0E-03	-8.0E-03	-3.4E+02	1.1E+03	-1.0E+01	1.0E+02	-4.1E+04
Landfill	4.3E+05	-1.0E-03	-4.0E-03	-1.4E+02	7.7E+02	-5.7E+00	1.0E+02	-3.6E+03
Composting	9.9E+03	-1.1E-02	3.1E-01	3.1E+02	1.2E+03	-3.5E+01	6.9E+02	8.9E+04

Table U.5 : Other Environmental Impact Categories 2050

	Climate change	Human toxicity	Human toxicity non-cancer	Terrestrial acidification	Eutrophication terrestrial	Eutrophication freshwater	Eutrophication marine	Ecotoxicity freshwater
Unit	kg CO₂-eq	CTUh	CTUh	mol H⁺ eq.	mol N eq.	kg P eq.	kg N eq.	CTUe
Anaerobic digestion	-3.3E+04	-1.5E-02	-6.8E-02	9.7E+02	5.2E+03	-3.7E+01	8.2E+02	-1.9E+05
Incineration	-6.1E+04	-1.0E-03	-4.0E-03	1.7E+02	1.8E+03	-1.1E+00	1.6E+02	-3.0E+04
Landfill	4.7E+05	0.0E+00	-1.0E-03	1.6E+02	1.1E+03	-6.0E-01	1.5E+02	2.9E+03
Composting	-8.1E+02	-1.1E-02	3.0E-01	2.2E+02	1.1E+03	-3.6E+01	6.8E+02	8.7E+04

Table U.6: Normalization factors (Hauschild et. al., 2013; JRC 2011)

Impact category	Normalization factor	Unit
Human toxicity	0.0000542	cases/person
Human toxicity non-cancer	0.0011	cases/person
Terrestrial Acidification	49.6	AE/person
Eutrophication Terrestrial	115	AE/person
Eutrophication Freshwater	0.62	kg P-eq/person
Eutrophication Marine	9.38	kg N-eq/person
Ecotoxicity Freshwater	665	PAF.m ³ .d/person

APPENDIX V: WASTE COMPOSITION (URBAN SCALE)

Table V.1: Waste Biological and physico-chemical properties of waste materials. LHV: lower heating value, VS: volatile solids, BMP: biochemical methane potential, d.w.: dry weight- **in 1000 kg waste** (EASETECH database; Riber et al., 2009)

Fraction name	TS (kg)	Water (kg)	VS (kg)	Ash (kg)	Energy (MJ)	C bio (kg)	C bio and (kg)	C fossil (kg)	Ca (kg)	Cl (kg)	H (kg)	K (kg)	N (kg)	Na (kg)	O (kg)	P (kg)	S (kg)
Beverage cans (aluminium)	4.59	0.42	-	4.59	-	-	-	-	0.00	-	-	0.00	-	0.00	121.51	1.35	0.93
Clear glass	20.24	2.76	0.00	20.24	-	-	-	-	1.37	-	-	0.07	-	0.45	-	0.00	0.00
Food cans (tinplate/steel)	5.21	0.79	0.00	5.21	-	-	-	-	0.00	-	-	0.00	-	0.00	-	0.00	0.02
Plastic bottles	11.64	1.37	10.93	0.71	425.14	0.04	-	8.94	0.04	0.02	1.31	0.00	0.01	0.02	-	0.00	0.00
Green glass	19.31	0.69	0.00	19.31	-	-	-	-	1.33	-	-	0.15	-	0.48	0.61	0.00	0.01
Non-recyclable plastic	80.82	6.18	76.38	4.45	2,583.10	0.29	-	57.06	0.88	3.78	7.84	0.10	0.40	0.09	-	0.00	0.00
Plastic products (toys, hangers, pens)	23.30	1.70	17.43	5.87	642.85	3.46	-	10.38	0.18	0.64	1.61	0.01	0.51	0.01	8.97	0.45	0.04
Batteries	5.47	0.53	0.78	4.69	3.12	0.24	-	0.24	0.00	0.09	0.06	0.22	0.01	0.01	0.83	0.00	0.01
Dirty cardboard	46.06	6.94	39.19	6.86	781.59	19.67	6.54	0.20	1.59	0.06	2.67	0.04	0.14	0.13	0.15	0.00	0.01
Other clean paper	48.15	3.85	39.78	8.37	633.20	18.35	10.55	0.09	2.11	0.03	2.41	0.04	0.10	0.05	16.49	0.02	0.06
Textiles	45.12	2.88	43.50	1.62	893.38	17.64	-	5.87	0.20	0.16	2.71	0.03	1.44	0.16	18.54	0.00	0.09
Juice cartons (carton/plastic/aluminium)	5.87	1.13	5.31	0.56	139.54	2.97	0.51	0.06	0.05	0.01	0.45	0.00	0.01	0.01	15.70	0.10	0.18
Diapers, sanitary towels, tampons	27.80	23.21	25.49	2.31	617.60	13.84	1.08	1.54	0.27	0.04	2.22	0.04	0.25	0.61	1.81	0.00	0.00
Garden waste, Grass and leaves	24.20	30.80	10.89	13.31	217.32	4.14	2.08	0.12	0.21	0.03	0.75	0.30	0.29	0.04	7.59	0.02	0.02
Vegetable food waste	77.63	259.88	73.59	4.04	1,420.54	36.87	32.84	0.19	0.43	0.43	5.12	0.99	1.47	0.24	3.99	0.04	0.03
Other combustibles	76.02	7.98	55.57	20.45	1,857.17	10.34	-	30.94	2.15	0.17	6.16	0.13	0.68	1.10	30.66	0.18	0.14
Other non-combustibles	9.51	5.49	0.22	9.29	-	0.06	-	0.06	0.44	0.00	0.01	0.10	-	0.40	7.32	0.04	0.13
Animal food waste	48.26	64.24	44.06	4.20	1,184.84	26.74	17.37	0.55	1.97	0.79	3.81	0.26	3.38	0.52	0.07	0.00	0.01

Waste Amount Calculation

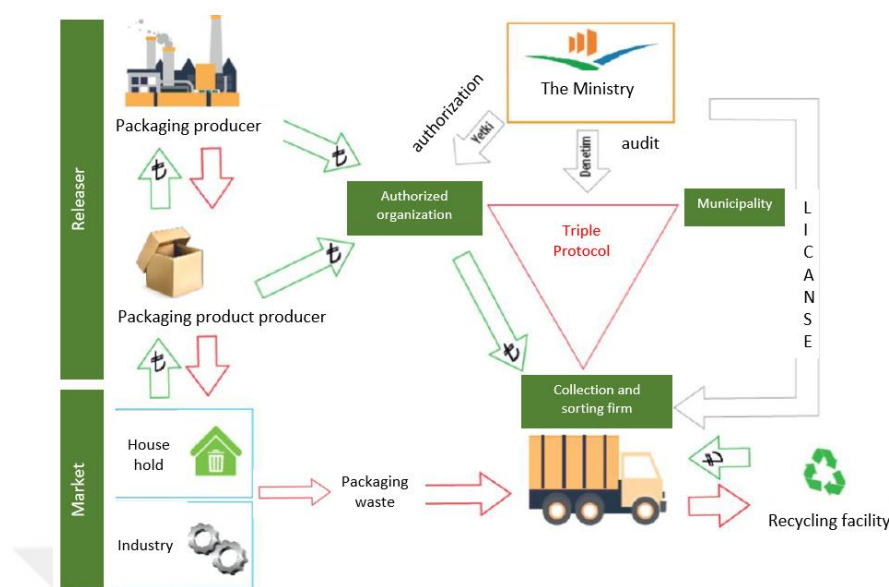


Figure V.1 : The triple protocol for the packaging waste management in Turkey (Şenaydın O., 2018).

The MoEU publish packaging waste bulletin every year. However, this bulletin covers all recycling activities reported to the Ministry and this means it contains both municipal and industrial activities (Şenaydın O., 2018). It is known that CSFs prefer collecting waste from industries since they can collect same amount of waste from less location compared to households. Recycling facilities are generally located close to the industrial regions, hence transportation distance and cost are also reduced (Şenaydın O., 2018). However, actual domestic and industrial amount are unknown but most of them belongs to industry according to previous study (Şenaydın O., 2018). The municipal (household) packaging collection rate is below the targets since it is known that packaging waste reports are not compatible with real life. The source separated packaging waste collection from industries cause to increase in the overall recycling ratio. The separate record of domestic and industrial waste collection processes will help to obtain more accurate data.

Also, illegal (unrecorded) packaging waste collection and recycling are still popular in Turkey. The green arrows show accurate flows according to the Packaging Waste Management Control Regulation while red arrows show illegal or unrecorded waste flows. Another problem for the data collection is waste pickers. The number of waste pickers are based on assumptions. Some studies claim that total waste pickers in Turkey around 200 000 (Fakioğlu E., 2010) other studies claim these number reached to 500 000 in recent years (Türk-İş, 2018). However, exact number is still unknown. The waste pickers in Turkey collecting packaging wastes from streets and sell them to the middleman. The collected waste sometimes sent to unlicensed facilities and sometimes sent to licensed facilities (such as CSFs). However, after a certain point

these illegal waste collection may be recorded with other source separated waste collections. This could lead to misleading or double counting for the real waste collection performance of district municipalities and CSFs. However, this unrecorded packaging waste amount only affects actual secondary material production in the system. The amount of waste sent to landfills is still accurate and does not affect by this problem. The CSFs sometimes serve more than one city and this could affect the data accuracy. Because, the MoEU collects data based on the statement of CFCs and previous study (Şenaydın O., 2018) claim that CFCs could mix data from different cities. This leads to the increment of the uncertainty.

Provincial Environmental Status Report of Istanbul is another source for WM statistics in Istanbul (IPDEU, 2020). The packaging waste recycling data by the authorized organizations is taken from this report. The separately collected recyclable wastes for all 39 districts of Istanbul under the ZWM project are also given in the report. However, these numbers are not compatible neither recycling data of IMM (IMM, 2020) nor total waste production in Istanbul (İSTAÇ, 2020). Hence, recycling data under the ZWM project, given in the Provincial Environmental Status Report, was neglected.

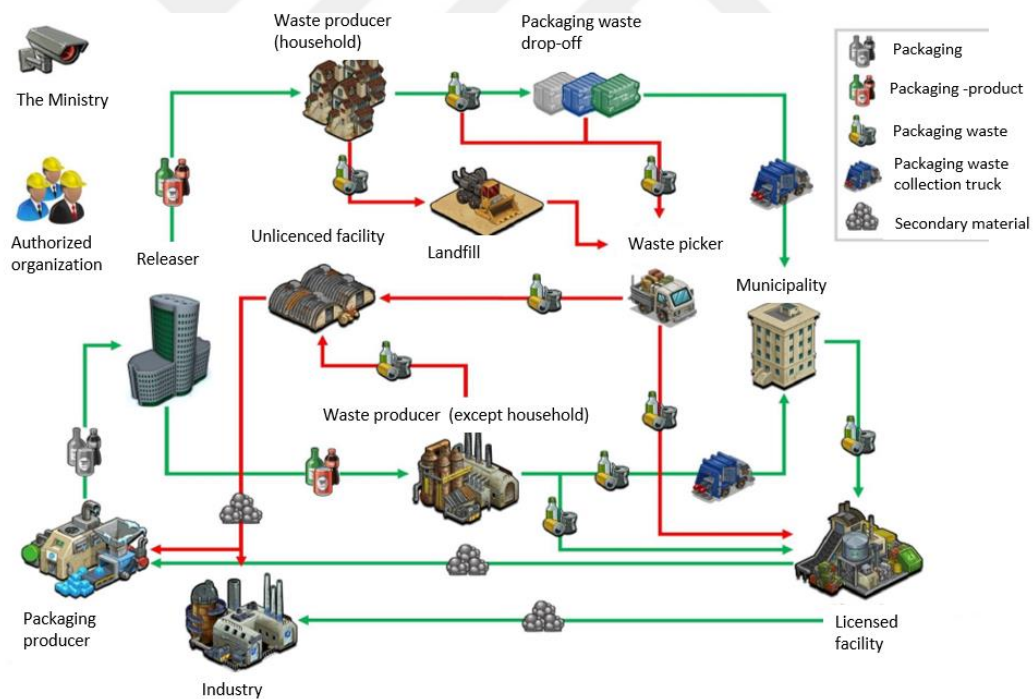


Figure V.2 : The current packaging waste management in Turkey (Ateş M., 2016).

Table V.2 Waste characterization of Istanbul (Ak and Sezer K., 2020).

Waste Composition	Content	%
Paper-Cardboard	All kinds of paper, corrugated cardboard	10.32
Glass	All kinds of color-colorless bottles. jars	4.67
Pet	Water and beverage bottles	1.45
Nylon bags	Any kind of grocery bags	9.67
Plastics	Any kind of plastics waste except PET	2.60
Textile	All kinds of textile materials	4.89
Composite	Fruit juice boxes. milk boxes	0.88
Diapers	All kinds of diapers	5.17
Metals	All kinds of metal materials	1.01
Electronics	Computer. telephone. radio. etc.	0.17
Hazardous waste	Battery. detergent boxes. medicine boxes. paint boxes	0.94
Biodegradable waste	Food waste. vegetables. fruit. park-garden wastes	47.93
Other Combustible	Shoe. carpet. bag. foam. food packaging. board	8.31
Noncombustible	Ash. stone. rubble. ceramic. dust	1.99

The total separately collected waste by municipalities were given in (IMM. 2020) however, there is no information given about purity and characterization of separately collected waste. Hence, waste characterization assumption was done by using previous study conducted for Uskudar district (Ukşal. F., 2019). However, the original data from (Ukşal. F., 2019) was changed since source separated waste have some purity ratio and additional waste components other than recyclables such as organic and other wastes.

Table V.3: The total separately collected waste by municipalities of Istanbul for 2020

Waste components	%	tonne/year
Paper/cardboard	60	281,483
Glass	2	9,383
PET bottles	3	14,074
Plastic bags	5	23,457
Plastics	8	37,531
Metals (Iron)	2	9,383
Aluminum	3	14,074
Other metals	2	9,383
Organic	5	23,457
Other	10	46,914
TOTAL	100	469,139

The waste collection amount by authorized institutions was taken from Provincial Environmental Status Report of Istanbul (IPDEU, 2020). It is believed that these wastes are mostly coming from the industrial area (Şenaydın O., 2018). The total waste coming from households was assumed as 30% for this study.

Table V.4 The total separately collected waste by authorized institutions for 2020

Waste component	Waste coming from households
Plastic	2,768
Metal	50
Composite	4
Paper/cardboard	47,178
Glass	9,893
Mix	68,212
Total	128,104

The waste pickers are important part of packaging waste collection from households. However, their exact number is still unknown. Total number of waste pickers and waste components were assumed by using previous studies (Tutar M., 2020; Güğüş E.A., 2019; Uğşal. F., 2019; Deniz G.C., 2010; Fakioğlu E., 2010). The daily waste collection per a waste picker was assumed as 90 kg. It is clear that waste picker calculations have higher uncertainty since both waste amount and waste characterization data were assumed.

Table V.5: The Number of waste pickers in Istanbul

Number of waste pickers in Istanbul (literature based assumption)	
Asia	6,000
Europe	12,000

Table V.6: The waste collection by waste pickers (scavengers) for 2020

Waste component	Europe (tonne/year)	Asia (tonne/year)	Total (tonne/year)
Paper/cardboard	256,230	128,115	384,345
Glass	11,826	5,913	17,739
PET bottles	39,420	19,710	59,130
Plastic bags	19,710	9,855	29,565
Plastics	43,362	21,681	65,043
Metals (Iron)	7,884	3,942	11,826
Aluminum	7,884	3,942	11,826
Other metals	7,884	3,942	11,826
Total	394,200	197,100	591,300

The street cleaning waste amount was taken from annual report of İSTAÇ (İSTAÇ, 2020) and total waste amount taken from (İMM, 2020). The waste collection amount during coast and beach cleaning was taken from annual report of İSTAÇ (İSTAÇ, 2020) and waste characterization taken from (Bilgili et al., 2019).

Table V.7: The waste collection from coast and beach cleaning for 2020

Waste Component	Beach (tonne/year)	Surface Cleaning (tonne/year)	Coastline (tonne/year)
Biodegradable waste	1,139	999	691
Plastic bags	216	155	465
Paper–cardboard	135	74	361
Composite	13	9	27
Plastics	57	89	118
Bottles	125	106	290
Glass	195	85	745
Metals	49	28	88
Electrical–electronic waste	3	4	-
Hazardous waste	5	8	47
Diaper	34	-	7
Textile	117	17	54
Other combustible	304	315	475
Other non-combustible	208	4	3
Total	2,600	1,891	3,372

Population and Waste Forecasting

Population Forecasting for household and commercial waste amount

Turkish census system has changed almost 15 years ago as a “**address-based population registration system**”. The population data after 2005 were used in calculations.

Estimations of future populations were done according to populations for winter and summer conditions. During calculations different methods were used such as;

- Arithmetic Increasing Method
- Geometric Increasing Method
- Province Bank Method
- United Nations Development Programme (UNDP) Method
- CHM2mill

Arithmetic Increase Method

It is based on the assumption that the rate of growth of population (k_a) is constant. It means that the each year population increase by the same increment.

$$Ng = Ni + \left[\frac{(Ns - Ni)}{ts - ti} \right] * (tg - ti)$$

$$k = \frac{Ns - Ni}{ts - ti}$$

Ng : projected population
 Ni : initial population data
 Ns : final population data
 ti : initial year
 ts : final year
 tg : projected year
 ka : average population growth

Geometrical Increasing Method

It is based on the hypothesis that rate of change of population is proportional to the population. According to this, method it is assumed that the rate of increase of population growth in a community is proportional to the present population.

$$kg = \frac{\ln\left(\frac{Ns}{Ni}\right)}{ts - ti} \ln(Ng) = \ln(Ns) + kg * (tg - ts)$$

Ng : projected population
 Ni : initial population data
 Ns : final population data
 ti : initial year
 ts : final year
 tg : projected year
 kg : population growth

Bank of Provinces Method

This method is known as the modified version of geometrical projection method. Also, this method is the most common method used for population projections.

$$\text{Growth rate calculation: } p = \left(\sqrt[a]{\frac{N_s}{N_i}} - 1 \right) * 100$$

According to Bank of Provinces,

$p < 1$ $p=1$

$1 \leq p \leq 3$ calculated p value is taken as it is

$p > 3$, $p=3$

$$\text{Future population calculation } N_s = N_i * \left(1 + \frac{p}{100} \right)^{(ts-ti)}$$

N_i: initial population data

N_s: final population data

t_i: initial year

t_s: final year

p: growth rate

a: time between two populations

United Nations Development Programme (UNDP) Method

A population estimation is already done by the UN and 5 year-period constants are determined. According to those constants, population estimation is done within the scope of this project. Annual average rate of change (AAROC%) table is below. This rates are changing due to area's characteristics, according to if it is rural or urban.

$$Y_s = Y_i + (Y_i \times \text{AAROC} / 100)$$

Y_i: initial population data

Y_s: final population data

AAROC: Annual average rate of change

CHM2mill

Within the scope of the Istanbul Solid Waste Management Feasibility Study, CH2M-Hill/Antel Consortium calculated the populations of the Asian and European sides of Istanbul separately for three different growth rates, and the high growth rate scenario of solid waste amounts were selected. The results obtained by extending it until 2050 (ISTAC,2005).

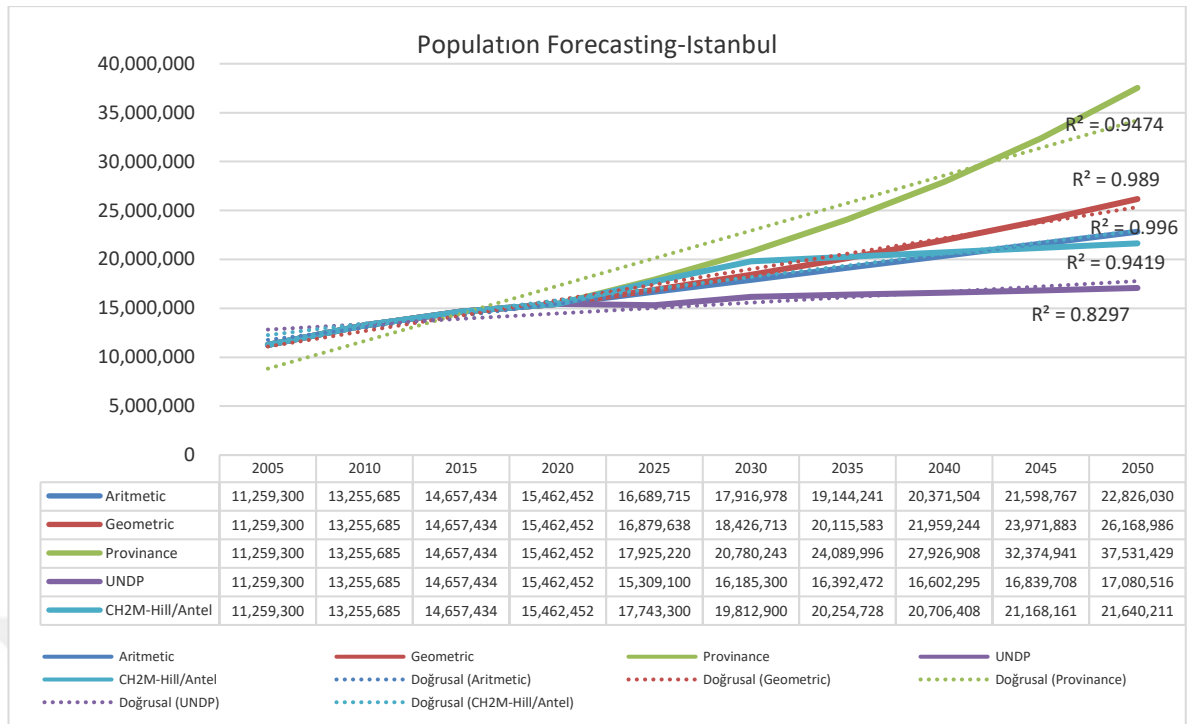


Figure V.3 : Population forecasting of Istanbul

Waste Amount Forecasting

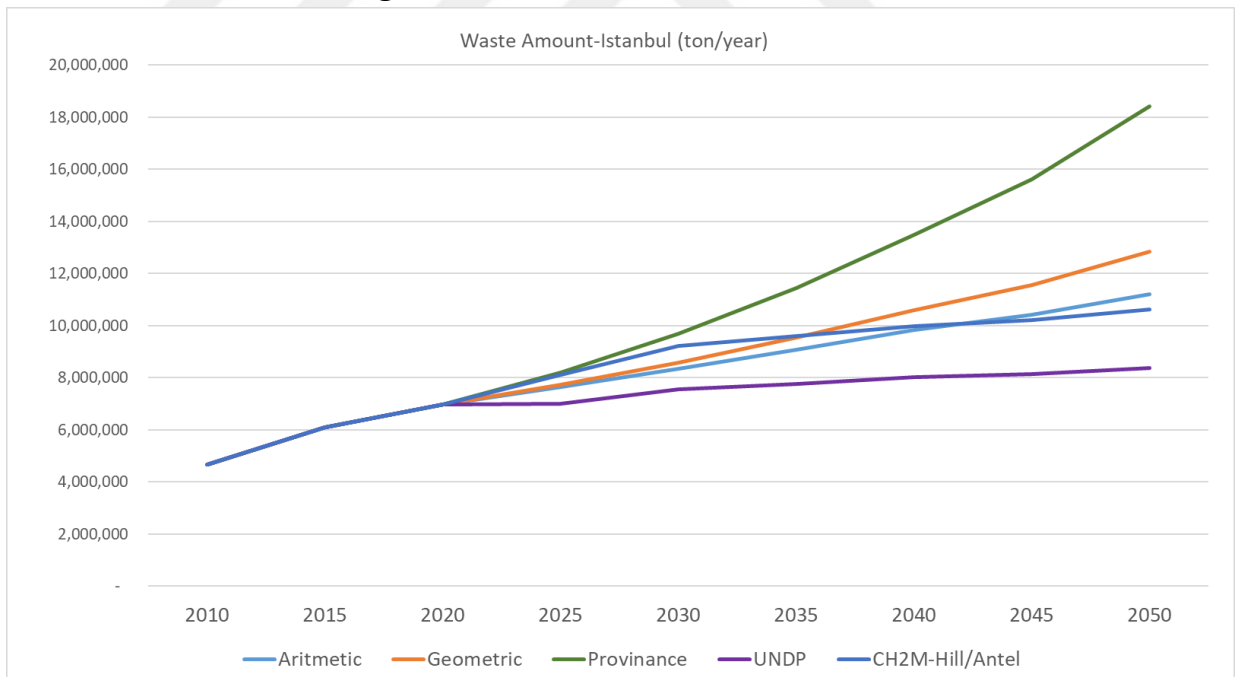


Figure V.4: Waste forecasting of Istanbul

There could be different secondary data sources especially for the urban scale since lots of actor are involved. Istanbul Metropolitan Municipality (IMM) and the Istanbul Environmental Protection and Waste-Processing Corporation (ISTAC) are leading actors for the WM in Istanbul (Kanat G., 2010). ISTAC was established by IMM in

1994 and ISTAC is implementing integrated WM practices in Istanbul (Kanat G., 2010). Waste collection (and transfer to transfer stations) is responsibility of the district municipalities (GNAT, 2005). The district municipalities report their waste collection data to IMM hence the accuracy of the collection data is linked to data monitoring quality of districts. The waste transportation, composting, recycling and disposal are responsibility of IMM and they are practiced by ISTAC. Therefore there are two different sources existing, for the composting and waste disposal to landfills, which are IMM's open data portal (IMM, 2020) and annual reports of ISTAC (ISTAÇ, 2020).

Waste Characterization Forecasting

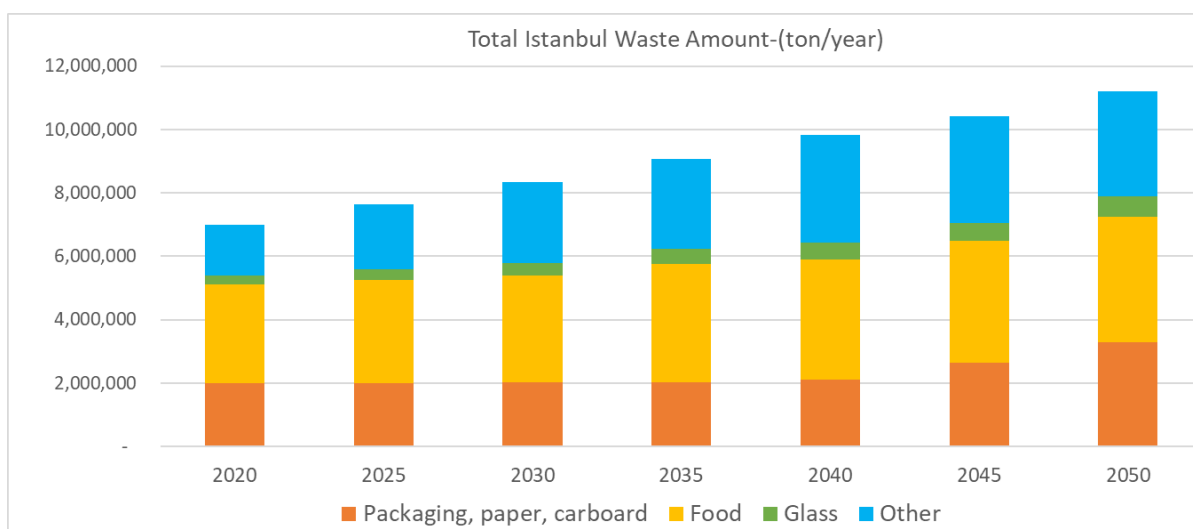


Figure U.12 : Waste characterization forecasting of Istanbul

The recycling activities have additional sources other than IMM and ISTAC since more actors are involved. The district municipalities can collect packaging waste by themselves or they can sign a triple protocol. The Ministry of Environment and Urbanisation (MoEU), the authorized organizations (ÇEVKO, TÜKÇEV, PAGÇEV, AGED), municipalities and collection and sorting firms (CSFs) are other related actors for the packaging wastes. The triple protocol is signed by these last three actors and it is audited by the MoEU.

APPENDIX W: WASTE AMOUNT FORECASTING ASIAN AND EUROPEAN SIDE OF ISTANBUL (MFA RESULTS)

Table W.1: Istanbul future waste management scenarios-I

Capacities	Total waste amount (tonne/year)	Food waste amount (tonne/year)	Source separated food amount (tonne/year)	AD capacity (tonne/day)	Source separated food (%)	Actual Food waste recycling (%)
<u>[BAU] 2020</u>	7,404,375	3,122,754	-	-	-	2.7
<u>[BAU]2035</u>	9,232,680	3,759,957	-	-	-	2.2
<u>[BAU]2050</u>	11,204,919	3,959,859	-	-	-	2.1
<u>[Prosp] 2035</u>	9,232,680	3,759,957	1,879,979	5,151	50	42.0
<u>[Prosp +inc] 2035</u>	9,232,680	3,759,957	1,879,979	5,151	50	42.0
<u>[Prosp +inc-AD]2035</u>	9,232,680	3,759,957	939,989	2,575	25	22.0
<u>[Prosp] 2050</u>	11,204,919	3,959,859	3,959,859	10,849	100	80.6
<u>[Prosp inc] 2050</u>	11,204,919	3,959,859	3,959,859	10,849	100	80.6
<u>[Prosp +inc-AD] 2050</u>	11,204,919	3,959,859	1,979,930	5,424	50	41.7

Table W.1: cont. Istanbul future waste management scenarios-II

Capacities	Packaging waste amount (tonne/year)	Source separated Packaging amount (tonne/year)	MRF capacity (tonne/day)	Source separated Packaging (%)	Actual Packaging recycling (%)	Incineratio n capacity (tonne/day)	Actual landfill diversion (%)
<u>[BAU] 2020</u>	2,283,951	1,206,492	3,646	55.4	35.3	-	15.5
<u>[BAU]2035</u>	2,772,950	1,221,812	3,646	44.1	26.7	-	11.4
<u>[BAU]2050</u>	3,920,505	1,238,666	3,646	31.6	20.0	-	11.1
<u>[Prosp] 2035</u>	2,772,950	1,663,770	4,558	60.0	34.0	4,250	44.0
<u>[Prosp +inc] 2035</u>	2,772,950	1,663,770	4,558	60.0	34.0	7,250	56.0
<u>[Prosp +inc-AD]2035</u>	2,772,950	1,663,770	4,558	60.0	34.0	7,250	48.0
<u>[Prosp] 2050</u>	3,920,505	3,920,505	10,741	100.0	74.0	7,250	78.0
<u>[Prosp inc] 2050</u>	3,920,505	3,920,505	10,741	100.0	74.0	12,550	95.2
<u>[Prosp +inc-AD] 2050</u>	3,920,505	3,920,505	10,741	100.0	74.0	12,550	81.5

MFA studies were conducted by using STAN software. MFA figure of selected scenarios (Europe BAU 2020, and Asia BAU 2020) were shown in this section.

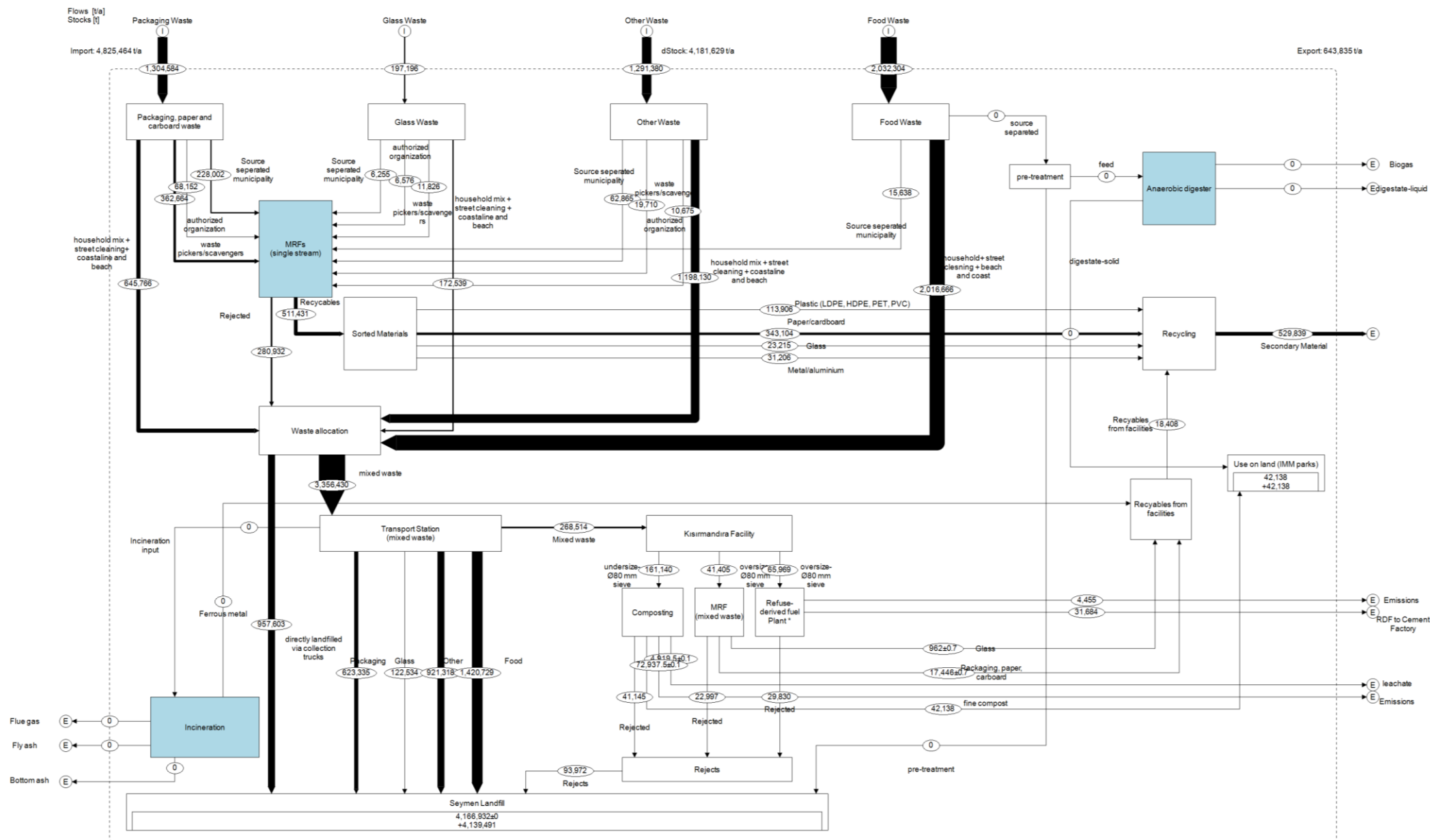


Figure W.1:MFA of European Side-BAU 2020

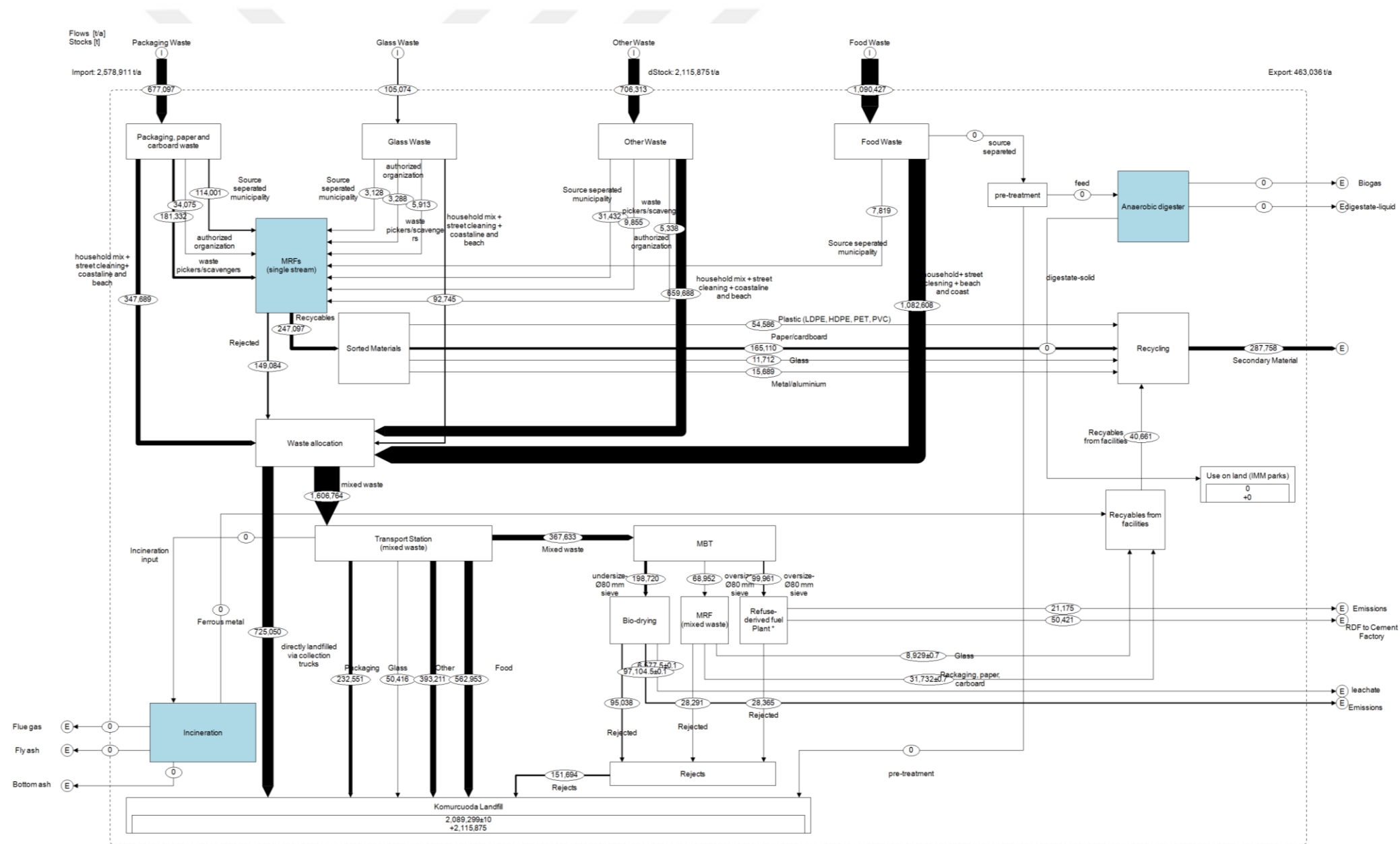


Figure W.2:MFA of Asian Side-BAU 2020



APPENDIX X: WASTE MANAGEMENT FACILITIES IN ISTANBUL

Collection and Transport

Exhaust emissions from the trucks are compatible with EURO-5 standard. Diesel consumption of collection trucks 5.69 L/tonne waste in Sarıyer (Yaman, 2011) and transportation trucks 0.018 L/tonne-km (Yaman, 2011; Sezer, 2019) in Istanbul. Diesel (0.42 L/tonne) and electricity (4 kWh/tonne) consumption in the Transport Station (TS) were taken from literature (Turner et al., 2016).

Table X.1: Campus- Transport Distance-Europe

Facilities	Distance (km)
Kısırmandıra Facility - Seymen Landfill	86
MRF-Seymen Landfill	110
MRF (single stream) packaging-Recycling Center	52.5
MRF glass-Recycling Center	27.5
MRF mixed waste -Recycling Center	22.5
TS - Seymen Landfill	112
TS – Kısırmandıra Facility	26
TS - Komurcuoda Landfill	50
TS –MBT Facility	26
TS-Incineration Facility	26.5
RDF to cement factory	50
Fine compost to use on land	20
Compost leachate to landfill	8
Bottom ash to landfill	22.5

Energy Sources

The missing data in the background systems were obtained from the Ecoinvent 3.7.1 (Ecoinvent, 2020) and EASTECH database. Electricity grid of Istanbul was assumed as same with Turkey electricity mix. The marginal electric mix data was taken from literature (Ecoinvent, 2020; EPDK, 2020). However, electricity mix calculated by using fossil and renewable electricity source values, instead of directly taken Turkish mix value in Ecoinvent, since the data based on 2017 and electricity mixture in Turkey change thanks to new hydro-electricity facilities (EPDK, 2020).

Table X.2: Licensed Electricity Generation and Consumption in Turkey (EPDK.2020)

Resource Type	2019 (GWh)	Share (%)
Hydro Gam	88.850,17	30,20
Import Hard Coal	60.381,27	20,52
Natural Gas	56.522,71	19,21
Lignite	46.893,73	15,94
Wind	21.636,28	7,35
Geothermal	8.929,73	3,03
Biomass	4.266,32	1,45
Hard Coal	3.518,87	1,20
Asphaltite Coal	2.323,95	0,79
Fuel Oil	732,92	0,25
Solar	194,37	0,07
Diesel	1,00	0,00
Total	294.251,32	100,00

Table X.3: Turkey Electricity mix GWP 100 for 2020.

Climate change w/o LT; midpoint; GWP100;	
Unit	kg CO ₂ -Eq
Sum	0.5532
electricity, high voltage,heat and power co-generation, lignite,TR	0.1971
electricity, high voltage,heat and power co-generation, hard coal,TR	0.2651
electricity, high voltage,electricity production, wind, >3MW turbine, onshore,TR	0.0014
electricity, high voltage,electricity production, natural gas, combined cycle power plant,TR	0.0811
electricity, high voltage,electricity production, hydro, run-of-river,TR	0.0012
electricity, high voltage,electricity production, oil,TR	0.0023
electricity, low voltage,electricity production, photovoltaic, 3kWp slanted-roof installation, multi-Si, panel, mounted,TR	0.0000
electricity, high voltage,electricity production, deep geothermal,TR	0.0020
electricity, high voltage,heat and power co-generation, biogas, gas engine,TR	0.0030

Electricity mix in 2050 for sensitivity analysis was calculated by using Keramidas et al, 2020 (Geco-2020).

Table X.4: Turkey Electricity mix forecasting- 2050 Keramidas et al, 2020 (Geco-2020)

Resource Type	GECO normal scenario 2050 (%)
Coal	7.8
Gas	15.4
Nuclear	1.1
Hydro	15.6
Wind	36.2
Solar	19.0
Other	5.3

Table X.5 : Turkey Electricity mix GWP 100 for 2050.

Climate change w/o LT; midpoint; GWP100 Unit	kg CO2- Eq
Sum	0.0496
electricity, high voltage,electricity production, natural gas, combined cycle power plant,TR	0.0019
electricity, low voltage,electricity production, photovoltaic, 3kWp slanted-roof installation, multi-Si, panel, mounted,TR	0.0251
electricity, high voltage,electricity production, wind, >3MW turbine, onshore,TR	0.0130
electricity, high voltage,electricity production, deep geothermal,TR	0.0031
electricity, high voltage,electricity production, hard coal,TR	0.0062
electricity, high voltage,electricity production, nuclear, pressure water reactor,FR	0.0002

Table X.6 : Data Quality Assesment of the study

	Reliability	Completness	Temporal Correlation	Geographical Correlation	Technological Correlation	DQR	Source
Waste characterization and composition							
Municipal waste	2	1	1	1	1	1.6	Ak and Sezer, 2020
packaging MSW	2	1	1	1	1	1.6	IMM, 2020
authorized institutions	2	1	1	2	1	1.7	CEVKO, 2019
beach and cost	1	1	1	1	1	1.0	Bilgili et. al.,2019
street cleaning	2	1	1	1	1	1.6	ISTAC, 2019
scavengers	3	4	2	3	3	3.4	Tutar M., 2020; Güğüş E.A., 2019; Uğşal. F., 2019; Deniz G.C., 2010; Fakioglu E., 2010
Chemical Composition	1	1	4	4	1	3.0	Riber et al., 2009
Waste Amount							
Municipal waste	2	1	1	1	1	1.6	Ak and Sezer, 2020
packaging MSW	2	1	1	1	1	1.6	IMM, 2020
authorized institutions	2	1	1	2	1	1.7	CEVKO, 2019
beach and cost	1	1	1	1	1	1.0	Bilgili et. al.,2019
street cleaning	2	1	1	1	1	1.6	ISTAC, 2019
scavengers	3	4	2	3	3	3.4	Tutar M., 2020; Güğüş E.A., 2019; Uğşal. F., 2019; Deniz G.C., 2010; Fakioglu E., 2010
Collection and Transportation							
Fuel consumption	2	1	1	1	1	1.6	IMM, Sezer, 2019
Distances	1	1	1	1	1	1.0	IMM, Sezer, 2019
Waste Management and Facilities							
Composting-TC	1	1	4	1	1	2.7	Sezer and Arıkan, 2010
Composting-Electricity consumption	1	1	3	3	1	2.3	Boldrin et al., 2011
Composting-Use on land	2	2	1	4	2	3.0	Cobo et al.,2018
RDF-Diesel consumption	1	1	2	3	2	2.3	Grzesik and Malinowski, 2016
RDF-Electricity consumption	2	2	3	3	2	2.7	Burnley et al.,2011
MRF-TC	1	2	1	4	2	2.9	Pressley et al., 2015
MRF-Electricity consumption	1	2	3	2	1	2.3	Kurdoğlu, 2013; Yıldız-Geyhan, 2016
MRF-market substitution ratios	2	2	1	4	2	3.0	Fernández-Braña et al., 2020
AD-TC	2	1	1	1	1	1.6	IMM;2020 Akça et al., 2020
AD-Electricity consumption	2	3	2	3	3	2.8	Boldrin et al., 2011
AD-Use on land	2	2	1	4	2	3.0	Cobo et al.,2018
Incineration-TC	2	1	1	3	3	2.4	Bisinella et al., 2021
Incineration-Emissions to air	2	1	1	3	3	2.4	Bisinella et al., 2021
Incineration-Ancillary material consumption	2	1	1	3	3	2.4	IMM 2012
Incineration-electricity efficiency	2	1	1	3	3	2.4	IMM 2012
Incineration-heat efficiency	2	1	1	3	3	2.4	IMM,2012
Landfill -gas collection	2	1	3	3	4	3.2	EASETECH
Leachate Treatment	2	1	3	3	4	3.2	Turkish water institute, 2019
Leachate Amount	3	1	1	3	4	3.1	Schroeder, et al., 1994; Yang et al., 2015
Background data							
Electricity mix-TR	2	1	1	1	3	2.2	ecoinvent v3.7.1.
Heat mix-TR	2	1	1	1	3	2.2	ecoinvent v3.7.1
Electricity -forecasting-TR	2	1	1	1	3	2.2	Keramidas et al, 2020 (Geco-2020)

Perturbation analyses and uncertainty analysis

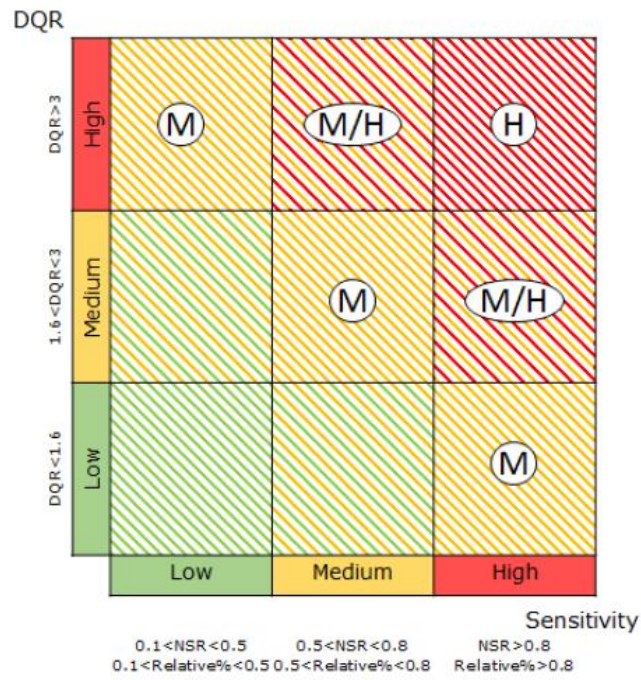


Figure A.59 : Critical parameters classification by using NSR and DQA figure by (Andreasi Bassi et al., 2017)

Normalized sensitivity ratios were calculated as:

$$\text{Normalized Sensitivity Ratio (NSR)} = \frac{\text{abs}(SR)}{\max(\text{abs}(SR))}$$

Where;

$$\text{Sensitivity Ratios (SR)} = \frac{\frac{\Delta \text{ Results}}{\text{Initial Result}}}{\frac{\Delta \text{ Parameter}}{\text{Initial Parameter}}}$$

Perturbation analyses was carried out. A perturbation analysis was performed by increasing by 10% the parameters.

Table X.7 : NSR and DQA

	NSR	DQA	Final color
Marginal Electricity Process	low	medium	
Marginal Electricity Process	low	medium	
Marginal Electricity Process	low	medium	
Marginal Electricity Process	low	medium	
Marginal Electricity Process	low	medium	
Substitution of Glass Manufacturing without cullets, 84% cullet ratio in recycling	low	medium	
Substitution steel, market ratio 85%	low	medium	
Substitution PET, market ratio 81%	medium	medium	
Substitution Aluminium Market ratio 100%	medium	medium	
Substitution virgin newspaper and CTMP, Market response 100%	high	medium	
Substitution of virgin cardboard, market response 85%	medium	medium	
Polyethylene high density granulate (PE-HD), 1999, RER, ELCD	medium	medium	
As	low	medium	
Cd	low	medium	
Cl	low	medium	
Cr	low	medium	
Cu	low	medium	
Hg	low	medium	
Mo	low	medium	
Ni	low	medium	
Pb	low	medium	
S	low	medium	
Sb	low	medium	
Zn	low	medium	
Electricity recovery	high	medium	
Heat recovery	low	medium	
Electricity consumption	low	medium	
Sodium hydroxide (NaOH)	low	medium	
Activated carbon	low	medium	
Limestone (CaCO ₃)	low	medium	
Ammonia (NH ₃)	low	medium	
Hydrogen chloride (HCl), gas	low	medium	
Ammonia	low	medium	
Antimony	low	medium	
Arsenic, ion	low	medium	
Barium	low	medium	
Cadmium, ion	low	medium	
Calcium, ion	low	medium	
Carbon monoxide, fossil	low	medium	
Chloride	low	medium	
Chromium, ion	low	medium	
Cobalt	low	medium	
Copper, ion	low	medium	
Dioxins, measured as 2,3,7,8-tetrachlorodibenzo-p-dioxin	low	medium	

Table X.7 (cont.) : NSR and DQA

Fluoride	low	medium	
Hydrogen chloride	low	medium	
Hydrogen fluoride	low	medium	
Iron, ion	low	medium	
Lead	low	medium	
Magnesium	low	medium	
Manganese	low	medium	
Manganese	low	medium	
Mercury	low	medium	
Molybdenum	low	medium	
Nickel	low	medium	
Nickel, ion	low	medium	
Nitrogen oxides	low	medium	
PAH, polycyclic aromatic hydrocarbons	low	medium	
Particulates, > 10 um	low	medium	
Selenium	low	medium	
Silicon	low	medium	
Sulfur dioxide	low	medium	
Zinc, ion	low	medium	
Substitution Aluminium, Market response 100%	medium	medium	
Limestone substitution (CaCO3)	low	medium	
leaking of gas	medium	high	
Oxidation in top cover, daily (-10%)	low	high	
Oxidation in top cover, intermediate (-10%)	low	high	
Oxidation in top cover, final (-10%)	low	high	
leaking of gas	high	high	
Oxidation in top cover, daily (-10%)	low	high	
Oxidation in top cover, intermediate (-10%)	low	high	
Oxidation in top cover, final (-10%)	low	high	
leaking of gas	medium	high	
Oxidation in top cover, daily (-10%)	low	high	
Oxidation in top cover, intermediate (-10%)	low	high	
Oxidation in top cover, final (-10%)	low	high	
Electrical efficiency of the gas collected (+10%)	medium	high	
Heat efficiency of the gas collected (+10%)	medium	high	
biochemical methane potential (yield) (%) garden	low	medium	
biochemical methane potential (yield) (%) vegetable	low	medium	
Leaking CH4 (+10%)	low	medium	
N-plant uptake	low	medium	
P-plant uptake	low	medium	
N-plant uptake	low	medium	
P-plant uptake	low	medium	
N-plant uptake	low	medium	
P-plant uptake	low	medium	



APPENDIX Y: CONTRIBUTION ANALYSIS OF CLIMATE CHANGE IMPACT CATEGORY

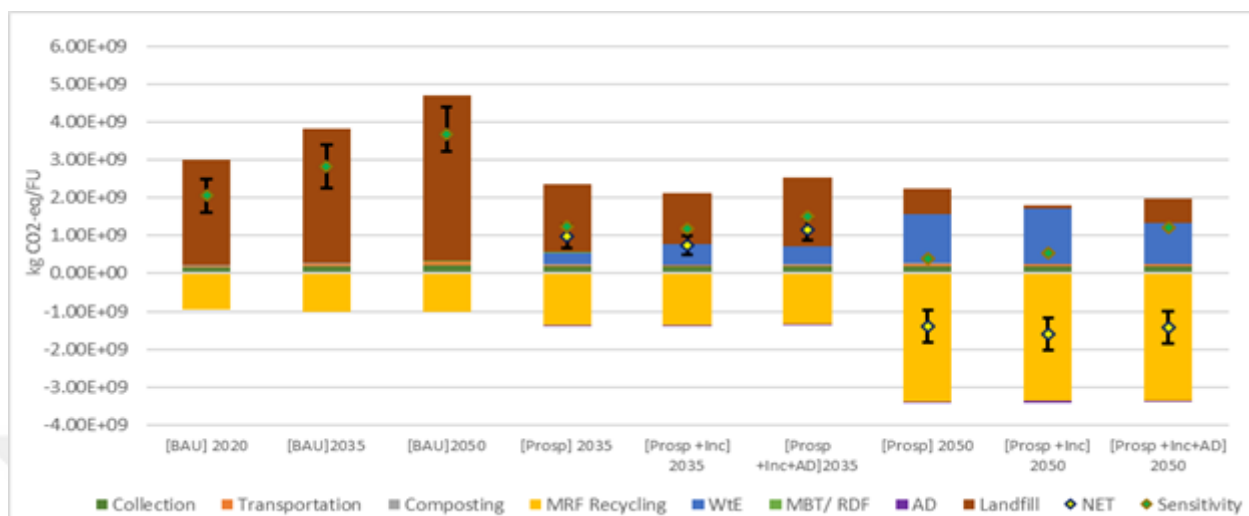


Figure Y.1 : Uncertainty result of Istanbul

Recycling Sensitivity Scenario

Impurities are increasing with higher source separation (Haupt et. al., 2018b).

For year 2035 municipality recycling target is 60%, real recycling according to (Haupt et. al., 2018b);

Real recycling = 60% x 0.94 = **56%**

For year 2050 municipality recycling target is 100%, real recycling according to (Haupt et. al., 2018b);

Real recycling = 100% x 0.58 = **58%**

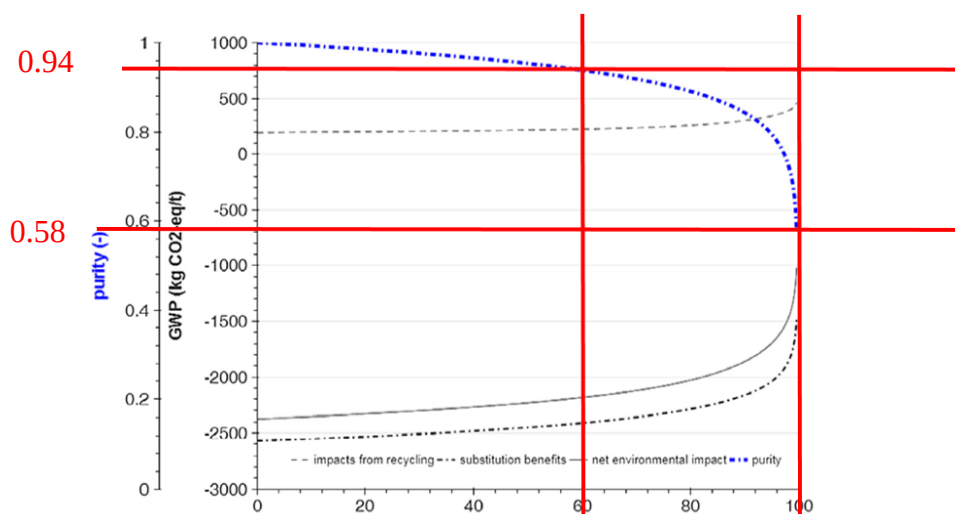


Figure Y.2 : Source separation vs. Impurities

Results of All Impact Categories

Climate Change

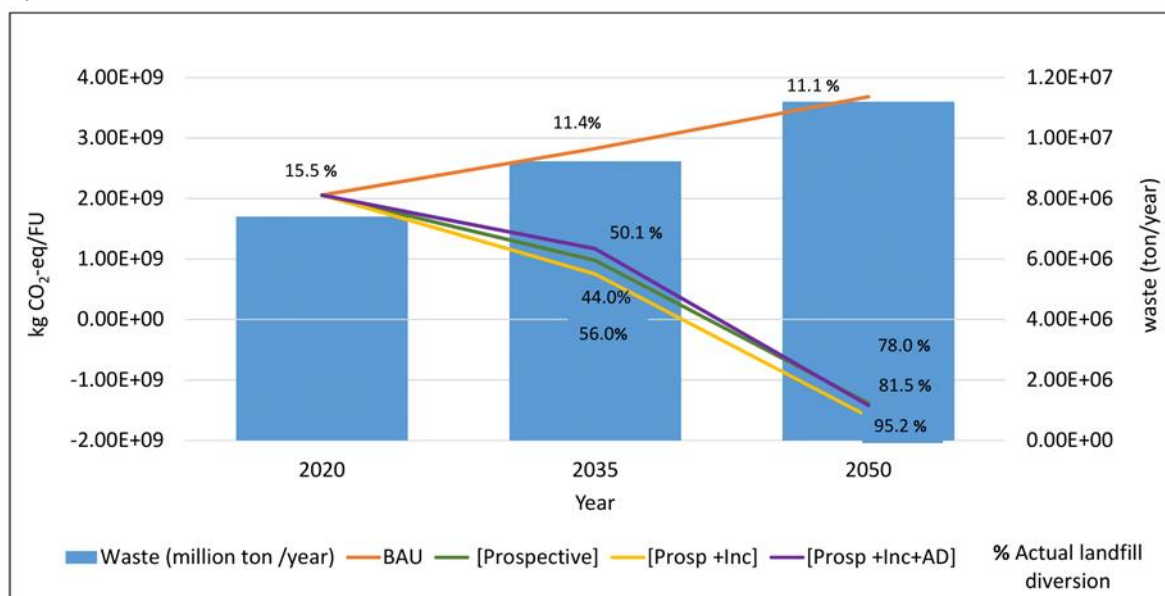


Figure Y.3 : Climate Change

Climate change decoupling analysis was done based on contribution analysis results. Decoupling analysis comparison of all impact categories were done by using normalized results.

Climate change

Phasing out landfill recommended before to close gap between WM sector emissions and sustainability policy goals (Magazzino, et. al., 2021). In addition (Astrup et al., 2015) showed that WtE is preferable to landfill in most of the cases. Hence, future scenarios in this study based on landfill diversion by using WtE technologies. Decoupling analysis of climate change category results show that coupling occurs with BAU scenario while decoupling occurs with due to WtE technologies. In addition, source separated waste effect on decoupling important topic to be addressed. Comparison of scenarios show source separation of packaging waste has significantly positive affect on decoupling. Overall results complying with previous study which was claiming waste recycling capacity and energy recovery, especially via incineration, are key points for decoupling of WM (Magazzino, et. al., 2020).

Landfill diversion was found to reduce emissions, and diverting food waste from WTE generally increased emissions. The analysis further found that when a 20 year GWP was used instead of a 100 year GWP, every scenario including WTE was preferable to every scenario including landfill. (Hodge et al., 2016).

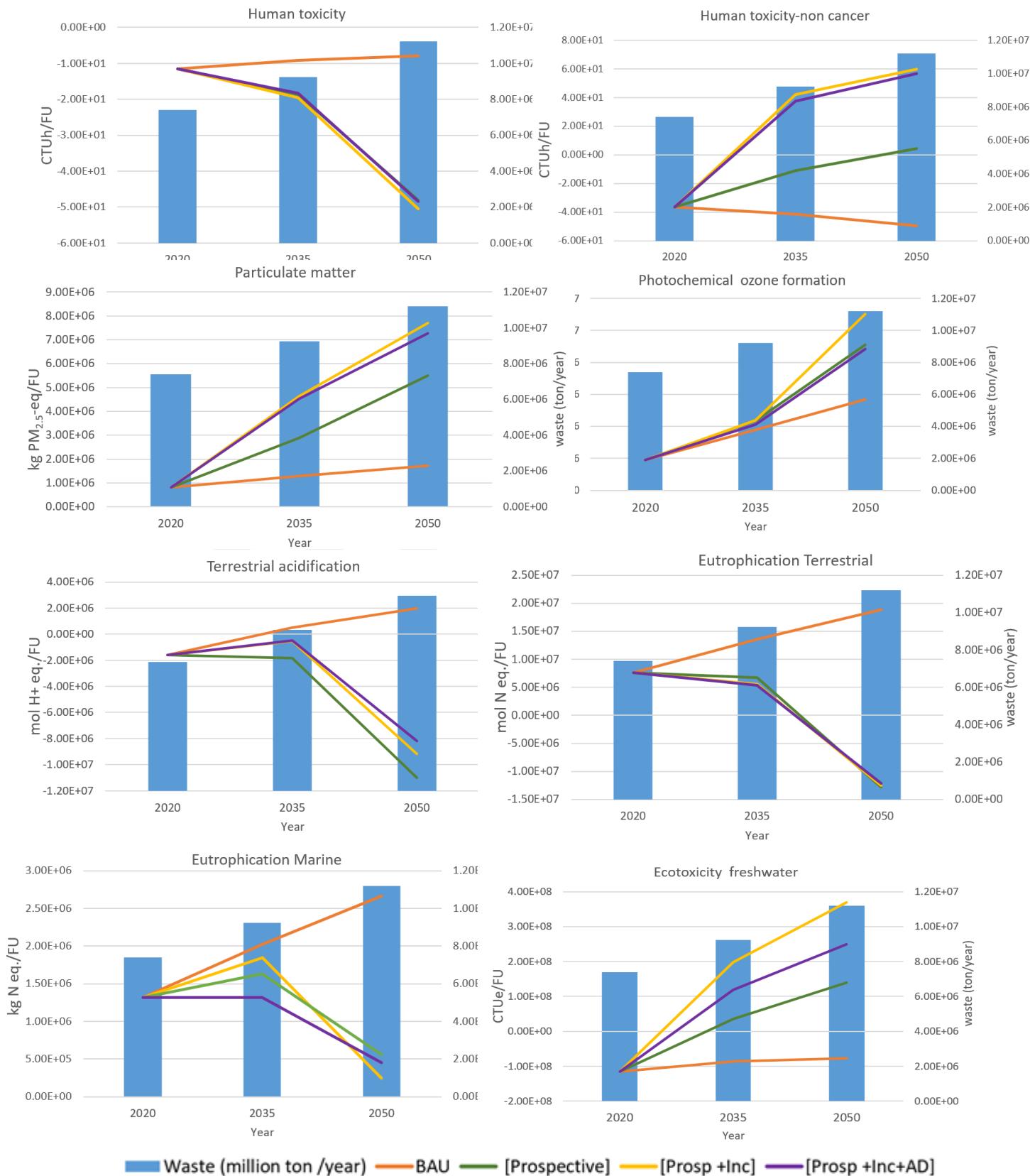


Figure Y.4 : Waste management emissions of different impact categories - Istanbul

GWP 20 YEARS EFFECTS IPPC 2021

The efficiency of landfill gas collection varies over time because the waste is landfilled continuously while the landfill is in operation (Wang et al., 2021). Net savings only achieved in 2050 due to high landfill burdens in GWP 20 year effect. There is a decrease in 2035, which leads to a decoupling of WM emissions, but that is not enough to meet net zero or net savings targets.

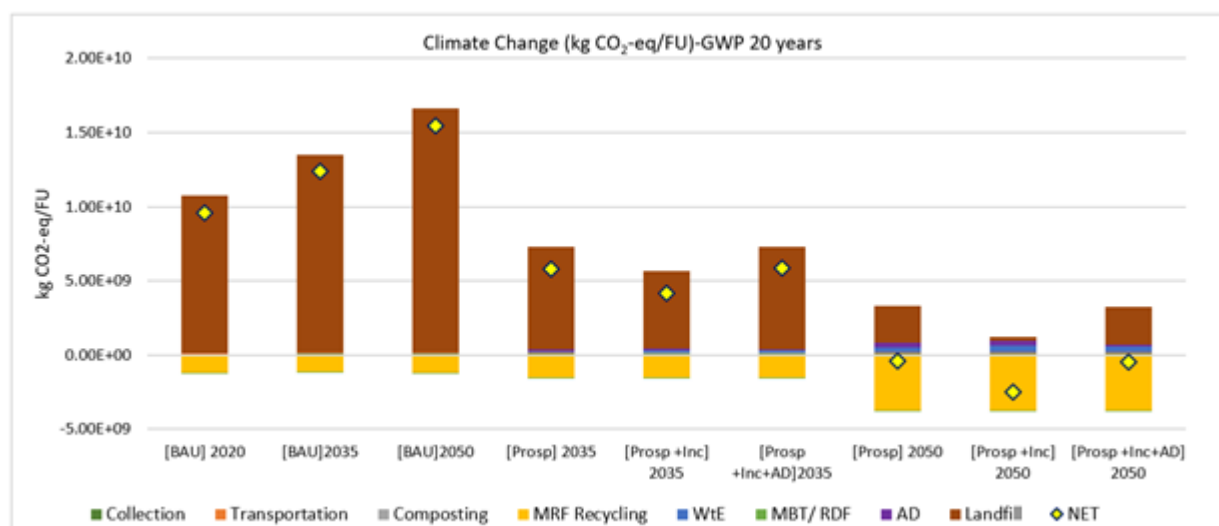


Figure Y.5 : Contribution analysis of WM scenarios- GWP 20 year

All future scenarios (except BAU scenario) have decoupling in GWP-20 year calculations. However, net impacts in BAU 2050 scenario almost 4.5 times of the GWP-100 years results due to higher burden coming from landfill. WtE technologies are essential to achieve decoupling of WM emissions over the short and medium term.

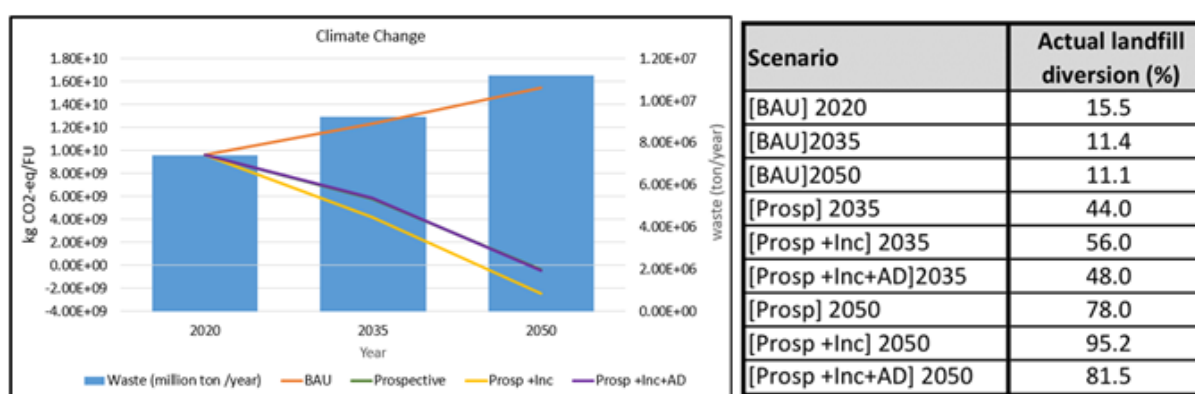


Figure Y.6: Net impacts (GWP 20 year) and landfill diversion (%) of the WM scenarios

APPENDIX Z: ECONOMIC INVENTORY AND CALCULATIONS

Table Z.1 : Cost Inventory of Istanbul Waste Management

Cost Inventory	Value	Reference
Biomass energy-incentive (€/kWh)	3.2	
Thermal waste treatment-incentive (€/kWh)	3.1	(Official Gazette of the Republic of Turkey 2021)
Landfill gas-incentive (€/kWh)	1.9	
Minimum wage (€/month)	347	(Official Gazette of the Republic of Turkey, 2020)
Electricity (€/kwh)	0.12	(Energy Map, 2020)
Natural Gas (€/kwh)	0.04	(IGDAŞ, 2020)
Waste transportation (€/ton)	9.6	(Primary data, 2020)
Transport- MRF (€/ton)	9	(IMM, 2020)
Transport- landfill (€/ton)	17	(IMM, 2020)
Collection (€/ton)	22	(IMM, 2020)
CAPEX		
AD (€/ton)	71	(IMM, 2012)
Incineration (€/ton)	85	(IMM, 2018)
Landfill (€/ton)	63	(Primary data, 2020)
Composting (€/ton)	72	(Primary data, 2020)
OPEX		
AD (€/ton)	55	(IMM, 2012)
Incineration (€/ton)	68	(IMM, 2018)
Landfill (€/ton)	40	(Primary data, 2020)
Composting (€/ton)	47	(Primary data, 2020)
Revenue		
Cardboard-paper (€/ton)	100	
Plastic bag (€/ton)	350	
Plastic (€/ton)	400	
Pet (€/ton)	400	Secondary Material Market (2020,2021 and 2022)
Metal (€/ton)	93	
Aluminium (€/ton)	1,750	
Glass(€/ton)	90	

Table Z.2 : Net Climate Change (kgCO₂-eq/ ton) and net cost of (€/ton) the scenarios

Scenarios	Total waste amount (ton/year)	Climate Change (kg CO ₂ -eq/ ton)	NET (€/ton)
[BAU] 2020	7,404,375	278	85 €
[BAU]2035	9,232,680	306	130 €
[BAU]2050	11,204,919	329	160 €
[Prosp] 2035	9,232,680	106	124 €
[Prosp +inc] 2035	9,232,680	81	127 €
[Prosp +inc-AD]2035	9,232,680	127	129 €
[Prosp] 2050	11,204,919	- 124	99 €
[Prosp +inc] 2050	11,204,919	- 143	122 €
[Prosp +inc-AD] 2050	11,204,919	- 127	107 €

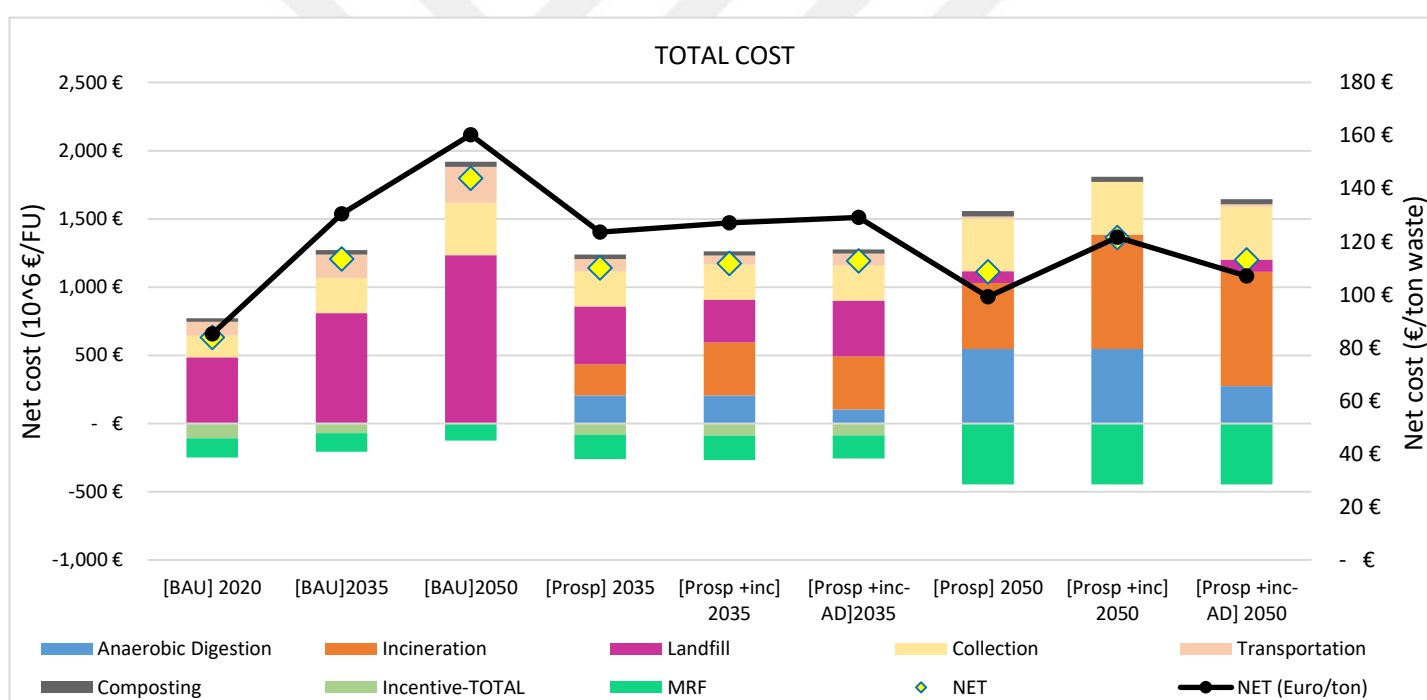


Figure Z.1 : Total cost of the scenarios (10⁶ €/year)

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